Modeling ecological risks at a landscape scale:
Threat assessment in the upper Tennessee River basin

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ABSTRACT

There is no single methodology toward freshwater conservation planning, and few analytical tools exist for summarizing ecological risks at a landscape scale. I constructed a relative risk model, the Ecological Risk Index (ERI), to combine the frequency and severity of human-induced stressors with mappable land and water use data to evaluate impacts to five major biotic drivers: energy sources, physical habitat, flow regime, water quality, and biotic interactions. It assigns 3 final risk rankings based on a user-specified spatial grain. In a case study of the 5 major drainages within the upper Tennessee River basin (UTRB), U.S.A, differences in risk patterns among drainages reflected dominant land uses, such as mining and agriculture. A principal components analysis showed that localized, moderately severe threats accounted for most of the threat composition differences among watersheds. Also, the relative importance of threats is sensitive to the spatial grain of the analysis.

An evaluation of the ERI procedures showed that the protocol is sensitive to how extent and severity of risk are defined, and threat frequency-class criteria strongly influenced final risk rankings. Multivariate analysis tested for model robustness and assessed the influence of expert judgment by comparing my original approach to a quantile-based approach. Results suggest that experts were less likely to assign catchments to high-risk categories than was the quantile approach, and that 3 final risk rankings were appropriate.

I evaluated the influence of land use on freshwater ecosystems by studying the relationship between land cover changes and the persistence of freshwater mussels. First, historical species data were collected and the Upper Tennessee River Mussel Database (UTRMD) was constructed. The UTRMD contains >47,400 species records from 1963-2008 distributed across nearly 2,100 sampling sites.

My study suggests that 30 years of land cover change does not explain observed freshwater mussel declines. Quantitative surveys are recommended basin-wide to provide more accurate information about mussel distribution and abundance. Lastly, results suggest that streams with repeated mussel surveys have increasing populations, including active recruitment in several beds. Additional quantitative surveys since 2004 have probably provided more accurate species and population counts, although actual population sizes are still uncertain.

DEDICATION

This dissertation is dedicated to my amazing patient family.

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I have much appreciation for everyone who has been part of this journey. My path has been unique, and longer than most, and the generosity and patience of those around me has been humbling. I could not have completed this degree without all of their support.

I acknowledge my committee first, because without their respectful support, I would not have been able to complete this journey. I especially want to thank my Committee Chair, Dr. Paul L. Angermeier, for his commitment to my success. I am forever grateful for his mentorship, persistence in encouraging the completion of this degree, and patience. Another special acknowledgment goes to Dr. Eric Hallerman, whose positive energy and optimism helped further my journey by inspiring me to finish. My other committee members, Stephen Prisley, Carlyle Brewster, and Mary Leigh Wolfe, have graciously stuck with me throughout, and I so appreciate their time, respect, and research support.

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I had countless meetings with several researchers and biologists in the completion of my final dissertation chapter. The chapter changed directions several times, and even veered off of the road once or twice. Only the encouragement of those around me got it complete.
I had no idea what a long process this would be when I began it. I was not prepared for ups and downs of research, but my committee saw to it that I would grow from the experience and succeed.

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This journey did not happen in exile, and many life events happened along the way. All of these events influenced my journey in one way or another. This is the point where the dedication and patience of my family must be acknowledged, especially my husband, Scott, and my amazing children, Davin, Beck, and Suvi. My family grew over the years, which prompted a longer timeline for the completion of each chapter. There have definitely been times when I didn't think the commitment was worth it, as my perspective changed often. I am so grateful for the not so subtle determination of my husband to see me finish my journey. I know this would not have happened without him, and he deserves many accolades. It has been quite an adventure.

As I complete my degree and look forward, I would like to acknowledge the beautiful landscape that I studied. The historical perspective of my research has given me an appreciation for time, which it turns out is as complicated as spatial analysis itself, as some aspects of a landscape seemingly remain static while change is obvious in others. The saying "time stands still" was apparent in watersheds that have seemingly remain intact over the years. On the other hand, "time flies" even in those watersheds, where aquatic communities have responded to surrounding human influences in numerous ways, giving me an appreciation for the complicated nature of conservation planning.

I hope that scientific inquiry and the interconnectedness of humans with our natural world have renewed interest within our society. It has been a pleasure working with schoolchildren of all ages and teaching them about scientific principles, especially about the scientific process and ecological principles. Without a new generation to carry on the movement that the great conservationists started, the work that I and my colleagues do is for naught. Our natural environment is in good hands.

TABLE OF CONTENTS

GENERAL INTRODUCTION…………………………………………………………………1

CHAPTER 1: Integrating human impacts and ecological integrity into a risk-based protocol for conservation planning 7

ABSTRACT………………………………………………………………………………………………7
INTRODUCTION…………………………………………………………………………………………8
The role of ecological integrity in risk-based assessments 9
Risk-based approaches in freshwater systems 11
Ecological Risk Index 14
Applying the ERI to the Upper Tennessee River Basin 18
METHODS 18
RESULTS 20
DISCUSSION 22
Informing Conservation Planning 23
ACKNOWLEDGMENTS 24
REFERENCES 25

CHAPTER 2: Robustness of rank-based risk assessments of freshwater ecosystems in conservation planning 47

ABSTRACT 47
INTRODUCTION 47
Ecological Risk Index 50
STUDY AREA 51
METHODS 52
Ecological Risk Index 52
Testing robustness 55
Comparison of frequency classification methods 55
Effects of expert opinion on severity scores 56
Appropriateness of threat risk categories 58
RESULTS 59
Frequency classifications and severity scores 59
ERI-C scores and ranks 60
DISCUSSION 62

Strengths and weaknesses 64
Informing conservation 67
Management Implications 68
ACKNOWLEDGMENTS 70
REFERENCES 71
APPENDIX 2.A Ranges of scores used in Monte Carlo simulations to assess effects of using severity measures based on expert judgment in the Ecological Risk Index. A triangular distribution was used. The severity score was the apex, and the minimum and maximum scores were the low and high ends of the triangular distribution. The smallest and largest scores among the experts are the minimum and maximum values, respectively. Severity scores were published previously (Mattson and Angermeier 2007) and may range from 5 to 15. NPDES stands for National Pollutant Discharge Elimination System. 95

CHAPTER 3: A freshwater mollusk database to facilitate research and conservation: knowing the past to plan the future 96
ABSTRACT 96
INTRODUCTION 97
Database questions 99
STUDY AREA 100
METHODS 103
Database Structure 103
RESULTS 105
DISCUSSION 106
MANAGEMENT IMPLICATIONS 109
ACKNOWLEDGMENTS 111
REFERENCES 112
APPENDIX 3.A. Freshwater mussel counts for mussels found within the Clinch River basin, USA from 1963-2008. 121
APPENDIX 3.B. Sample of the agencies and organizations contacted, description of available data, and issues related to individual datasets that were identified as eligible for import into the Upper Tennessee River Mussel Database. 124

CHAPTER 4: Streams, spatial scale, and human impacts: how data availability impedes freshwater mussel conservation 126
ABSTRACT 126
INTRODUCTION 128

METHODS 132
Study Area 132
Freshwater mussel species data 133
Land cover change 134
Impacts of land use on mussel populations... 135
RESULTS 136
Mussel distribution and abundance... 136
Land cover change 138
Impacts of land cover change on mussels... 140
DISCUSSION 142
Mussel distribution and abundance... 142
Observed mussel patterns and data quality... 142
Influences of spatial grain on observed land cover patterns 143
Impacts of land and water use on mussels 145
Conservation implications 146
CONCLUSIONS AND RECOMMENDATIONS 150
ACKNOWLEDGEMENTS... 151
REFERENCES 151
GENERAL CONCLUSIONS... 175
Ecological Risk Index 177
Testing the Ecological Risk Index 178
Database considerations.. 180
Land cover change and freshwater mussels.. 182
CONCLUSIONS... 183
REFERENCES 184

LIST OF TABLES

Table 1.1. Comparison of terms used in the continuum of risk-based analyses. 34

Table 1.2. Matrix of severity ranks (low=1, moderate=2, and high=3) for major threats within the Upper Tennessee River Basin (UTRB). 35

Table 1.3. Frequency scores (0- not present, 1-minimum, 2-moderate, and 3-maximum impact) used to compute risk indices for 12 major threats within the Upper Tennessee River Basin (UTRB). Integer frequencies are the actual number of occurrences in a given subunit. Land cover represents the percent of area in a given subunit. Equal interval classes were used when no related risk-based studies were found. 36

Table 1.4. Loadings of major threats on the first two principal components (PCI and PCII) of 107 subunits within the UTRB. 37

Table 1.5. Mean frequencies and variances of 12 threats in watersheds of major drainages of the UTRB. The number of subunits (N) in each drainage is also shown. 38

Table 2.1. Test of homogeneity of Monte Carlo simulation results (10,000 iterations) comparing ERI-C scores computed via the original ERI approach versus a quantile-based approach. 79

Table 2.2. Group means for 3 risk levels (low [1], medium [2], high [3]) computed from final ERI-C values for the original ERI and quantile-based approaches 80

Table 2.3. Results of backward-stepwise discriminant analysis of ERI-C scores, which were computed via the original ERI approach and a quantile-based approach 81

Table 2.4. Canonical discriminant functions for ERI-C scores computed via the original ERI and quantile-based approaches. 82

Table 3.1. Field codes and descriptors compiled in the upper Tennessee River Mussel Database (UTRMD). Field codes refer to the main data categories within the UTRMD, and individual variables associated with each category are listed below them. The type of information required for each variable is listed in the Data Required column. There are 2 input methods in the UTRMD. User input refers to when the user enters new data. Drop-down menus provide input choices for the user as either a list or button choice (Yes/No). UTM stands for universal transverse mercator. CPUE is defined as catch per unit effort. 117

Table 4.1. HUC12s containing freshwater mussel collection sites in both early (1963-1985) and late (1998-2008) time periods, within the Clinch River, USA. 160

Table 4.2. Statistics of comparing land cover between 1980 and 2001 within riparian areas of ten HUC12s containing freshwater mussel collection sites. Standard deviations and means of land coverages are shown. 161

Table 4.3. Wilcoxon signed rank test comparing mean mussel counts between early (1963-1984) and late (1998-2008) study periods for riparian areas within 2 km of freshwater mussel collection sites. 162

Table 4.4. Mussel species with active recruitment within the Clinch River basin since 2004. Data are from quantitative and qualitative surveys throughout the basin. Conservation status (G1 = critically imperiled, G2 = imperiled, G3 = vulnerable, G4 = apparently secure, G5 = secure) was obtained from the NatureServe database (http://www.natureserve.org/explorer/index.htm). Species in bold have greater counts today than in the early (1963-1984) sampling period. 163

LIST OF FIGURES

Figure 1.1. Schematic representation of risk-based assessments along gradients of spatial scale and methodological rigor. Horizontal position reflects the relative importance of quantitative analysis. Vertical position reflects spatial extent. Each ellipse represents one or more published assessment approaches. Refer to legend for letter explanations. 39

Figure 1.2. Flowchart depicting major steps in developing the Ecological Risk Index within a threat assessment framework. All major threats within each subunit are summarized and maps are produced with final ranking scores to allow visual comparisons. 40

Figure 1.3. The upper Tennessee River basin, USA Delineation is of 107 subunits based on 4^{th}-order streams. Major river drainages are depicted (shaded regions). 41

Figure 1.4. Maps of threat-specific indices of ecological risk (ERI-T) for 12 anthropogenic threats in 107 subunits within the UTRB.. 44

Figure 1.5. Maps of subunits within the UTRB with low, moderate, and high values of a composite index of ecological risk (ERI-C) to aquatic ecosystem integrity. 45

Figure 1.6. Ordination of 107 subunits in principal components space defined by indices of threat-specific ecological risk. Subunits are labeled by their composite index of ecological risk (ERI-C). Some points overlap.. 46

Figure 2.1. Flowchart summarizing the Ecological Risk Index (ERI) protocol Threat data may originate from various sources including land use/cover maps, point-source and nonpoint-source pollution, and other local and regional sources of anthropogenic stressors 83

Figure 2.2. Map of the Upper Tennessee River Basin (UTRB) located in the southeastern USA. Catchments depicted are based on 4^{th}-order streams and above (Strahler method). Only major streams are shown. The five major drainages within the UTRB are shown............................... 84

Figure 2.3. Summary of a Monte Carlo simulation. (10,000 iterations) of 107 ERI-C scores computed from the original ERI approach (light grey) and a quantile-based approach (dark grey). Percentages of scores in low- (1), moderate- (2), and high-(3) risk levels are shown. One-way ANOVA tests were significant ($P < 0.001$) for each risk level. 85

Figure 2.4. Comparison of final risk rankings derived from ERI-C scores for the original ERI. approach versus a quantile-based approach for computing threat-frequency classes. ERI-C scores are aggregated into low-risk (1), moderate-risk (2), and high-risk (3) levels to facilitate mapping. Number of catchments in each risk level is in parentheses. 86

Figure 2.5. Classification of risk groups using goodness-of-fit and jackknife results for the original ERI and quantile-based approaches. Risk groups refer to the number of clusters obtained from *K*-means cluster analysis. 87

Figure 2.6. Plots of canonical variable scores from discriminant analysis for 107 ERI-C scores. computed from (a) the original ERI approach, and b) a quantile-based approach. 95% confidence ellipses for scores are aggregated by low-, moderate-, and high-risk levels. For a), factor 1 represents a continuum from urban-related threats to bridges and road density (from left to right) and factor 2 represents industrial-related activity to railroad density and waste treatment facilities. Factor 1 in the quantile approach (b) represents developed areas to road density and pasturelands, and factor 2 is influenced by site-specific data such as impoundments and manufacturing sites to area-based threats such as agriculture and road density. 88

Figure 2.7. Boxplots from *K*-means cluster analysis of final risk rankings for twelve threat types in 107 catchments of the upper Tennessee River basin. Each graph shows one threat; the *x*-axis refers to the low-, moderate-, and high-risk levels for the original ERI (left panels) and quantile-based approach (right panels), and the *y*-axis depicts the standardized frequency scores for each individual threat. Notches indicate medians, and box lengths are confidence intervals (95th quantile). Asterisks are outliers within three times the length of the inter-quartile range, and circles are far outliers. NPDES stands for National Pollutant Discharge Elimination System. .. 94

Figure 3.1. The five major watersheds of the upper Tennessee River system, USA................ 118

Figure 3.2. Initial screen of the Upper Tennessee River Mussel Database (UTRMD). This image shows the drop-down format for author(s), a format consistent throughout the input screens. The start and end dates of a field survey, publication year, and citation information are all required fields. The user may also enter a new author name using the "Add Author" user input tabs (First name, Last name). Also shown on the left-side menu are options for entering a new survey, editing a previously entered survey, importing/exporting data, and user information. 119

Figure 3.3. Freshwater mussel status within 87 12-digit hydrologic units across the Clinch-Powell River basin, USA Those areas with active recruitment are also considered to have viable mussel beds, but the presence of mussel beds does not necessarily indicate active recruitment. Historical mussel surveys were collected from 1963-2008, and active recruitment refers to those mussel beds with juveniles present within 1998-2008. Field survey sites correspond to those hydrologic units containing mussel beds 120

Figure 4.1. The Clinch River basin is located in southeastern United States, and composes part of the upper Tennessee River basin Three hydrological-unit spatial grains are shown (HUC8, HUC10, and HUC12). Strahler 3rd-5th stream orders are shown at the HUC12 level, and mussel survey sites from 1963-2008 are indicated (black circles). Land use changes (1980-2001) are shown at each hydrologic unit level as varying shades of gray corresponding to no land conversion (no shading) to maximum land conversion (darkest shade: HUC8=5.6%, HUC10=5-8.75%, HUC12=15-20%). 164

Figure 4.2. Numbers of freshwater mussel field surveys during 1963-2008 within the Clinch River basin, USA 165

Figure 4.3. Frequencies of qualitative (black) and quantitative (gray) collection methods used at freshwater collection sites throughout the Clinch River basin during 1963-2008........ 166

Figure 4.4. Numbers of adult (black) and juvenile (gray) freshwater mussels collected during 1963-2008 within the Clinch River basin, USA Data are based on the numbers of live adult and juvenile mussels found in both qualitative and quantitative collections. Adult mussels were found in every collection year, ranging from 2 mussels in 1998 to 14,270 in 1979. Number of juveniles ranged from 0 in 19 of the 25 years, with collections of up to 356 in 2007 167

Figure 4.5. Estimated freshwater mussel population (all species) from 1963-2008 within the Clinch River Basin, USA Population trend is based on the number of live adult and juvenile mussels found using both qualitative and quantitative collection methods. 168

Figure 4.6. Land cover changes throughout the Clinch-Powell River basin, USA. Changes between 1982 and 2001 are represented as percentage of land cover differences on a pixel-by-pixel basis. Land cover change is represented by white through dark grey: <2% (white), 2- 5% (light grey), 6-10% (medium light grey), 11-15% (medium dark grey), and 16-20% (dark grey). Freshwater mussel collection sites are shown as dots. HUC12 numbers are shown for reference. 169

Appendix 4.A. Demographic trends for particular freshwater mussel species within 10 HUC12s of the Clinch River basin, USA 170

GENERAL INTRODUCTION

Conservation planning focuses primarily on conserving species and ecosystems of interest within an ecologically sustainable management program (Groves et al. 2002, Abell et al. 2002). In freshwater ecosystems, the process aims to record the impacts of modern society on aquatic ecosystems, and prepare plans for protecting biodiversity by sustaining the natural physical, chemical, and biological components that contribute to viable populations of representative conservation targets. The planning process often requires using individualized approaches, such as the use of satellite imagery or rank-based assessments, for quantifying the impacts of human actions, especially when retrieving data may be time consuming or cost-prohibitive. Furthermore, freshwater conservation planning often encompasses the protection of large watersheds in which reserve-based conservation is infeasible, and management goals must balance conservation and economic goals.

Threats associated with working landscapes may modify management objectives throughout a watershed based on where streams and rivers have been minimally or heavily altered by human actions. For example, forested riparian areas and undeveloped floodplains rank as least impacted (Omernik et al. 1981); whereas, human-dominated areas with impervious surfaces and industrial uses are commonly considered to have poor conservation value (Wang et al. 2001). Various land and water uses alter the ecological integrity, or natural variations, of a stream at multiple spatial scales, and watersheds are typically characterized by their degree of human dominance when land cover data are considered. A major goal of freshwater conservation planning is to mitigate the impacts associated with land and water uses and other human-related impacts, but it is often difficult to quantify the severity (numbers) and scope (magnitude) of those impacts, especially when their spatial configuration is considered (Turner et al. 1996, Wiley et al. 1997, Hughes and Hunsaker 2002).

The risk assessment field provides a foundation for using a functional approach in freshwater conservation planning. Ecological risk assessment has historically included following specific and measurable pollution events (e.g. an oil spill), but this approach is implausible for entire watersheds where multiple threats occur and impacts vary. Applications of risk assessment in watershed-based studies have recognized that alternatives to individual stressor assessments are needed (Barve et al. 2005), especially innovative conservation planning approaches that combine risk assessment tools with a holistic approach to conserving system functionality.

A common holistic approach to risk assessment is to use a ranking system to identify threats and rank their impacts. For example, land conversion occurring within a riparian zone may be ranked as having maximum impacts to a stream, but the impact may be ranked as minimal if the riparian zone is unaffected. Individual ranking methods vary, but there are commonalities among rank-based risk assessments that make them appealing to conservation planners. First, watersheds may be evaluated across different spatial (e.g., riparian versus watershed) and/or temporal (e.g., recent versus historical) scales to differentiate between human-related impacts and natural variations (Smogor and Angermeier 1999). Second, readily available data, such as satellite imagery, may be substituted for field-collected data within the planning process. Third, expert opinion may be used within rank-based assessments to provide information about threat severity and scope that may otherwise be unknown or unobtainable within a reasonable timeframe. These last two steps are imperative for watersheds having high conservation priority but with few resources for intensive data collection and analysis. Lastly, these approaches aim to inform regional planners about localities with least/most potential for protecting conservation targets so that local management decisions might be more informed and effective.

The overall goals of my dissertation research were to contribute a practical, widely applicable approach to integrating risk into freshwater conservation planning and to model ecological risks at a landscape scale. These goals were accomplished by 1) constructing a framework for freshwater conservation planning that includes an integrated protocol for assessing stressors associated with biotic health, 2) validating the risk ranking procedures within the protocol and characterizing its robustness to overall risk classification, and 3) relating historical impacts of local and regional land uses to freshwater mussels, a host of species sensitive to many changes in ecological integrity and widely imperiled, within the conservation planning framework.

I used a unique and biologically diverse catchment as a case study for my research. The upper Tennessee River basin (UTRB), USA, has been identified as a conservation priority due to its unique geomorphology, but also due to a loss of freshwater mussel species over the past few decades, as well as for declines in fish species and an increase in anthropogenic stressors (Abell et al. 2000, Hampson et al. 2000). Although the basin was once home to the greatest diversity of freshwater mussels in the United States, it currently has the greatest number of imperiled species per unit area in the continental U.S. (Hampson et al. 2000). My research aimed to quantify the threats throughout the basin and provide a framework for future conservation planning, especially since human activities have caused a continual decline in ecological integrity despite minimal development (Neves and Angermeier1990; Bolstad and Swank 1997; Diamond and Serveiss 2001; Diamond et al. 2002).

The process used to achieve my goals began with the construction of the Ecological Risk Index (ERI), which uses a rank-based risk assessment approach to model the impacts within a catchment and map risk regions (Mattson and Angermeier 2007). The ultimate purpose of the ERI is to empower managers with a tool that uses readily available spatial data for the purposes of protecting species diversity and retaining biological integrity. Chapter 1 describes the ERI, including its purpose, components, and rationale within freshwater conservation planning. Threats within the UTRB are identified, and their extent (scope) and severity are quantified. The ERI uses threat frequency as the extent component, and severity is a catchment-specific expert-derived score. One of the protocol's outputs is a map of risk zones (low, moderate, high) across a catchment, providing managers with a spatial reference for conservation planning.

Since the ERI is a novel rank-based risk assessment approach, there was a need to validate its methods and final risk map. I undertook this task in Chapter 2 by comparing the ERI methods to a purely statistical approach and comparing results (Wiegers et al.1998). Since expert opinion is one of its components, the ERI may be subjected to biases. Therefore, this chapter addresses those biases by replacing expert opinion with an objective approach. There are several parameters within the ERI that were tested: severity score assignment, frequency classification, and the precision of the final risk rankings. The functionality of the ERI depends on its ability to adequately distinguish between low, medium, and high risk areas, so this chapter focuses on the use of expert opinion in rank-based risk models, and answers the question of whether expert-derived assessments provide insight into such models.

Historical perspectives

I shifted focus in Chapters 3 and 4 to study the relationship between land cover changes within a portion of the UTRB and the persistence of freshwater mussels. With this in mind, I led a team that used six decades of field-collected freshwater mussel species data to construct a large online database, the Upper Tennessee River Mussel Database (UTRMD). In Chapter 3, I mapped out survey sites, mussel bed locations, and areas with active recruitment to determine overall mussel conservation status, and the work reported in Chapter 4 focused on relating changes in mussel species numbers, recruitment, and population fluctuations to land cover changes in local and catchment-scale assessments.

My priority in Chapter 3 was to construct a secure, online database that would provide information relating to historical and current mussel species locations and conditions. This chapter is a description of the database itself as well as an account of the process of developing a database and the issues involved in making it a success. The main tasks of this chapter included:1) collating historical freshwater mollusk data from across the upper Tennessee River basin; 2) constructing a comprehensive freshwater mussel species database, and; 3) discussing issues pertaining to constructing a central archive such as the UTRMD. Ultimately, a chapter goal was to make the database accessible to both regional mussel researchers as well as users who may be interested in linking regional, national, and global biodiversity patterns (Darwall et al. 2008).

Streams, spatial scale, and human impacts

The Clinch River basin is renowned for its historically large mussel diversity, but also for having one of the greatest losses in aquatic biodiversity within the U.S. My purposes in chapter 4 were to: 1) describe historical land cover patterns within watersheds of the Clinch River basin; 2) relate spatiotemporal patterns in freshwater mussel distributions to patterns in riparian land cover; 3) evaluate the impacts of human activities on species assemblages within existing mussel beds, and 4) discuss the conservation value of decades of freshwater mussel data.

The foundation for this chapter is that anthropogenic disturbance plays a critical role in the ability of a system to retain its functionality (Poiani et al. 2000), and is the cause of many freshwater species imperilments. For example, in the southeastern United States, mining, agriculture, and logging have degraded streams and caused population declines of many endemic species (Strayer and Malcolm 2012). Efforts to determine how the juxtaposition and patchiness of disturbance sites affect conservation efforts have met with mixed results (Diamond and Serveiss 2001, McRae and Allan 2004, Regnier et al. 2009), and an increasing emphasis in recent years on the effects of terrestrial processes on freshwater resources has highlighted the importance of focusing on freshwater conservation to maintain overall ecosystem health (Baron et al. 2002).

Since the Clinch River basin was evaluated in each chapter, comparisons may be made between the final risk rankings of the ERI approach and changes in land cover and mussel populations. The Clinch River basin has historically had consistent mining-related activities for decades, but few land cover changes. The ERI reflects this land use by ranking most of the basin as having moderate risk to the aquatic ecosystem. Since land uses have been consistent over time, there has been little land conversion, and mussel bed activity also appears to have responded to land use. For example, those beds with active recruitment are located in streams with fewer risks or at greater distances from land uses considered high risk than those beds containing only adult mussels. This indicates that land uses are impacting streams throughout the Clinch River basin at varying degrees, and land use is most likely a higher management priority than land conversion.

Chapter 1: Integrating human impacts and ecological integrity into a risk-based protocol for conservation planning

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ABSTRACT

Conservation planning aims to protect biodiversity by sustaining the natural physical, chemical, and biological processes within functional ecosystems. Often, data to measure these processes are inadequate or unavailable. The impact of human activities on ecosystem processes complicates integrity assessments and may alter ecosystem organization at multiple spatial scales. Freshwater conservation targets, such as populations and communities, are influenced by both intrinsic aquatic properties and the surrounding landscape and, hence, locally collected data may not accurately reflect all potential impacts. Changes in five major biotic drivers: energy sources, physical habitat, flow regime, water quality, and biotic interactions, may be used as surrogate metrics to inform conservation planners of the ecological integrity of freshwater ecosystems. Threats to freshwater systems may be evaluated based on their impact upon these drivers to provide an overview of potential risk to specified conservation targets. I developed a risk-based protocol, the Ecological Risk Index (ERI), to identify watersheds with least/most risk to conservation targets. The protocol combines risk-based components, specifically the frequency and severity of human-induced stressors, with biotic drivers and mappable land- and water-use data to provide a summary of relative risk to watersheds. I illustrate application of my protocol with a case study of the upper Tennessee River basin, U.S.A. Differences in risk

patterns among the major drainages in the basin reflect dominant land uses, such as mining and agriculture. A principal components analysis showed that localized, moderately severe threats accounted for most of the threat composition differences among my watersheds. I also found that the relative importance of threats is sensitive to the spatial grain of the analysis. My case study demonstrates that the ERI is useful for evaluating the frequency and severity of ecosystem-wide risk, which can inform local and regional conservation planning.

INTRODUCTION

Conservation planning focuses primarily on conserving species and ecosystems of interest within an ecologically sustainable management program (Groves et al. 2002, Abell et al. 2002). The goal of these efforts is to protect biodiversity by sustaining the natural physical, chemical, and biological components that contribute to viable populations of species or other representative conservation targets. Planning procedures currently identify conservation targets by using abiotic and biotic entities to represent biodiversity in large geographic areas, while landscape metrics, population data, and minimum dynamic area measures are employed to evaluate the ability of conservation targets to persist (Groves 2003). Some applications, such as the National Gap Analysis Program (GAP), use a species-based approach in determining appropriate conservation areas based on factors that include habitat availability and existing networks of protected lands (Jennings 2000). Only rarely do current assessments for conservation planning explicitly integrate stressor impacts with projections of target persistence even though the intensity of such disturbances can profoundly alter persistence. Including impact assessment in conservation plans could clarify the negative effects of human-induced risks on conservation targets and biodiversity, and enhance the long-term effectiveness of conservation planning.

Retention of ecological processes is essential for successful freshwater conservation, and assessing the magnitude of stressors on ecological condition is a key step in developing freshwater conservation plans (Cowx 2002, Groves 2003). Freshwater streams are influenced by intrinsic structures and functions as well as the surrounding landscape, making conservation actions complicated because both the regional context and local disturbances affect ecological integrity (Roth et al. 1996, Allan et al. 1997). Impacts to ecological integrity follow the amount of disturbance within a system, as measured by human-induced drivers that negatively alter ecosystem structures and functions. Initially, I intended to integrate established risk assessment methods into freshwater conservation planning, but found that these methods do not evaluate ecosystem drivers or use ecological integrity as an endpoint in assessing risk. With this in mind, I developed a risk-based approach to inform regional planners about the localities with least/most potential for protecting conservation targets so that local management decisions may be more informed and effective.

In this paper, I integrate ecological risk assessment with landscape ecology principles to build a tool for use in freshwater conservation planning. First, I outline links between biotic conditions and risk-based assessments. Second, I evaluate existing approaches that use risk-based assessments at local and regional scales. Third, I introduce a protocol for assessing ecological risk of human activities on stream systems. Finally, I discuss application of my protocol to the upper Tennessee River basin (UTRB) in the southeastern United States and its potential applicability to other regions.

The role of ecological integrity in risk-based assessments

Human land use affects both local and regional assessments of stream conditions (Roth et al. 1996, Lammert and Allan 1999, Wang et al. 2001). Impairments to the ecological integrity of streams may be classified using physical, chemical, and biological components collected locally (Detenbeck et al. 2000) or summarized by region (Hughes and Hunsaker 2002). Certain landscape metrics, such as patch size and interspersion, describe causal links between regional land use patterns and local stream conditions (O'Neill et al. 1997, Hughes and Hunsaker 2002); however, specific impairment pathways are often unknown, making causal links difficult to confirm.

Risk assessment estimates the likelihood of exposure of an endpoint to a stressor (Table 1.1), and the purpose of most risk assessment studies is the quantitative assessment of the likelihood and severity of alterations in function and condition of selected endpoints (e.g. Mebane 2001, Preston and Shackleford 2002). Common to all watershed-based risk assessments is the goal of estimating the variability in magnitude of stressor impacts. Unfortunately, such estimation is often infeasible with empirical quantitative evidence, especially at large spatial scales (O'Neill et al. 1997). Stressors should be easily identified and impacts quantifiable if conventional risk assessment approaches are to be applied at regional scales.

Protecting ecological integrity is the ultimate goal of conservation planning, and executing risk-based assessments requires consideration of potential declines in physical, chemical, and biological components. The effects of stressors on ecosystem integrity may be assessed using determinants of biological degradation. A system has ecological integrity, in part, if its drivers have not been altered by humans (Karr and Dudley 1981, Karr et al. 1986). Conversely, a system's integrity is compromised to the extent that its drivers and responding biotic attributes deviate from natural reference conditions. This notion of integrity is widely used as a conceptual foundation for assessing local and regional stream conditions and for comparing impacts across watersheds (Karr and Chu 1999). Recognition of the relationship of human impacts to ecological integrity is essential for proper watershed assessment and management.

I suggest that ecological integrity is an appropriate assessment endpoint for evaluating the risk of human impacts to stream systems, much like biotic integrity is already used as an endpoint for assessing the impacts themselves. Alterations in the major drivers (i.e., energy sources, physical habitat, flow regime, water quality, and biotic interactions) of freshwater systems ultimately affect species distributions and abundances, and such drivers may be used as surrogates to evaluate overall ecological integrity (Karr et al. 1986, Poff et al. 1997). Stream monitoring at local scales documents changes in biotic drivers, indicating adverse effects from land or water uses. When applied to larger spatial scales, data from local studies may aid in estimating potential stressor impacts (Lammert and Allan 1999). Focusing on biotic drivers during risk assessment emphasizes the importance of sustaining ecosystem functions in order to minimize loss of valuable populations and communities (Walker 1992, Baron et al. 2002). Applying threat evaluation to a framework for freshwater conservation planning, and specifically threats to watersheds, may provide a means for assessing stressor impacts and aid in identifying areas where conservation efforts would be most cost-effective.

Risk-based approaches in freshwater systems

Stressor impacts on stream systems have been studied across a range of hierarchical scales without a consensus on the most appropriate spatial scale(s) or techniques for predicting system responses (Lammert and Allan 1999). Risk-based approaches have been applied to regional analyses by incorporating multiple endpoints, such as shoreline habitat and instream condition, that may be affected by a variety of known risks (Wiegers et al. 1998, Norton et al. 2002). Conventional single-stressor versus single-endpoint relationships become impractical at larger spatial extents as focus shifts to cumulative impacts of multiple stressors (Molak 1997). Locally collected physical, chemical, and biological data are often used to identify impacts to streams and rivers (Cormier et al. 2002, Norton et al. 2002), and may be aggregated to identify multiple endpoints and summarize risk within watersheds (Suter and Barnthouse 1993).

I reviewed the meanings of terms commonly used in risk-based analyses and sought shared features that could be applied at regional scales (Table 1.1). I used accepted definitions in my evaluation to compare similarities and differences between risk and impact assessment (Stem et al. 2005). In particular, I defined components appropriate for use in regional studies, and identified established methods that may include evaluation of ecological integrity as an endpoint. Herein, I describe how study focus, statistical tools, and data collection differ at various spatial extents. I chose representative studies from the literature to summarize approaches used for risk-based assessments over a range of spatial scales (Figure 1.1).

Methods to assess human-induced disturbances that negatively affect ecosystem functions and processes may be ordered along a quantitative-to-qualitative axis as well as along an axis of spatial extent (Figure 1.1). The quantitative to qualitative axis ranged from the use of randomized, replicated experimental designs for measuring toxic effects on specific populations (i.e., very quantitative) to studies without replication that characterize effects of multiple stressors (i.e., very qualitative). The spatial scale axis spanned from small spatial extents (i.e., individual stream reaches) to region-wide study units (e.g. drainage basins). The juxtaposition of approaches along these axes provides insight into their utility for large-scale conservation planning. Representative studies form a positive relationship displaying the increasing reliance on qualitative measures as the spatial extent increases. I found no studies that focused on a single stressor at larger spatial scales and, conversely, studies at small spatial scales did not rely on risk ranking or qualitative summaries.

As expected, detailed parameterizations and causal links were most often sought at smaller spatial extents (Moore 1998, Rabeni 2000, Suter et al. 2002), whereas correlations between stressors and degradation became more common as the study scale increased. At large spatial extents, methods included assigning scores to land use and land cover to reflect positive or negative influences on biota (Bryce et al. 1999, Wiegers et al. 1998, Walker et al. 2001), ranking impacts based on risk classes (Slob 1998), and using ranks in land use intensity or land cover change to compare watersheds (Turner et al. 1996, Detenbeck et al. 2000). These studies relied on abiotic factors for assessing impacts to watersheds, and hydrologic data were also commonly used to explain structural components, such as species composition and habitat availability, and functional attributes such as water quality (Muhar and Jungwirth 1998, Bryce et al. 1999).

In summary, much recent work applied risk-based approaches to conservation planning, but there are not yet any standards. Data requirements and analyses necessarily differ among spatial scales, and one approach does not appear to be more advantageous than others. Stressor impacts on stream systems were most commonly measured through the impacts upon populations of fishes and macroinvertebrates, or on water chemistry and physical habitat data. Although risk characterization was ubiquitous, results were often linked to land use patterns as systems became more complex (Hughes et al. 2000, Muhar et al. 2000, Slob 1998, Walker et al. 2001). I found that although risk-based approaches worked well for their intended purposes, none explicitly addressed the consequences of human actions on the major determinants of biological degradation. I found that the basic tenets of risk assessment, namely that risk is estimated based on threat frequency and severity of harm within the system of interest, were not explicitly applied as the spatial extent increased.

Ecological Risk Index

I gleaned three key concepts from my literature review as a foundation for threat assessments. First, sources of stress within a system were identified, regardless of their likelihood of occurrence, with respect to their effects on a specific endpoint. Next, threats were commonly weighted according to frequency, and impacts pertained to structural and functional properties of ecosystems. Lastly, aggregates of locally collected data were useful in identifying regional threat patterns. I used these concepts to build a protocol for assessing the impacts of anthropogenic stressors on the ecological integrity of watersheds.

My protocol, the Ecological Risk Index (ERI), integrates the frequency of various land uses with estimates of their potential impact on biotic drivers (Table 1.1). Briefly, the ERI uses a ranking procedure to identify areas of low, moderate, and high risk to stream biota based on the frequency and potential harm of identified threats to the flow regime, physical habitat, water quality, energy sources, and biotic interactions of a freshwater system. I incorporated two aspects of risk assessment, frequency and severity, into my protocol. Frequency, defined as the number of individual threats, was used to indicate observed intensity of human land and water use. Severity, defined as the potential impact of a stressor on ecological integrity, was used to indicate the expected magnitude of changes in biotic drivers independent of threat frequency. These definitions are analogous to those used in disturbance ecology, in which frequency and severity are often used to describe the extent and magnitude of an event on a natural system (Turner and Dale 1998).

The ERI uses readily available data to identify geographic areas that may respond cost-effectively to conservation efforts. This approach serves two purposes. First, it recognizes the difficulty in collecting standardized field data over large spatial extents. Instead, national databases of land cover and use-related data are used as surrogates for field-collected data. For example, toxins are released from roads in two ways: truck spills and surface runoff (Forman and Alexander 1998). Spills are sporadic and unpredictable, with damaging incidences occurring on or near bridges. Bridge data, therefore, were deemed appropriate surrogates for spills. Second, informative, readily available data provide a cost-effective means of representing complex relationships. Biological effects of road runoff are a function of distance to stream and the juxtaposition of other landscape features (Forman and Alexander 1998). Measuring runoff across a region would be cost-prohibitive, so road density within a buffered distance around streams was used as a surrogate for estimating loadings of toxins into a stream.

I developed the ERI in tandem with a species-based aquatic gap analysis to inform managers about areas with more/less risk to species viability. GAP analysis seeks to protect biota by overlaying distributions of species and communities with maps of land stewardship to identify areas most or least likely to perpetuate those species and communities (Scott et al. 1983, Stoms 2000). With this in mind, the ERI had to be applicable to all stream biota and compare threats with a common biological currency. Thus, the ERI protocol comprises five main steps (Figure 1.2): 1) identify mappable land and water uses, termed threats to ecological integrity; 2) assign severity scores based on potential impacts of each threat to ecological integrity; 3) estimate frequencies of each threat within pre-defined subunits; 4) compute a threat-specific index of ecological risk for each subunit; and 5) compute a composite index of ecological risk over all threats for each subunit. A risk index is then computed for an array of subunits within a larger region to allow comparison of subunit-specific threats. Index values can be readily mapped and integrated with projections of occurrences of conservation targets, such as species or community types, to facilitate identification of areas most or least in need of protection.

The viability of conservation targets is affected by the frequency and severity of threats to ecosystem structure and function (Moss 2000). The ERI quantifies risk levels by accounting for the location of threats on a per- spatial subunit basis and estimating potential impacts of identified threats. Frequency scores are assigned based on total frequency counts per subunit. I assigned frequency classes at equal intervals of occurrence for lack of ecological data to inform me otherwise. Exceptions, for which empirically-derived frequency classes have been referenced frequently in the literature, included urban and agricultural land uses (Fitch and Adams 1998, Finkenbine et al. 2000, Wang et al. 2000), roads (Forman and Alexander 1998), and dams (Ligon et al. 1995). These studies gave degradation thresholds, and we assigned corresponding frequency scores to reflect no occurrence (0), minimum (1), moderate (2), or maximum (3) occurrences or thresholds to each threat. I chose three categories of frequency scores to enable us to separate lower risk areas from higher risk areas.

Potential harm is characterized by expected impacts on ecological integrity. Severity scores are based on local effects to stream conditions from a particular threat (Step 3). For example, bridges affect water quality and physical habitat more severely than they do flow regime, energy sources, or biotic interactions (Table 1.2). A matrix of ranks summarizes the impact of individual threats (i.e., as low (1), moderate (2), or high (3)) on biotic drivers. Each threat component of the matrix and severity score are ranked independently and cumulative threats are considered only in the final step.

Ecological risk index scores are coarse estimates of the risks imposed by human activities within subunits of a region. An index of threat-specific ecological risk (ERI-T) is assigned for each subunit by multiplying individual threat severity scores (step 2) by each respective frequency score (step 3). This index measures threat prevalence (Step 4), and subunits with relatively low or high impacts from individual threats can be identified. Maps can be generated to illustrate the spatial distribution of subunits with low, moderate, and high ERI-T values. This procedure facilitates a comparison of individual threats across a region, thereby providing a coarse overview of land/water uses and their possible influence on biotic conditions.

A composite index of ecological risk (ERI-C) can be computed as a summary of ERI-T values to quantify overall risk to ecological integrity across the study area. Again, maps can be produced to show the spatial distribution of subunits with low, moderate, and high ERI-C values. Index values are specified based on the respective possible values of the threat-driver matrix and frequency classes (i.e., scores of 5-15 from the threat-driver matrix multiplied by 0, 1, 2, or 3 frequency classes), and not actual threat risk rankings. These final steps provide an overview of cumulative impacts as well as an assessment of individual threats across a region. Results may then be used to prioritize conservation actions.

The ERI protocol was developed to provide a standard procedure for studying human impacts on stream biota within a larger framework for conserving and managing watersheds. Conceptually similar to multi-stressor risk assessments, it applies concepts of ecological integrity as a basis for assessment. The protocol is meant to be adaptable to the number of threats and severity of harm incurred so that it may be updated as needed. Parts of the protocol are based on expert opinion and local circumstances, which make it generally applicable. Conservation planners may use the ERI as a tool for selecting areas within large regions for conservation actions.

Applying the ERI to the Upper Tennessee River Basin

The upper Tennessee River basin (UTRB) includes the entire drainage of the Tennessee River upstream of Chattanooga, TN (55 400 km^2) (Figure 1.3). It encompasses part of the Cumberland Plateau and the mountainous regions of the Valley and Ridge and Blue Ridge physiographic provinces, in which steep slopes and narrow valleys form trellis-patterned stream networks. These and other unique physiographic characteristics, such as karst formations, have contributed to the evolution of many endemic fish, mussel, and other aquatic species (Hampson et al. 2000). The UTRB comprises mainly forest (65%) and agricultural lands (25%), with 6% of the basin urbanized (Hampson et al. 2000). Although only a small portion of the basin has been developed economically, human activities have caused a decline in ecological integrity (Neves and Angermeier 1990, Bolstad and Swank 1997, Diamond and Serveiss 2001, Diamond et al. 2002). Today, the UTRB has the greatest number of imperiled species per unit area in the continental United States. (Hampson et al. 2000).

METHODS

I identified 12 major threats within the UTRB that could be characterized as either point data or land cover categories: agriculture (row crops and pastures), urban areas, industrial areas, major dams, mining sites, bridges, manufacturing sites, solid waste facilities, railroad density, National Pollutant Discharge Elimination System (NPDES) permit sites (USEPA 2004), and road density. Many of these threats have been identified previously (Hampson et al. 2000, Upper Tennessee River Roundtable 2000, Diamond and Serveiss 2001, Diamond et al. 2002, Smith et al. 2002), and represent major pollution sources within the UTRB (Carpenter et al. 1998).

Pasturelands account for the majority of agricultural uses, with row crops occupying <3% of the entire study area (Hampson et al. 2000). Impacts on the riparian corridor due to poor pasture management may be long-lasting (Harding et al. 1998). Urbanization is an increasing and chronic threat to aquatic ecosystem integrity (Wang et al. 2001), and human populations in portions of the UTRB are expected to increase up to 30% by 2020 (NCDWQ 2002). The amount of impervious surface is not entirely dependent on population growth, but stream channel changes and sedimentation are likely to become more common hazards as additional areas are developed.

Agricultural, urban, and industrial area-based data were obtained from National Land Cover Data (USGS 1992) and summarized from 30-m^2 cells. I used surface hydrography (National Hydrography Data 1999) to identify 4^{th}-order Strahler stream reaches. I then delineated 107 subunits (watersheds) in which headwaters drained to a single outlet. Subunits associated with downstream reaches had an input from upstream and a 4^{th}-order output. Surface flow within the five major watersheds listed in Figure 1.3 corresponds to 8-digit USGS hydrologic units. Dam location information was extracted from the National Inventory of Dams (NID 2001) database maintained by the US Army Corp of Engineers. The TIGER/Line 2000 database (U.S. Census Bureau) was used to obtain spatial data for railroads, bridges, and road density. Railroad density was estimated by the length of track in each subunit. Bridge data were constructed by intersecting the data layer of primary and secondary roads with the hydrography layer. Road density was chosen based on the correlation between road length and stream proximity (10, 30, 50, and 100m) as an indicator of surface erosion.

I used the U.S. Environmental Protection Agency (US EPA) regulated site inventory data and industrial code definitions from the Occupational Safety Health Administration to obtain locations of primary or secondary sites of mining, manufacturing, and solid waste. US EPA NPDES-permitted facilities include various types of animal feeding operations, sewer and storm water overflows, and water pretreatment facilities. Effluent data from municipal and manufacturing-related sites provided complementary, unduplicated information on point-source threats to ecological integrity.

RESULTS

Threats with direct or continual influences on streams such as row crops generally exhibited higher severity scores than threats located farther from streams or with intermittent effects (Table 1.2). This pattern reflects land uses lacking adequate riparian buffers as well as threats occurring within stream channels, respectively including manufacturing sites located next to a stream and dams (Ligon et al. 1995). The resultant ecological changes, such as water temperature changes, increased sediment, habitat alteration, and vegetation changes, have both local and regional impacts on ecosystem functions (Hughes and Hunsaker 2002). All threats were weighted equally, and any differences in upstream versus downstream impacts were not considered in my analysis.

Maps of ERI-T scores reflect drainage subunit-specific risk patterns for individual threats (Figure 1.4). Pasturelands, row crops, and urbanized areas incurred higher risk in subunits characterized by valleys and lower elevations. Even though pastureland has a lower severity score than row crops, its higher frequency elevated its risk rankings. Point sources, such as manufacturing, waste disposal, and NPDES permit sites, suggest that industry-related land uses are much more prevalent than their areal extent may indicate. This outcome may be due to inherent differences in point data versus area measures. No single threat at the subunit level dominated the UTRB as a high risk to ecological integrity; instead, each watershed had its own predominant threats (Figure 1.4).

ERI-T values were summed over all threats in a subunit to obtain a composite index of ecological risk (ERI-C) to aquatic system health (Figure 1.5). ERI-C scores suggest that few subunits have especially high composite risk levels; however, there are substantial impacts throughout the UTRB. High risk areas may be characterized by high risk frequencies or by low frequencies of severe threat. The spatial pattern of subunits with high ERI-C values suggests that threats with moderate severity but high frequency contribute more cumulative risk than do very severe but infrequent threats (Figure 1.6). Dams, pastures, and manufacturing-related threats within the highest ranked subunits appeared to pose the greatest risk to ecological integrity. No single threat in the composite index stands out as the main source of risk over the entire UTRB.

A principal components analysis of the ERI-C scores indicated that subunits varied considerably in threat composition. Watersheds with greater frequencies of intensive land use had higher ERI-C scores, and threats with high severity scores (i.e., magnitude) affected risk rankings independent of their frequency. The first two principal components accounted for 54% of the variance in threat composition among subunits (Table 1.4). The first component primarily represented variation in ecological risk from point sources with direct influence on stream quality, namely manufacturing sites, waste disposal facilities, NPDES sites, and mines (Table 1.4). Impacts from these threats are generally localized and of moderate severity. The second principal component primarily represented variation in risk from major dams, industrial areas, row crops, and urbanized areas (Table 1.4). Impacts from the latter three threats are spatially extensive and severe. The risk attributable to major dams was inversely related to the risk attributable to the other three threats (Table 1.4), suggesting that dams impact UTRB streams independently of other threats.

Differences in risk patterns among drainages reflect predominant land uses (Table 1.5). For example, the majority of mining sites are found in the Clinch-Powell and Holston drainages, whereas the Little Tennessee drainage has a high frequency of all threats. Dam sites had the lowest frequency among all of the threats within the Clinch-Powell drainage, and were also given a high severity score in this drainage. The importance of waste facilities, bridges, pastures, row crops, and manufacturing sites varied significantly among the drainages (Table 1.5). These results are consistent with those of other studies that have found different causes of impairment as the spatial extent of analysis is varied (Moss 2000, Rabeni 2000).

DISCUSSION

I found that readily available data were adequate for providing an overview of current threats within the UTRB. Severity was scored for each threat independently, and synergistic or cumulative effects from multiple threats were not considered in severity scores. Although additional data pertaining to global threats and external influences, such as air pollution controls or precipitation patterns, may provide a more accurate assessment of local impacts, including such variables was not within the scope of my study. I also did not address land use changes, as the ERI is not temporally or spatially explicit at this time. The ERI was constructed so an alternative suite of threats could be used and/or severity scores could be updated as more knowledge becomes available.

My case study demonstrates that the ERI is a potentially useful tool for evaluating risk in local and regional conservation planning. Risk indices combined the potential impact of human activities on system drivers with frequency of occurrence measures to summarize potential harm to system resources. Conservation targets tend to shift away from specific species to retention and maintenance of ecosystem properties in the indices, which may make it difficult to determine which threats are driving cumulative degradation. Furthermore, large national databases seem adequate for use in prioritizing conservation areas when anthropogenic effects are explicitly addressed. I also found that the relative importance of threats is sensitive to the spatial grain of analysis.

Future applications of the ERI will investigate the inclusion of a spatially explicit component to address issues of mitigating effects of landscape features, distance of threats from streams, and cumulative impacts downstream. I expect risk rankings to change as elevation and spatially explicit components, such as threat dispersal, are added. The ERI has not been tested for its predictive capabilities, and a biological response indicator coupled with data on land use change would also provide valuable information.

Informing Conservation Planning

The ERI is an assessment tool for evaluating the frequency and severity of threats to ecological integrity, and can inform conservation planning in several ways. It is meant to be a coarse filter for identifying patterns of regional land uses and impacts, and may be used in conjunction with higher resolution data for local planning. Due to its regional scope, the ERI also provides more information on the types and degree of risk than other conservation frameworks, and directs regional planning of conservation needs. For example, Zhang and Chen (2014) used the ERI to quantify human impacts across a large spatial extent using readily available land use data, and found the approach useful for prioritizing management plans. The Nature Conservancy has used the ERI as a framework for prioritizing conservation efforts in watersheds throughout South America (unpublished source).

Other freshwater-based classification approaches implicitly include risk in their respective viability assessments (e.g., Abell et al. 2000, Groves et al. 2002). The ERI complements these classification frameworks by addressing threat- frequency and severity explicitly so that risk at various spatial extents may be integrated and compared. Paukert et al. (2010) used a similar approach when they made an ecological index based partly on the ERI. The approach is appealing because it provides an objective overview of the degrees of impact related to human activities with data that are easily obtained, and the index may easily be included in other conservation planning frameworks (e.g., Higgins et al. 2005).

Improving the conservation planning process does not require a reinvention of techniques and concepts. Risk-based assessments provide an adequate basis for characterizing the impacts of human activities on conservation targets. Expert opinion may also be used to rank regional-level impacts to ecological integrity; analogous techniques have been shown to be useful for large-scale studies (Wiegers et al. 1998, Bryce et al. 1999, Walker et al. 2001, Barve et al. 2005). Linking anthropogenic stressors with ecosystem drivers may prove useful in identifying areas that should be considered for conservation actions.

Given that all applications and techniques have limitations, borrowing a framework and tools from an established field is advantageous to developing a new approach (Stem et al. 2005, Paukert et al. 2010). Explicitly addressing the risks to biotic drivers to inform conservation planners of threats to conservation targets affords a cost-effective and holistic view of the impacts of human activities on both terrestrial and aquatic systems.

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Table 1.1. Comparison of terms used in the continuum of risk-based analyses.

	Conventional Risk Assessment	Ecological Risk Assessment	Threat Assessment
Threat	not applicable	Potential or actual source of stress	anthropogenic source of stress
Hazard/Stressor	The source of an adverse effect (e.g., industrial plant)	act or entity that has the potential to do harm; proximal and distal stressors may be identified	anthropogenic source of stress
Risk	probability of occurrence due to exposure to hazard	probability of occurrence of harm due to exposure to hazard	likelihood of a negative effect on system components
Harm	quantitative measure of the hazard to human health (e.g., tumor growth)	quantitative measure of change in an ecological system (e.g., fish kill)	qualitative or quantitative measure of the negative effect of a threat to ecosystem integrity (e.g., change in water quality)
Risk/Exposure factor	coefficients relating endpoint assessment to amount of harm incurred (e.g., human population within 10 miles of industrial facility)	uses natural system coefficients (e.g., fish species within 10 miles downstream of toxic release)	none measured; changes in the natural range of variability could be considered
Impact	quantitative measure of the amount of harm to an endpoint	quantitative measure of the amount of harm of a threat/hazard to an endpoint	measure of existing or potential harm of a threat to a study area
(Exposure) Pathway	route that substance (hazard) takes through system; usually human health-related (e.g., endocrine-affecting substances travel through water, are ingested, attack liver, pancreas)	route from a threat/hazard to an endpoint; usually ecosystem derived (e.g., stream route of toxin output)	usually not determined due to system complexity
Assessment Endpoint	object being assessed; usually human health (e.g., increase in cancer rate)	ecosystem structure or function, vertebrate species health or population viability	describes the system state to be attained (i.e., ecological integrity)
Test/Measured Endpoint	quantitative measure of the response to a hazard; usually human health-related (e.g., occurrence of damage in liver, pancreas)	quantitative measure of the response to a threat/hazard; usually species-related (e.g., number and species of fish killed)	measures system components related to ecosystem condition

Table 1.2. Matrix of severity ranks (low=1, moderate=2, and high=3) for major threats within the Upper Tennessee River Basin (UTRB).

Threats	Impact	Water Quality	Habitat Quality	Biotic Interactions	Flow Regime	Energy Sources	Severity Score
Row crops	Low						
	Medium			X			14
	High	X	X		X	X	
Pasturelands	Low			X			
	Medium	X			X		11
	High		X			X	
Urbanized areas	Low						
	Medium			X			14
	High	X	X		X	X	
Industrialized areas	Low					X	
	Medium				X		12
	High	X	X	X			
Mining sites (old and current)	Low					X	
	Medium			X			12
	High	X	X		X		
Waste facilities	Low						
	Medium		X	X	X		12
	High	X				X	
Bridges	Low						
	Medium			X	X	X	12
	High	X	X				
Major dams	Low						
	Medium						15
	High	X	X	X	X	X	
Manufacturing Sites	Low					X	
	Medium			X	X		11
	High	X	X				
NPDES permit sites	Low			X			
	Medium				X		12
	High	X	X			X	
Road density (30m buffer)	Low			X	X	X	
	Medium						9
	High	X	X				
Railroad density	Low			X	X	X	
	Medium	X	X				7
	High						

Table 1.3. Frequency scores (0- not present, 1-minimum, 2-moderate, and 3-maximum impact) used to compute risk indices for 12 major threats within the Upper Tennessee River Basin (UTRB). Integer frequencies are the actual number of occurrences in a given subunit. Land cover represents the percent of area in a given subunit. Equal interval classes were used when no related risk-based studies were found.

	Frequency Rank Scores				
Threat	0	1	2	3	Classification Method or literature used in rankings
Row crops (%)	<2%	2-9%	10-49%	>50%	Wang et al. (2000)
Pasture (%)	<2%	2-9%	10-49%	>50%	Wang et al. (2000)
Urbanized areas (%)	<2%	2-9%	10-49%	>50%	Finkenbine and Mavinic (2000), Wang et al. (2001)
Industrialized areas (%)	<2%	2-9%	10-49%	>50%	Finkenbine and Mavinic (2000), Wang et al. (2001)
Mining sites	0	1	2	>2	equal interval
Waste facilities	0	1	2-3	>3	equal interval
Bridges	0	1-16	17-54	>54	equal interval
Major dams	0	1	2	≥ 2	expert opinion
Manufacturing sites	0	<3	3-10	>10	equal interval
NPDES permit sites	0	1-2	3-7	>7	equal interval
Road density (km/km^2)	0	<0.1068	0.1069-0.1622	>0.1622	Forman and Alexander 1998
Railroad density (km/km^2)	0	<251	251-1420	>1420	equal interval

Table 1.4. Loadings of major threats on the first two principal components (PCI and PCII) of 107 subunits within the UTRB. Variance in threat composition among subunits explained by PCI and PCII is also shown.

Threat	PCI	PCII
Row crops	0.151	0.465
Pasturelands	0.246	0.263
Urbanized areas	0.301	0.345
Industrialized areas	0.233	0.435
Mining sites	0.337	0.001
Waste facilities	0.385	-0.085
Bridges	0.295	-0.344
Major dams	0.209	-0.447
Manufacturing sites	0.402	-0.141
NPDES permit sites	0.384	-0.203
Road density	0.008	0.113
Railroad density	0.264	0.086
Variance (%)	40	15

Table 1.5. Mean frequencies and variances of 12 threats in watersheds of major drainages of the UTRB. The number of subunits (N) in each drainage is also shown.

Threats	Clinch-Powell (N=15)	Holston (N=13)	French Broad (N=47)	Hiwassee (N=6)	Little Tennessee (N=26)
Row crops (%)	1.3 (0.72)	2.8 (1.89)	2.8 (7.21)	0.6 (0.73)	2.1 (6.08)
Pasturelands (%)	12.4 (58.83)	22.5 (60.40)	13.0 (135.92)	2.4 (6.57)	9.8 (95.27)
Urbanized areas (%)	2.4 (5.12)	6.0 (43.87)	3.4 (16.51)	0.7 (1.1)	2.9 (35.61)
Industrialized areas (%)	0.7 (0.44)	1.7 (1.84)	1.0 (1.02)	0.18 (0.05)	0.8 (2.23)
Mining sites	4.0 (13.98)	2.0 (5.67)	1.0 (6.23)	1.0 (1.77)	2.0 (18.88)
Waste facilities	7.0 (37.54)	7.0 (60.74)	2.0 (7.17)	0.0 (0.17)	8.0 (483.13)
Bridges	56.0 (1652.83)	65.0 (2295.91)	39.9 (866.30)	18.5 (288.70)	28.6 (792.49)
Major dams	2 (5.55)	1 (1.58)	1 (1.04)	1 (2.57)	1 (3.15)
Manufacturing sites	14.0 (167.35)	28.0 (803.10)	7.0 (92.48)	1.5 (4.30)	18.4 (2203.28)
NPDES permit sites	9.0 (55.60)	10.0 (48.97)	5.9 (32.91)	2.7 (10.67)	5.1 (36.47)
Road density (km/km^2)	0.1 (0.0)	1.9 (42.11)	0.19 (0.02)	0.2 (0.02)	0.7 (7.93)
Railroad density (km/km^2)	191.9 (20899.72)	249.8 (158877.11)	99.3 (16094.68)	34.3 (3721.24)	140.3 (48912.17)

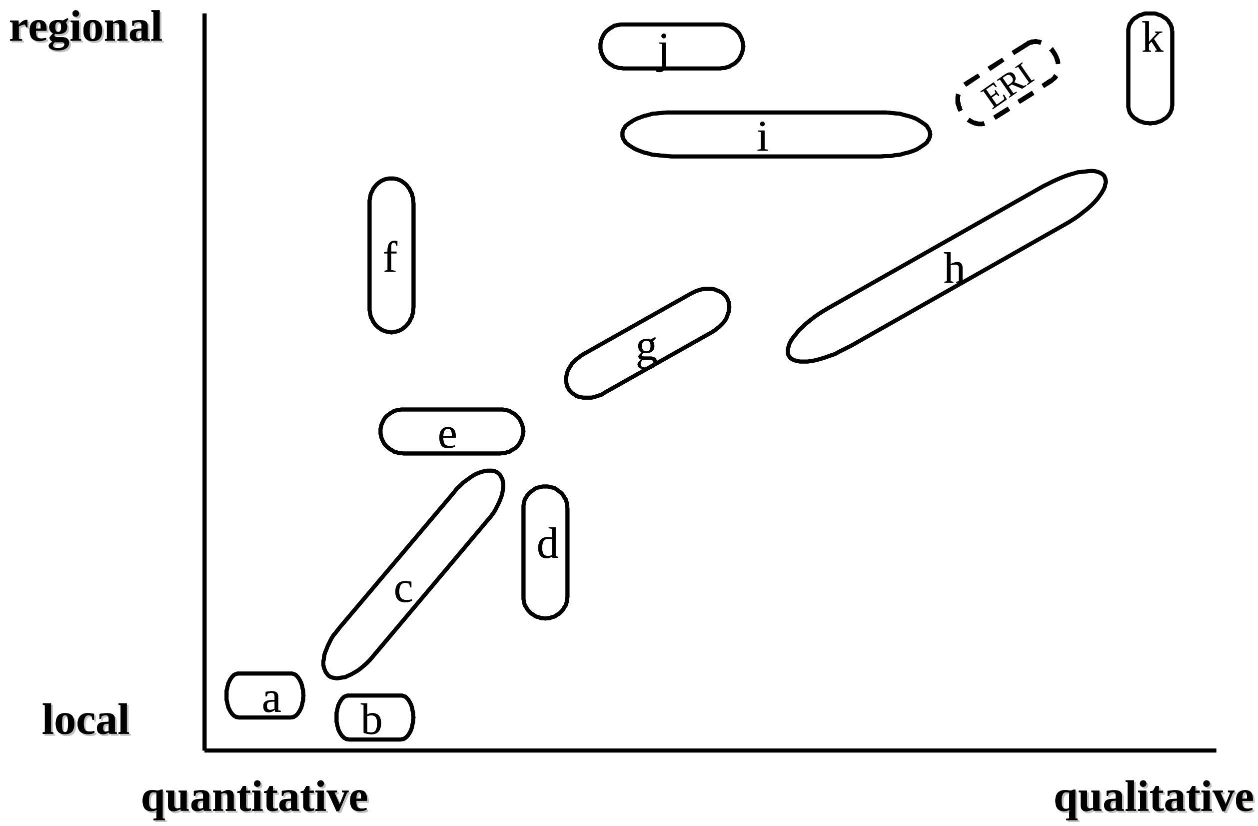


Figure 1.1. Schematic representation of ecological risk-based assessments along gradients of spatial scale and methodology. Horizontal position reflects the relative importance of quantitative analysis. Vertical position reflects spatial extent. Each ellipse represents one or more published assessment approaches. (a) Moore 1998, Carbonell et al. 2000; (b) Richter et al. (1996), Suter et al. (2002); (c) Hughes et al. (2000), Wang et al. (2001); (d) Preston and Shackleford (2002); (e) Rogers et al. (2002); (f) Richards et al. (1996), Rabeni (2000), Diamond et al. (2002); (g) Mebane (2001), Cormier et al. (2002), Norton et al. (2002); (h) Slob (1998), Osowski et al. (2001); (i) Graham et al. (1991), Wiegers et al. (1998), Walker et al. (2001); (j) Turner et al. (1996), Detenbeck et al. (2000); (k) Russell et al. (1997), Bryce et al. (1999), Muhar et al. (2000).

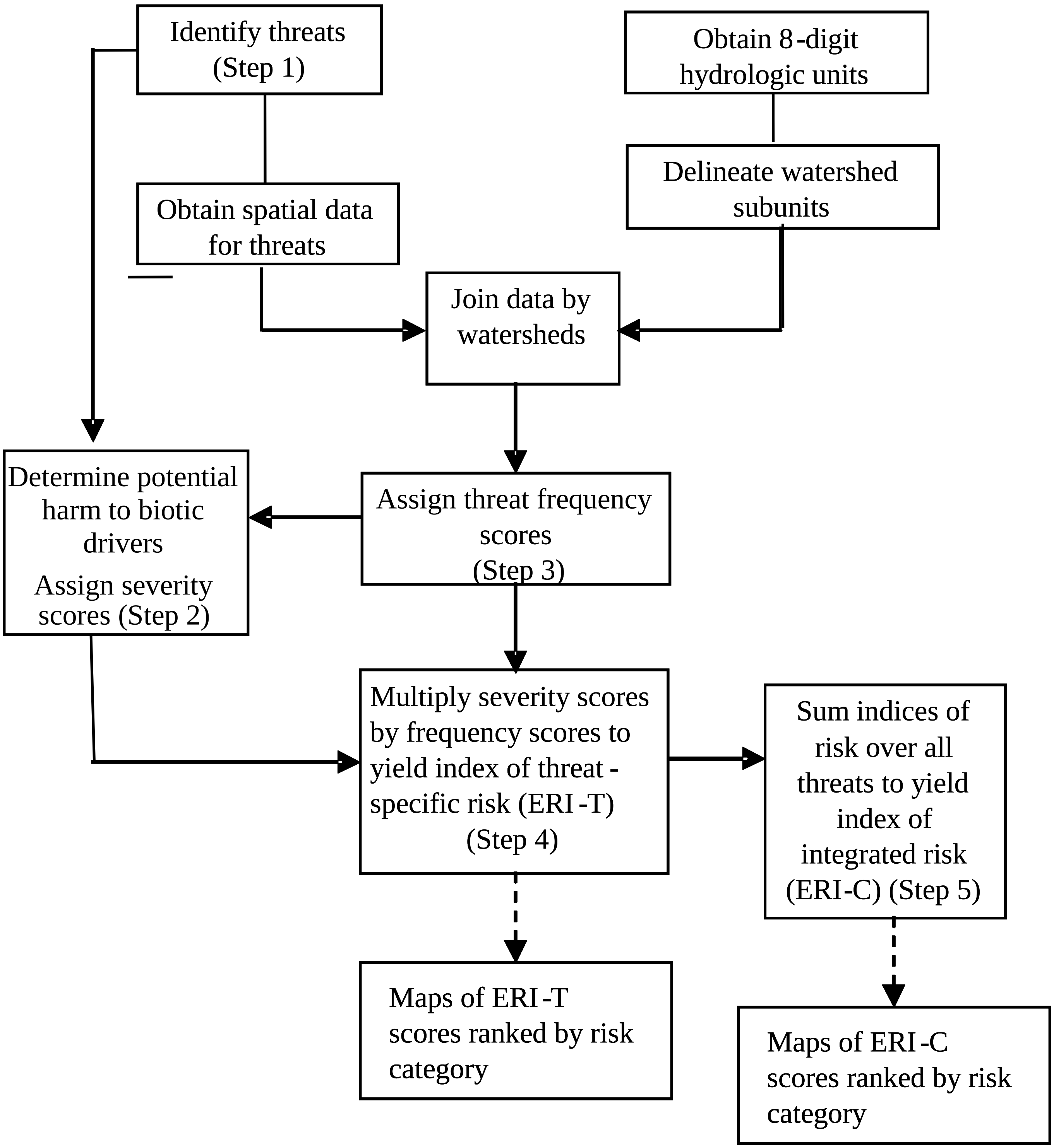


Figure 1.2. Flowchart depicting major steps in developing the Ecological Risk Index within a threat assessment framework. All major threats within each subunit are summarized and maps are produced with final ranking scores to allow visual comparisons.

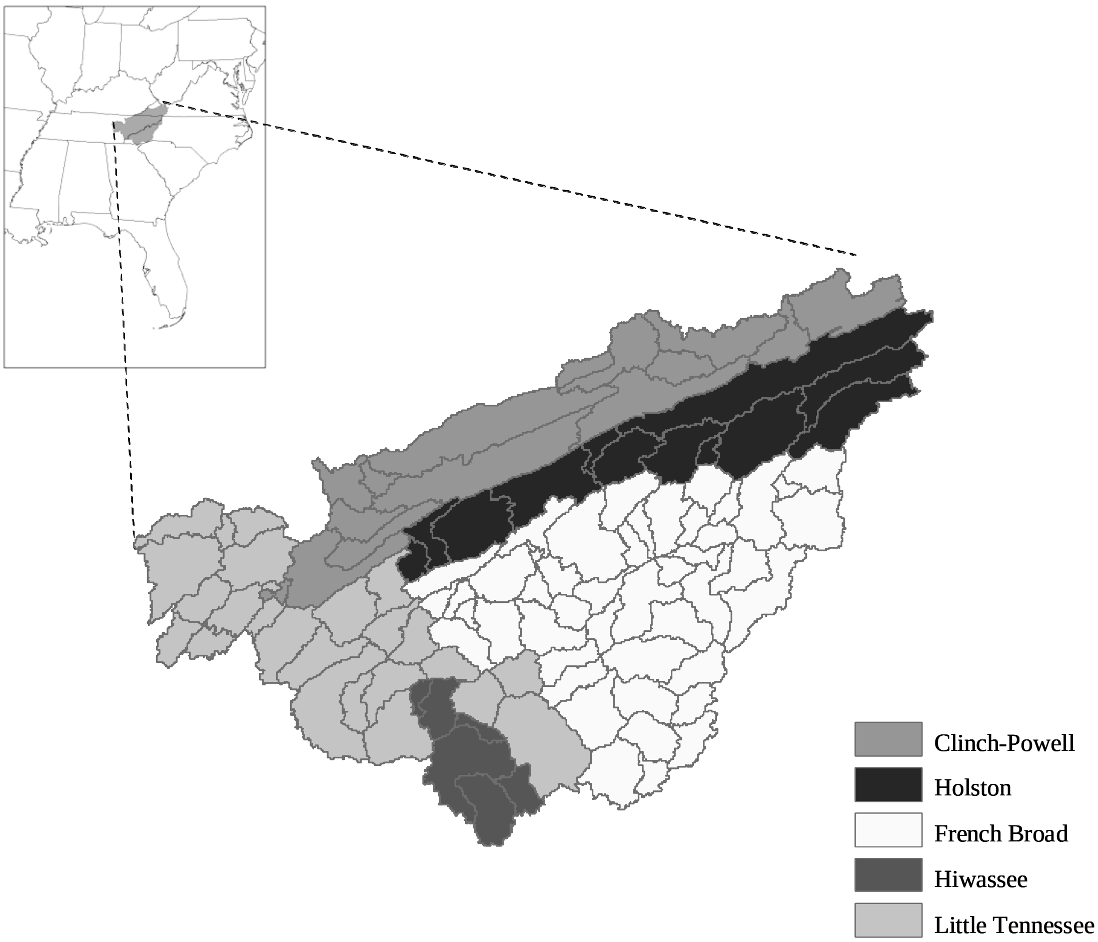
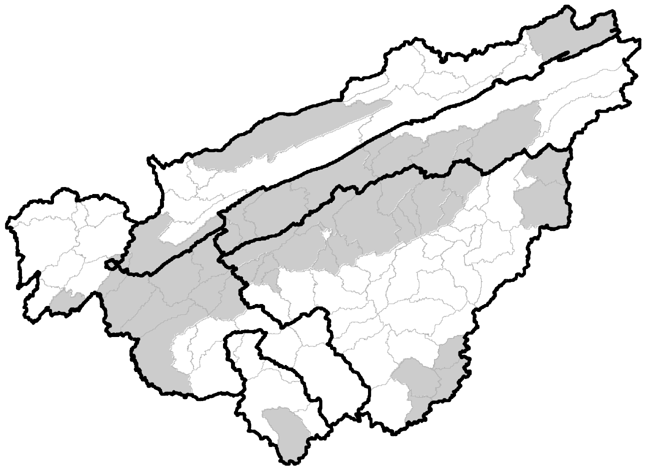
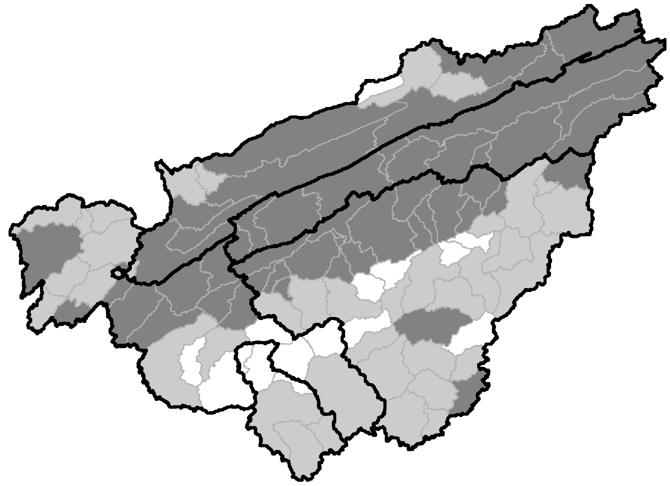


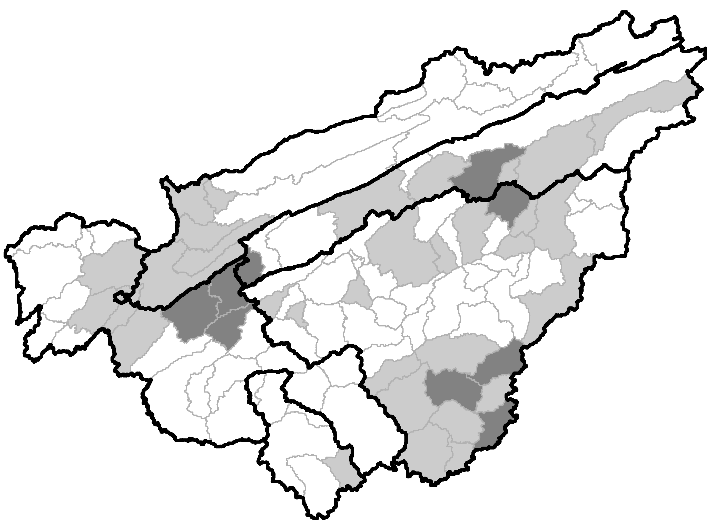
Figure 1.3. The upper Tennessee River basin, USA. Delineation is of 107 subunits based on 4th-order streams. Major river drainages are depicted (shaded regions).



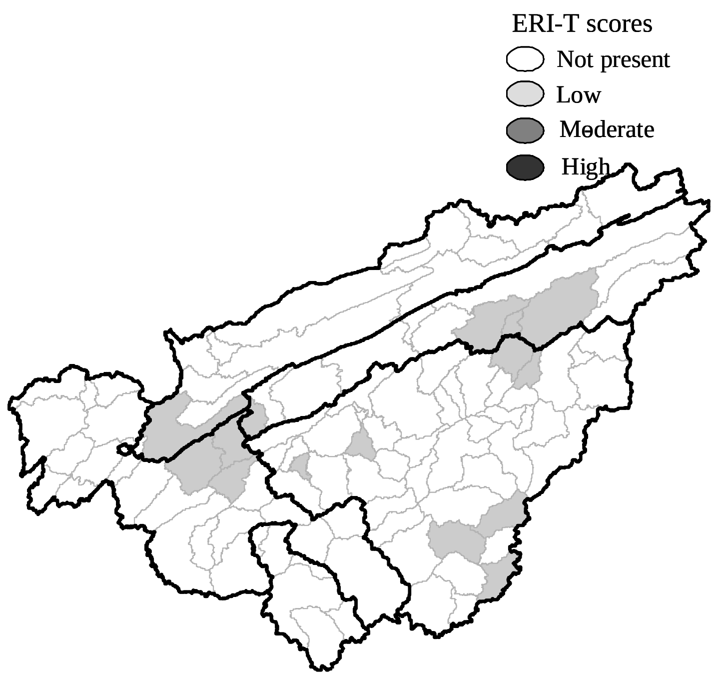
Row crops



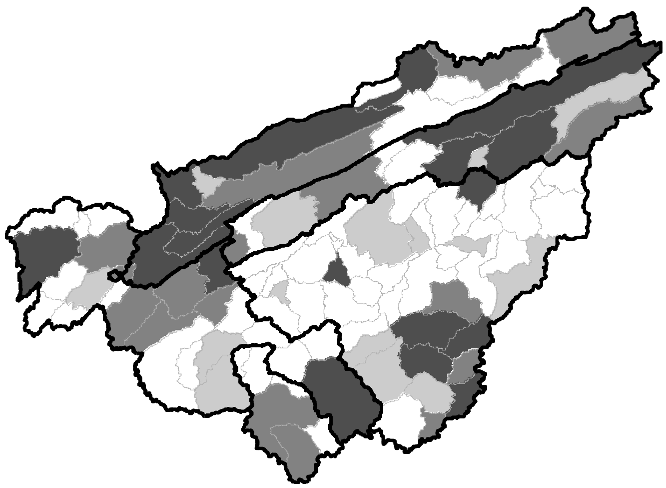
Pasturelands



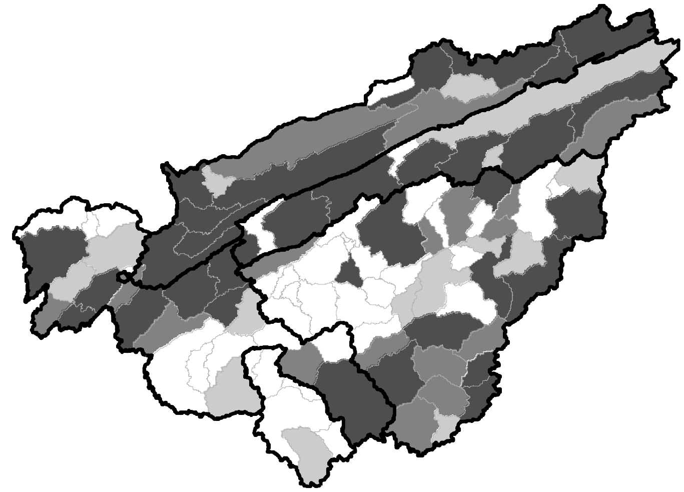
Urbanized areas



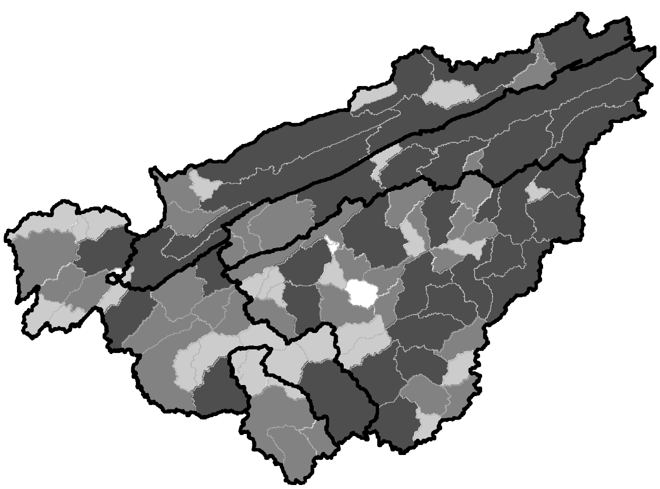
Industrial areas



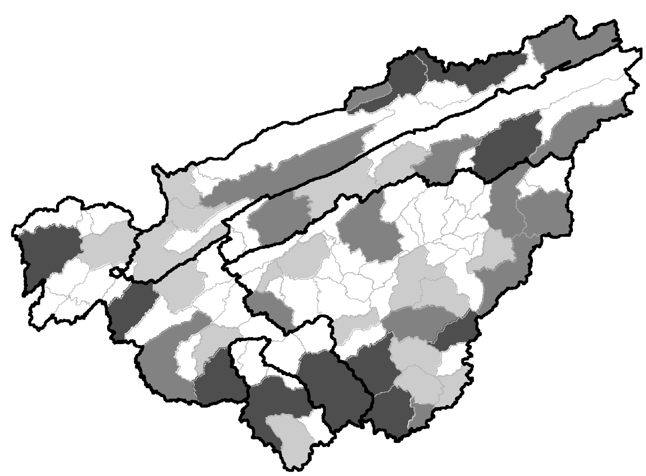
Mining sites



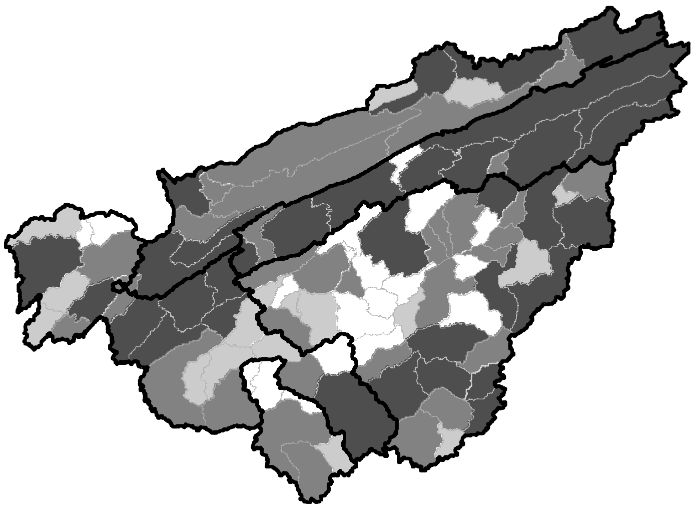
Waste facilities



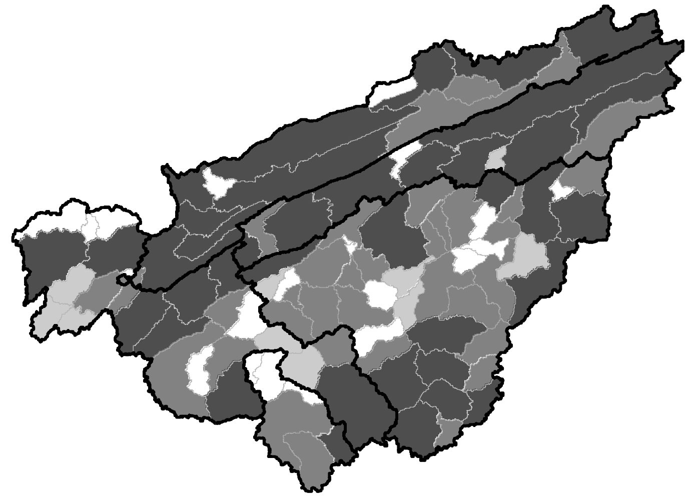
Bridges



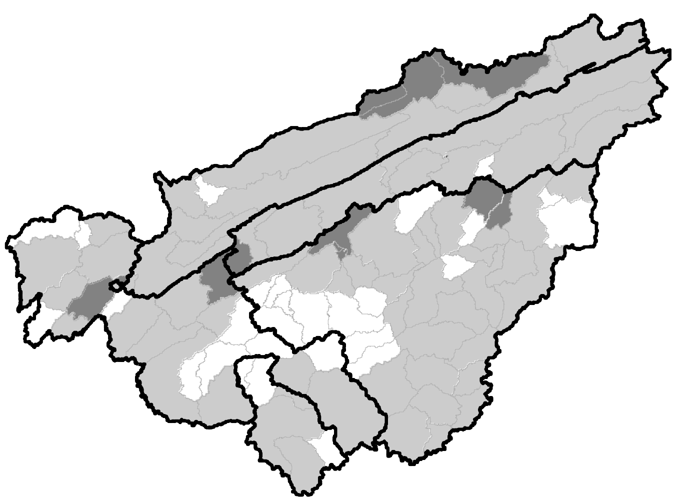
Major dams



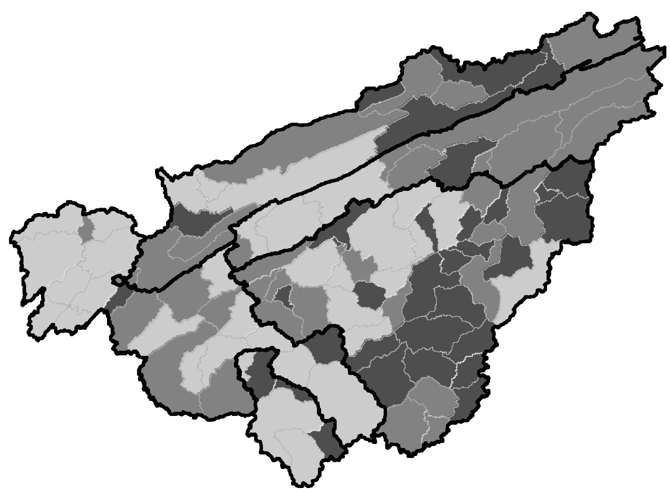
Manufacturing sites



NPDES permit sites



Railroad density



Road density

Figure 1.4. Maps of threat-specific indices of ecological risk (ERI-T) for 12 anthropogenic threats in 107 subunits within the UTRB. Low, moderate, or high frequency scores were assigned by subunit.

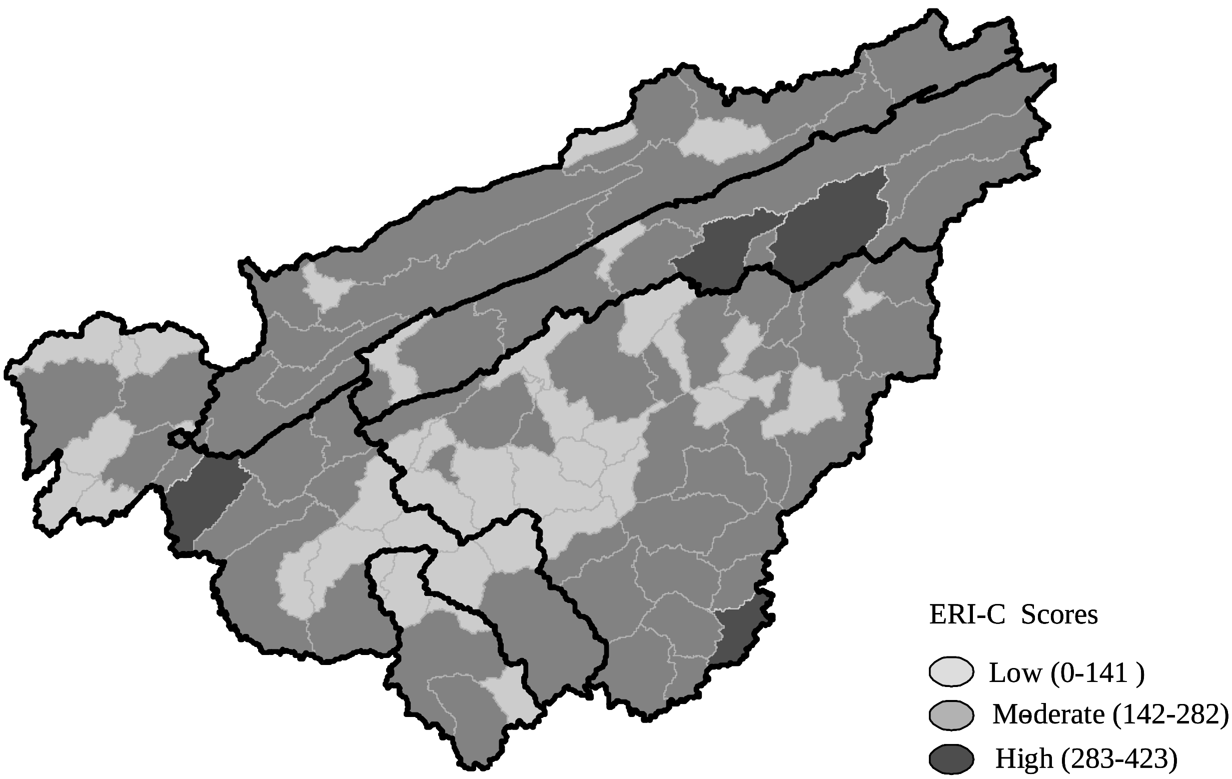


Figure 1.5. Maps of subunits within the UTRB with low, moderate, and high values of a composite index of ecological risk (ERI-C) to aquatic ecosystem integrity.

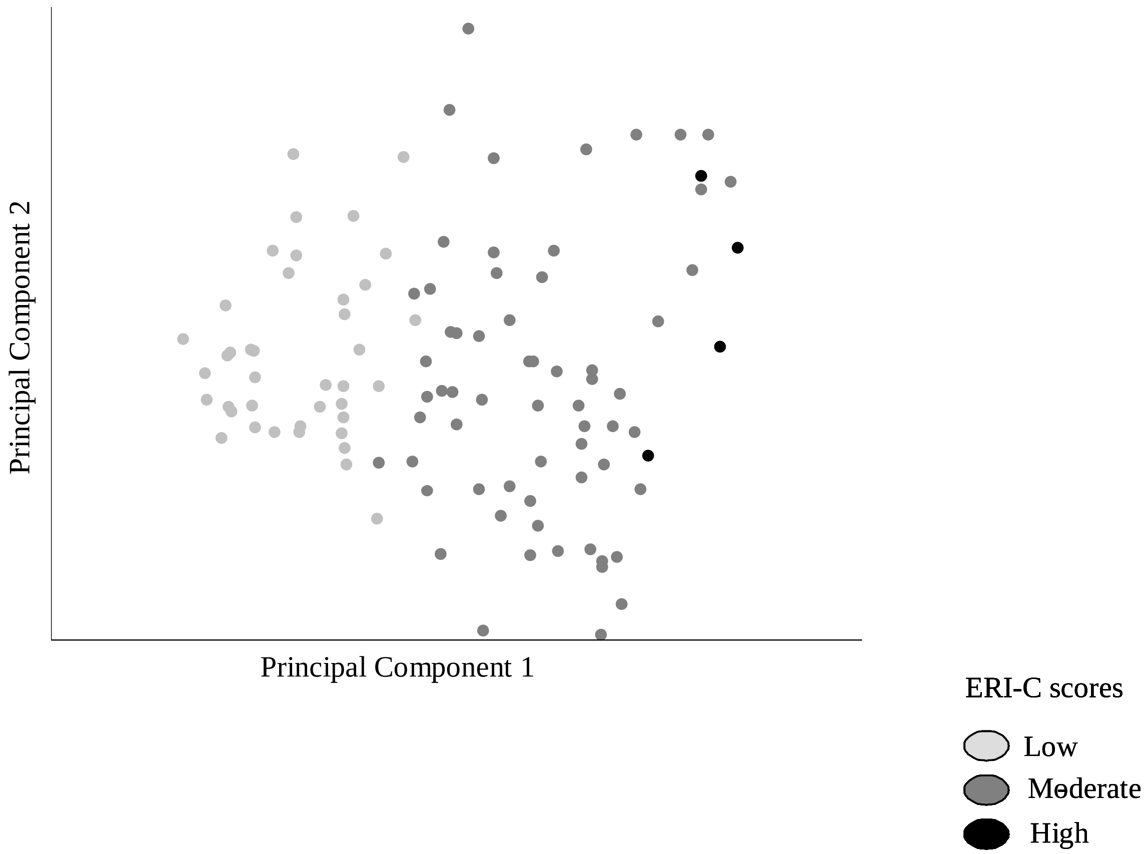


Figure 1.6. Ordination of 107 subunits in principal components space defined by indices of threat-specific ecological risk. Subunits are labeled by their composite index of ecological risk (ERI-C). Some points overlap.

CHAPTER 2: Robustness of rank-based risk assessments of freshwater ecosystems in conservation planning

ABSTRACT

I developed the Ecological Risk Index (ERI; Mattson and Angermeier 2007) to provide a coarse-scale approach to quantifying threats to aquatic systems. Measures of frequency and severity of harm are used to rank the risk of potential impacts to stream quality. This approach to risk assessment, termed relative risk modeling, is commonly applied at large spatial extents, but has been criticized for its use of expert judgment and model parameters that are rarely assessed before management decisions are made. I evaluated the parameters of the ERI in order to address these issues. I assessed the robustness of the qualitative parameters of the ERI, namely the judgment-based methods used to assign threat-related parameters. My objectives were to: 1) assess the ranking procedures within the index, and 2) characterize the robustness of model outputs to overall risk classification. I suggest that freshwater conservation planners evaluate risk-based models for their purpose and robustness prior to using them in planning decisions. Despite some limitations, risk-based ranking methods are useful in conservation planning.

INTRODUCTION

Conservation planners commonly evaluate ecosystem vulnerability to anthropogenic threats using rank-based risk assessments because they afford a quick, coarse-scale approach to identifying those target areas that are most susceptible to degradation (Groves et al. 2002, Higgins et al. 2005). Rank-based methods bridge the gap between rigorous quantitative ecological risk assessment and the practical need to quickly evaluate multiple threats at increasingly larger spatial extents. Although derived from classical ecological risk assessment, rank-based approaches have been criticized because of their qualitative nature and for the variety of methods used in the ranking process (Suter et al. 2002, Wolman 2006). Despite these criticisms, rank-based risk assessments have proven useful for characterizing ecological degradation in terrestrial, marine, and freshwater settings, as well as for prioritizing management actions in the presence of multiple threats (e.g., Bryce et al. 1999, Higgins et al. 2005, Halpern et al. 2007). Even though rank-based risk assessments are frequently used in terrestrial-, marine-, and freshwater-based conservation planning, their legitimacy as management tools would be improved if the reliability of their qualitative components were evaluated as part of the assessment process (Suter et al. 2002). Much of the uncertainty associated with ranking procedures involves two qualitative factors: threat categorization and the ability to appropriately assign overall risk rankings (Smith et al. 2006).

Incorporating quantitative data into risk-based models is an especially challenging task in freshwater conservation planning because of the complex relationship between impacts of land/water use and the resulting stream condition (Dudgeon et al. 2006). Risk assessment approaches for freshwater systems might include summarizing relative degrees of impact from human activities within riparian areas as well as identifying catchments with the greatest need of protection or restoration (Jackson et al. 2004, Van Sickle et al. 2004). Rank-based risk assessments are commonly used in conservation planning to characterize ecosystem threats and to determine their impacts on those targets (Groves et al. 2002). A major advantage to using ranking procedures in freshwater conservation planning is their flexibility for incorporating various qualitative and quantitative data that reflect real-time anthropogenic threats, which affords a reliable means of responding to urgent management concerns. For example, data related to urban land use and extent of impervious surfaces are readily available and adequately depict areas with compromised stream integrity, thereby providing a cost-effective surrogate for expensive field-based stream sampling (Rabeni and Sowa 2002).

A commonly used approach is to combine quantitative, risk-based measures with less quantifiable inputs, such as expert judgment, into a single description of threat pattern (e.g., Wiegers et al. 1998, Halpern et al. 2007, Mattson and Angermeier 2007). This approach enables evaluation of large study areas using data collected for other purposes, and provides an instructive summary of the threats consequent to anthropogenic activities. Ranking procedures may also be applied at smaller spatial scales to identify specific disturbance patterns, or to predict vulnerability to biodiversity losses (Wilson et al. 2005), and identify biodiversity threats when quantifying the scope of related impacts (Lowell et al. 2000).

Currently, there are few standard models or formal evaluation procedures for rank-based approaches comparable to those used in conventional ecological risk assessments. Evaluations may be uncommon because of the difficulty of spending additional resources on evaluation procedures and because conservation plans are often put into practice prior to evaluation (Bottrill and Pressey 2012). I suggest that the utility of rank-based risk assessment methods may be improved by characterizing the influence of such subjective components as expert judgment on model robustness and by evaluating the effect of various parameter estimation techniques on risk rankings. Similar to ecological risk assessment, one of the purposes of rank-based risk procedures is to quantify the extent and scope of threats to conservation targets (Van Sickle et al. 2006). Threats are one component of risk-based planning such that risk is based on the magnitude of threats and the subsequent vulnerability of assets. In conservation planning, an asset is a conservation target(s), whether it is an ecosystem, species of concern, or a broader entity such as biodiversity. Threats are defined as any factor that has the potential to negatively impact conservation targets. Most identified threats are human-caused, may have chronic or acute impacts, and are quantified by their extent and scope of impact. The extent is the frequency, or numbers of threats within a study area, and the scope of an impact is the amount of negative influence, or the degree of severity of a threat on conservation targets. Lastly, vulnerability may be defined as inherent properties that make a conservation target susceptible to being impacted by threats. Degrees of severity and vulnerability are difficult to measure, making assessment of these factors more reliant on expert judgment (Halpern et al. 2007, Richter et al. 1997). The overall risk to a conservation target is defined by the combined magnitude of threats and vulnerability.

Ecological Risk Index

Assessing threats within ecoregions requires a holistic approach that considers compromises in ecological integrity, especially considering how disturbances affect the ability of an ecosystem to retain its structural and functional capacity (Angermeier and Schlosser 1995). Impacts to ecological integrity alter the major biotic drivers that influence the stability and resilience of stream systems. These drivers include stream components related to the flow regime, water chemistry, energy transfer among biotic and abiotic components, the physical habitat available to aquatic biota, and those biotic interactions found within streams and rivers (Karr et al. 1986). The degrees to which these drivers are altered by anthropogenic disturbances influence the functional capacity of a stream and ultimately indicate the ability of a catchment to retain its ecological integrity. With this in mind, I developed an expert-derived, rank-based risk procedure to assign impact thresholds and overall risk classifications, similar to that of Halpern et al. (2007). I constructed the Ecological Risk Index (ERI) in response to a need for a tool to estimate the impacts of human land and water uses on stream components across large catchments (Mattson and Angermeier 2007). The ERI was developed to combine easily obtainable geo-referenced data, expert judgment, and biological knowledge from the published literature to estimate the ecological risk of anthropogenic factors on the abiotic and biotic components of streams and rivers (Mattson and Angermeier 2007); this approach to summarizing threats and creating risk indices has been applied useful worldwide (Fore et al. 2014, Kracker 2006, Paukert et al. 2011, Zhang and Chen 2014).

The ERI relies on expert judgment to rank the potential severity of an individual threat on biological integrity (i.e., conservation target). Computing the ERI (Figure 2.1) is a five-step process that ranks land/water uses (i.e., threats) by their expected impacts on the major biotic drivers that influence the stability and resilience of stream systems. The criteria for assigning threat-frequency classes include quantitative and qualitative elements which ultimately drive the final risk rankings. It was important to ascertain the robustness of the ranking methods used to quantify risk within the ERI. This has more than heuristic value. Validation provides conservation planners with the information needed to support the use of qualitative methods in risk-based ranking approaches, as well as the opportunity to understand the limitations associated with this modeling technique. The inclusion of expert judgment should provide a unique and valuable contribution to the risk rankings, especially since the ERI is meant to summarize threats to specific conservation targets. Otherwise, quantitative methods would be preferred. I assessed the robustness of the qualitative parameters of the ERI, namely the judgment-based methods used to assign threat-related parameters. My specific objectives herein are to: 1) assess the ranking procedures within the index, and 2) to characterize the robustness of model outputs to overall risk classification.

STUDY AREA

The upper Tennessee River basin (UTRB) is located within the Appalachian Mountains in the southeastern United States (Figure 2.2). The study area encompasses the Clinch-Powell (11430 km^2), Holston (9780 km^2), French Broad (13271 km^2), Little Tennessee (6804 km^2), and Hiwassee (6993 km^2) river drainages (Figure 2). The five major drainage basins within the UTRB were divided into 107 hydrologic units corresponding to 4^{th}-order Strahler stream levels. As noted below, anthropogenic activities were summarized within each catchment. It must be noted that catchments were evaluated individually, and any aggregated degradation from upstream catchments was not considered in this analysis.

Most of the basin is forested (65%) in multi-use areas on National Forest lands. Agriculture, primarily riparian pasturelands, comprise about 25% of the land use (Hampson et al. 2000). The remaining land cover includes urbanized areas (6%), barren lands (mainly inactive and active mining facilities), rivers, and reservoirs (USGS 2001). Anthropogenic impacts have been significant for decades, and are associated with continual declines in the region's biotic condition, including extensive fish and mussel imperilment (Neves and Angermeier 1990, Diamond and Serveiss 2001, Krstolic et al. 2014, Price et al. 2014, Zipper et al. 2014).

The UTRB is one of the most diverse freshwater ecosystems in the United States (Abell et al. 2000). Historically, the basin contained over 150 native fish species, including at least 15 species that are federally threatened or endangered, and another 50 species that have state conservation status (Hampson et al. 2000). Freshwater mussel species were also historically diverse and numerous within the basin. Today, there are 60 extant mussel species; most of those are listed by the respective states as species of conservation concern, along with 30 species with federal protection (Hampson et al. 2000, VDGIF 2010).

METHODS

Ecological Risk Index

I collected geo-referenced data to quantify the extent and severity of human impacts on freshwater biota. Land use, land cover, and hydrologic data were obtained from federal and state agencies and compiled into an ArcGIS (ESRI Inc., Redlands, CA, 1999-2009) platform. Twelve major land and water uses were identified as threats to stream integrity, including row crops and orchards, pasturelands, urbanized areas, industrial areas, mining sites (active and inactive of all types), waste facilities, stream crossings, impoundments, manufacturing sites, National Pollutant Discharge Elimination sites (NPDES), railroad crossings, and roads (Figure 2.1).

The ERI quantifies risk by accounting for the frequency of catchment-specific threats and estimating their degrees of impact on the five major biotic drivers – flow regime, water chemistry, energy transfer, habitat availability, and biotic interactions. For a complete description of the ERI protocol, see Mattson and Angermeier (2007); the following is a brief summary of the ERI as it pertains to the current analysis. The ERI assigns a composite risk ranking (ERI-C) to catchments across an ecoregion by combining the frequency and severity of individual threats into a single index, and ranking catchments as facing low, moderate, or high ecological risk. The initial steps involve quantifying the number of anthropogenic disturbances (i.e., threats) within each catchment, determining the potential impact(s) (i.e., severity) associated with those threats, and summarizing each into an ERI-T score. Computations of both the frequency and severity scores incorporate the expert judgment of a panel of three individuals, who were chosen based on their extensive knowledge of the aquatic communities within the UTRB. An ERI-C score is the sum of individual threat-specific index ERI-T scores.

Accordingly, individual threats per catchment were quantified and grouped into four frequency categories that generally fit into a no, low, moderate, and high threat classification system. Point data types were summed, and area-based impacts were accumulated into density measures. A zero-frequency classification indicated that a point-source threat was not present within a catchment. However, I used <0.5% as a no-presence indicator for all area-based threats, including urbanized, industrial, row crop, and pastureland cover types. When possible, the frequency of impacts associated with sharp changes in biotic response (i.e., degradation thresholds), as indicated in the literature, were used to classify the impacts associated with the frequency classes. For example, degradation thresholds of 2% (minimal), 10% (changes in biotic response), and 50% (complete degradation) have been cited for both urban and agricultural land uses (Finkenbine and Mavinic 2000; Fitch and Adams 1998; Wang et al. 2000), and the presence of even one dam is considered a significant disruption to stream integrity (Ligon et al. 1995). If degradation thresholds were not evident in the literature, threat-frequency classes were assigned by dividing the full range (except zero) of observed frequencies into three equal intervals.

The degree of severity of a threat is more difficult to quantify and depends on several factors, including its 1) frequency within a catchment, 2) proximity to vulnerable conservation targets, and 3) any mitigating or aggravating factors (e.g., two mine sites near 2 manufacturing sites) present. The severity score is characterized by using expert judgment and biological knowledge to quantify how threats potentially could alter the five major drivers of ecological integrity within individual catchments or collectively across a larger spatial extent. In my study, severity scores were assigned for each of the 12 major threats, with no regard for cumulative or synergistic impacts. Severity scores were derived from questionnaires given to three experts, who scored the potential impact (low [1], moderate [2], or high [3]) of each threat on the flow regime, water chemistry, production pathways, physical habitat structures, and biotic interactions of streams within the study area. It was assumed that as long as a threat was present, it would have at least a low impact on nearby streams. The severity scores were summed (the potential range was 5-15) for each threat, and identical severity scores were used for all catchments in the study area for the purpose of model testing. The potential impact of a threat within a specific catchment was computed by multiplying its threat-specific severity score by its threat-specific frequency class (Figure 2.1). This product is the threat-specific risk (ERI-T) score. Summing all ERI-T risk scores produces a composite (ERI-C) risk score for each catchment that then was classified into three final risk rankings for ease of mapping and management purposes. It is this composite score and risk ranking procedure that is examined here.

Testing robustness

The robustness of the ERI was evaluated to test the hypothesis that the scoring process and the use of expert judgment yielded repeatable, meaningful scores. Specifically, the analysis was used to 1) individually address the frequency and severity components, and 2) compare the outcome of my methods to that of a systematic approach of assigning frequency and severity classifications. My analysis considered three aspects of the ERI: 1) threat-frequency classes, 2) the number of final risk rankings used to group ERI-C scores, and 3) the role of expert judgment in a rank-based risk model. For each aspect, the insensitivity of ERI outputs to variation in ERI inputs is a measure of ERI robustness.

Comparison of frequency classification methods

Frequency classifications of the ERI were compared with those of a quantile-based approach by altering the method used to classify the frequencies of impacts within catchments of the UTRB. Quantiles were chosen over other classification methods so that it was not confused with the original ERI approach, which uses equal intervals for assigning some threat frequency classes. Catchment-specific threat frequencies used in the original ERI computations were reassigned into new frequency classes by ranking all non-zero values into 33rdpercent-quantiles, which provided an unbiased basis for comparison to the original approach. Both approaches used an ordinal (0, 1, 2, 3) scoring scheme for assigning frequency classes to a catchment. Each approach used a threat matrix in which rows represented catchments within the UTRB (n=107), and each column was a threat (n=12) assigned to a threat-frequency class as described above. Threat-frequency matrices for the two computational approaches were compared to evaluate differences in the number of catchments within each risk level (none, low, moderate, high) of the final risk rankings.

Testing procedures interpreted the differences in final risk rankings between the two classification approaches in order to indicate the robustness of frequency class designation on final risk rankings. Differences in frequency-class assignments were evaluated in three steps. First, ERI-C scores were computed using each approach, and a cross-tabulation of the final risk rankings for all 107 catchments within the UTRB was performed to test for differences between model outputs (Pearson χ^2, d.f.= 4; using SYSTAT, v. 11). Second, final risk rankings were mapped to identify catchments in which risk reclassification occurred. Third, differences between model outcomes were summarized by tallying the degree (number of levels) and direction (positive or negative) of risk level changes in the final risk rankings of both approaches.

Effects of expert opinion on severity scores

The severity score used in computing threat-specific (ERI-T) risk scores was quantified by having experts determine whether an identified threat has a low, moderate, or high impact on each of the five biotic drivers of streams. The original ERI approach assigned severity scores by having three experts complete a questionnaire regarding the potential impacts of individual threats on biotic drivers, and then calculating a single ERI-T score by averaging those scores (n=3). The same experts then were asked to consider scenarios in which each threat incurred minimum and maximum impact(s) on biotic drivers, and then they assigned a minimum and maximum severity score to each threat.

I tested the influence of expert opinion on ERI-C scores by altering assigned threat-specific severity scores using Monte Carlo simulations. The procedure employed the average severity score for each threat (see above) as the apex of a triangular distribution, and the expert-derived minimum and maximum severity scores were averaged and assigned as the lower and upper bounds, respectively, of the distribution (Appendix 2.A). In two cases, the original average score was greater than the mean maximum reported by experts in the later survey, so in these cases, the average score served as the most likely score and the upper bound. A single simulation incorporated the triangular distributions of all 12 threats.

It was necessary to include the frequency classifications obtained by the original ERI and quantile-based approaches into the Monte Carlo simulations so that differences in ERI-C scores would be recorded. For each method, every simulation (1 simulation = 10,000 iterations) included output that recorded ERI-C scores each iteration, and then counted the number of iterations in which a catchment had a low, moderate, or high ERI-C score (i.e., the probability that a catchment belongs to a particular risk level). Next, a homogeneity test (χ^2, d.f.= 2) assessed the effect of expert judgment on the distributions of the two approaches (Table 2.1). If the ERI-C distributions were similar, expert judgment would be regarded as having a similar effect on the final risk rankings for both approaches.

I also tested the effect of expert judgment on the final ERI-C scores by comparing the original ERI approach to randomly assigned severity scores. As before, the average severity scores (n=3) were used in a triangular distribution (@Risk software 4.5, Palisades Corp.) in which these scores were the most likely scores. Next, random distribution bounds were assigned within the Monte Carlo simulation such that any score between 5 (the lowest possible) and 15 (the highest possible) was chosen. No knowledge of the bounds provided by the experts was used for this analysis. The objective was to determine how the ERI-C scores were affected by potentially arbitrary assignments of severity scores by experts. One-way ANOVA (d.f. = 2, 9998, $P < 0.05$)

was used to compared mean frequencies of ERI-C scores between randomized severity scores versus the original ERI output to assess the effects of arbitrary expert assignments.

Appropriateness of threat risk categories

The appropriateness of using three final risk ranking levels (low, moderate, high) was assessed by applying multivariate analyses to the original ERI and quantile-based approaches. *K*-means cluster analysis was used to identify the optimal number of risk levels, and Chi-square distance sampling was performed on two, three, and four potential risk levels (Legendre and Legendre 1998). Prior probabilities were assigned in *K*-means cluster analysis to reflect a plausible distribution of the population among groups. There was no known pattern of threats across the UTRB that justified specific priors, so I used equal assignment probabilities for each risk-level group. The comparison between the original ERI and quantile-based approaches identified the grouping patterns for both frequency-class approaches. The optimal number of clusters was identified by comparing plots of error rates from goodness-of-classification-fit and jackknife-classification matrices. Box plots of standardized ERI-T scores based on three clusters were also plotted to visually compare cluster means from the two approaches ($\bar{x}$) using the expression:

$$(x_i - x_{min})/(x_{max} - x_{min}),$$

which resulted in all threat frequencies having a range of 0-1 (Legendre and Legendre 1998).

Lastly, after the optimal number of risk ranking levels was determined, I identified the individual threats driving the cluster groupings. Catchments were assigned to clusters based on results of the cluster analysis, and then backward-stepwise discriminant analysis (SDA) was performed on the ERI-T scores from the original and quantile-based approaches (*F* to enter/remove = 0.15). A bivariate plot comparing the jackknife method with a goodness-of-classification-fit cross-validated the error rates for the two approaches. ANOVA was performed on those threats identified as important in the SDA to test for differences in their means among final risk cluster groups. Canonical scores were also plotted to visually assess group separation. Collectively, these tests were used to characterize the variation associated with risk-level selection between approaches.

RESULTS

The ERI uses a series of classification methods to derive an ecological risk-based index for use in conservation planning. Expert opinion was considered an appropriate and useful component of the index. The final risk rankings varied depending on the classifications used within frequency estimates, and expert opinion had a more conservative outcome than did the quantile approach. Results showed that ERI-C scores were responsive to the method used to define threat frequency classes. Differences in severity score assignment indicated that an expert-based scoring system is a key factor in addressing the degree to which threats impact individual catchments. Results of my investigation of severity score assignments also suggested that expert judgment was more conservative in assigning risk than was an assigned classification system such as the quantile-based approach. Additionally, expert judgment produced more conservative risk rankings than did randomly assigned severity scores. Several threats were especially influential in determining ERI-C scores and risk rankings for each analytical approach, including industrial and manufacturing sites common to both. Lastly, use of three final risk ranking levels provided the most useful information for mapping and management purposes.

Frequency classifications and severity scores

Evaluation of the role of expert opinion on final risk scores suggested that experts judged several threats to have similar degrees of impact on stream communities. The reliability of local biological data influenced the assignment of risk, and experts identified fewer high-risk catchments than low-risk catchments (Figure 2.3). Mean differences in final risk rankings between the respective analytical approaches were statistically significant at all risk levels (P <0.001). In addition, the distribution of ERI-C scores reflected the expert knowledge used in assigning severity scores and frequency classes. Monte Carlo simulations of the average severity score distributions indicated that the ERI was sensitive to the risk ranking methods used, and that expert judgment affected the distribution of catchments among risk levels (Appendix 2.A). Overall, experts assessed risk to be lower than risk rankings derived from a quantile-based approach.

The original ERI and quantile-based approaches differed in final risk ranking characterizations (χ^2= 66.587, P < 0.001). The majority of catchments (24 of 38) reassigned by the quantile-based approach were classified as moderate risk by the original ERI. I found that 64.5% of the 107 catchments exhibited the same final risk ranking (i.e. low, moderate, or high) regardless of the approach used. Thirty-eight catchments (35.5%) shifted to a neighboring risk level; none shifted from low to high or vice versa (Figure 2.4). There were 67, 17, and 23 catchments in the low, moderate, and high risk levels, respectively, for the original ERI approach, and 55, 27, and 25 catchments in analogous risk ranking levels for the quantile-based approach. Most of the catchments that differed in risk levels (23 out of 38) between approaches shifted from moderate risk in the original ERI approach to high risk for the quantile-based approach. Also, 14 catchments shifted from low to moderate risk, and one shifted from moderate to low risk. None of the four catchments originally assigned to high risk by the original ERI shifted risk level in the quantile-based approach. Overall, cross-tabulation characterized ecological risk as being more prevalent using the quantile-based approach than in the original ERI approach.

ERI-C scores and ranks

Results of cluster analysis indicated that the ERI consistently distinguished among multiple risk levels. Group means were significantly different for all *K*-means clusters (P<0.001) for all tests (Table 2.2), which indicated that individual threats might be identified as key contributors to stream degradation. Plots of final risk rankings indicated that risk levels were misclassified in goodness-of-fit and jackknife classification procedures 6% to 24% of the time, with considerable differences between the original ERI and quantile-based approaches (Figure 2.5). The plots suggested that either two or three final risk ranking levels are justified for both the original ERI and quantile-based approaches (Figure 2.5). Because only one of the error tests suggested that four risk levels might be justified, the use of four risk levels was judged as unreliable. The results indicated that three risk levels, as in the original ERI, were most appropriate and informative, and three risk classifications were retained in subsequent analyses.

Clusters associated with the final risk rankings indicated that individual threats had unique patterns among risk ranking levels, suggesting that risk rankings might distinguish among the degrees of impact related to individual threats (Figure 2.6). Additionally, the relative importance of most major threats varied significantly following Kruskal-Wallis tests among clusters (P values < 0.01) for frequency classes in both approaches, but exceptions included pasturelands (P = 0.06) and row crops (P = 0.08) for the ERI approach, and road density (P = 0.08) for the quantile-based approach. Boxplots showed that although individual threats may be represented at all risk levels, there is a tendency for each threat to be aggregated at a single risk level (Figure 2.7).

Those threats identified as important contributors in defining cluster space differed between the original and quantile-based approaches, with only industrial areas and manufacturing sites common between them. Those threats remaining important in backward stepwise discriminant analysis (Table 2.3) for the quantile-based approach were row crops, industrial areas, dams, manufacturing sites, and road density, whereas urban areas, industrial areas, mines, manufacturing sites, and waste facilities largely determined risk classes for the original ERI method. Discriminant analysis testing also indicated that the two analytical approaches differed with respect to how threat-type was considered in cluster space. Canonical plots of the final risk rankings showed a clear separation among risk levels for both approaches (Figure 2.6). Site-specific data were helpful in discriminating between low- and high-risk areas for the original ERI approach, whereas area-related data aided in identifying moderate-risk areas (Table 2.4). For example, site-specific data such as dams, NPDES sites, bridges, and mining sites, were helpful in discriminating between low- and high-risk areas for the original ERI approach, whereas agricultural lands and other area-related data aided in identifying moderate-risk areas (Table 2.4). Similar patterns were apparent for the quantile-based approach, except that threats associated with urban areas and dams separated low-risk scores from the other risk levels.

Backward-stepwise discriminant analysis revealed that frequency class assignment is an important factor when using the ERI. Plots of cluster space suggested that low-risk scores were distinct from moderate- and high-risk scores for both approaches (Figure 2.6). However, moderate- and high-risk scores were more distinct with expert judgment (Figure 2.6a) than were those scores when the quantile-based approach was used (Figure 2.6b). There was also evidence that the low-risk ranking level for the quantile-based approach may best be separated into two risk groups based on whether impacts are area-based or point data (Figure 2.6b).

DISCUSSION

The ERI is a practical approach for summarizing anthropogenic impacts and ranking the conservation potential of freshwater ecosystems. The index is unique because it combines qualitative data from biologists with information about threat frequency and literature-based degradation thresholds for estimating the degree to which individual threats are likely to impact catchments. The ERI is subject to the general criticisms voiced about other risk-based planning tools (e.g. Wolman 2006), and it is similarly sensitive to the methods used to classify risk within its framework. My purpose in this study was to compare the results of quantitative and qualitative data in ERI model outputs to better inform conservation planners of the risks and benefits of using a rank-based risk planning tool.

The task of assigning risk within a catchment containing multiple threats is given to wide error margins, especially since experts may not have stream-specific information on ecological degradation, quantitative impact assessments, or even a species inventory. Conservation planning continues, however, and managers develop tools that summarize threats and assign potential risk to conservation targets (Game et al. 2013). Putting aside the qualitative nature of rank-based approaches, their great utility (and appeal) lies in their ability to summarize large quantities of spatial data in a relatively short timeframe (Bottrill and Pressey 2012). Moreover, expert judgment and subjective classification techniques may compensate for absent biological data when assigning degradation thresholds (Wilson et al. 2005). However, the method used to assign model parameters does impact final risk rankings. For example, differences in ERI risk-class assignment between the two analytical approaches that I tested were largely a result of the influence of expert judgment on frequency-class assignment and impact severity scores. I assigned threat-frequency classes to represent the distribution of human impacts across the landscape, whereas the intensity of those threats was quantified using severity scores. I relied on expert judgment to assist in assigning severity scores within individual catchments.

The ERI is sensitive to both the process used to classify frequencies and the use of expert judgment. A greater number of catchments was classified as low risk when results for the expert-derived ERI was compared to a quantile-based approach, meaning that expert judgment is useful in clarifying the scope and perceived threat of potential impacts. This conclusion is further supported by my finding that similar threats influenced risk groups in both the original ERI and quantile-based approaches. I suggest that expert opinion is most valuable when catchments must be prioritized for management purposes, especially when human-caused impacts are locally well-documented (Dudgeon et al. 2006). This means that the ERI and similar approaches prove useful for predicting further conservation challenges based on a general inventory of threat frequency and conservation targets when research interests are being prioritized within the conservation planning process.

Strengths and weaknesses

As with any index, the ERI has a number of strengths and weaknesses. It is a good organizational tool, and is most appropriate for providing a general overview of how human activities may be impacting conservation targets within a catchment. The elasticity of its model inputs makes the ERI amenable to studies with few quantitative data, and the index may be applied at any spatial grain. These attributes also contribute to the limitations associated with rank-based approaches. First, it is important to have the ability to classify threats, but the means used to classify risk may be subject to best judgment as opposed to peer-reviewed analysis of empirical datasets. This issue may not be important if the index is simply used as a planning tool, but conservation measures may be questioned if the use of expert judgment is not validated. As a planning tool, the ERI may be quite helpful in guiding researchers to identify data needs and pragmatic research opportunities.

One of the advantages of the ERI is that there are opportunities to delete or add individual threats, severity scores, and frequency classes so that the model may conform to a specific study area or research question. Since the ERI projects the potential risk of specific threats to a conservation target, the model inputs (i.e., the threats) may be adjusted to compare specific geographic areas, varying conservation targets, or to reflect changing threat patterns. In all cases, the impacts of threats must be quantified into risk categories. I suggest that similar ecological impacts may be summarized collectively if there is equal risk potential. Also, long-term datasets, such as stream monitoring data or species collection data, may be used in conjunction with the ERI as part of an overall conservation plan. Data collection would be warranted when the severity and scope of individual threats are not well known. For example, the ERI may provide a general overview of indicated threats within a large catchment, but the proposed risk associated with an individual threat may be under- or overstated because the actual scope of the impact is unknown. Additional data would prove helpful in this instance for identifying geographic areas with the most conservation potential. Finally, I suggest that readily accessible land use data are a good substitute for summarizing recent human activities within a target area, even if the area is geographically small, and since most land-use activities have known impacts upon the surrounding landscape, geospatial data and published threshold values are typically adequate substitutes for field-collected data (Theobold et al. 2010).

The ERI is a planning tool that informs the conservation community by summarizing the scope and severity of human activities into a format that is readily interpreted. The ERI is easily suited for large projects in which risk rankings are mapped for a variety of threats. The ERI helps organize potential and existing threats by severity and scope, and enables researchers to determine places within a study area that may need more conservation effort than others. Also, the ERI is flexible in that it can be applied multiple times if experts agree that an impact varies over a region. The index does not predict specific risk probabilities, using categories instead, and does not predict pollution events (factors in potential risk), but it does provide a potentially explicit output of threat patterns and potential risk across a study area. This is an important step for planning purposes and a conservative step in the development of conservation plans, and may help focus endangered species management, habitat preservation, or restoration potential.

It is clear that expert judgment has a useful role in conservation planning and biodiversity conservation. Although biological knowledge cannot be substituted for, experts understand the ecological systems that they study, even in instances when data collection is impractical (Halpern et al. 2007). Hence, the ERI is advantageous for assessment of threats to areas with a dearth of data, but in which experts have a pretty good idea of what is threatening an ecosystem. The ability to summarize information and score potential risks increases everyone's understanding of real and potential impacts, and may direct conservation efforts. I suggest that the opinions of multiple experts be used within the conservation planning process to identify threats and rank their real or perceived impacts. This approach provides a reliable source for model inputs and a means of supporting the final risk rankings. Obviously, having multiple experts involved in identifying threats and scoring impacts leads to a more robust average impact score.

Mapping threats across a study area affords researchers the opportunity to recognize current land or water-use patterns and to predict areas in which human activities will be most influential (Jackson et al. 2004, Halpern et al. 2007). In areas in which use of indices such as the ERI are warranted, there is much ecological knowledge of the system by experts and local communities, but not necessarily field survey data to verify claims. The ERI and similar indices provide tools for providing an overview of present threats and associated risks when field-collected data are unavailable, but the ERI may also direct field studies by identifying data deficiencies. Another advantage to mapping threats within the ERI framework is that spatial patterns emerge, and managers can use this information for prioritizing conservation efforts. For example, there were obvious trends in threat pattern among risk groups within the UTRB. Catchments with higher risk levels did not necessarily contain more severe threats; instead, threats tended to be spatially clustered. For example, catchments with mining operations tended to have a greater number of waste facilities and non-point source pollution areas, while developed areas were associated with greater road densities and industrial areas. Moreover, as threat frequencies increased, so did the likelihood of assigning higher risk. Low-risk catchments had fewer risks overall and fewer high ranking risks. These areas were also more likely to be vegetated. Therefore, mapping threats and understanding the drivers behind ranking categorization is an important step for using the ERI reliably.

Informing conservation

An important consideration when using ranking procedures is defining how risk is interpreted in the final stages of threat assessment. Catchments posing very low or extremely high amounts of risk to biotic integrity were easily identified, whereas the appropriate number of intermediate risk groups may be debatable (Smith et al. 2006). Those catchments that were least and most impacted were clearly separated into discrete risk classes and their current ecological condition was obviously different from one another (Van Sickle et al. 2006), whereas the intermediate groups may be considered as part of a risk continuum. An additional step, such as linking current land or water uses with existing levels of stream condition, may be warranted for catchments with moderate risk rankings to better distinguish conservation priorities (Wiegers et al. 1998, Van Sickle et al. 2006). The ERI uses expert opinion to distinguish those catchments with moderate risk that were more (or less) likely to respond favorably to conservation actions, thus altering the typical management scenario from reactive decision-making to one in which adaptive management is applied to catchments containing a range of ecological conditions (Game et al. 2013, Tockner et al. 2010).

There are some caveats associated with using the ERI. The ERI has a few unique attributes that make it a flexible tool for conservation planning. One is that it considers the risk(s) associated with individual threats. For example, experts were asked to consider the potential impact(s) posed by individual threats on the five biotic drivers based on regional geomorphic characteristics and current threat locations. This approach to risk assessment allows experts to determine the vulnerability of specific conservation targets. Since the ERI focuses on patterns and occurrences of threats, it can apply a coarse-scale approach, although additional parameters, such as soil erosion, might be added to account for physical land features and unique catchment attributes. Another caution is the focus on biotic integrity as a conservation target instead of having a species-based approach. Additionally, although using ecological integrity as an assessment endpoint encourages a holistic view of catchment condition, there may be different approaches for determining the impacts of individual threats as opposed to aggregated threats at larger spatial scales. The ERI was able to map the potential impact of individual threats across a range of catchments because of its broad purpose, so that biotic responses to the presence of anthropogenic activities were deliberately generalized (Van Sickle et al. 2006). However, if stream-related impacts had been summarized per catchment, the resulting severity scores would have better reflected degradation of biotic conditions within individual catchments. The addition of such detailed data would aid in focusing conservation actions, but is not compulsory for applying risk ranking techniques such as the ERI.

Management Implications

Quantitative data related to threat identification and the estimation of actual and probable risk is very appealing to managers because it is a repeatable process with scientific underpinnings

(Landis 2003). However, applying risk-based approaches to large spatial extents and mapping multiple threats across complex systems obscures the ability to identify specific causes and quantify impairments (Rashleigh 2004). Even at larger spatial extents, conventional risk assessments include probability estimates for ecosystem degradation and tend to relate quantifiable exposure thresholds, such as those for heavy metals, to risk estimation (Fisher et al. 2001). Conservation planning may be more inclined to focus on summarizing general land and water uses which have known or suspected negative ecological impacts to which quantified threshold values are difficult to determine due to the uncertainty of relating local impacts to a range of spatial grains. For example, it is widely known that agricultural lands negatively impact local streams (Roth et al. 1996), but there is much uncertainty regarding how those impacts relate to ecological resilience and ecological function at various spatial grains (Allan and Johnson 1997). A model such as the ERI may be improved, however, as more reliable threshold relationships are identified for larger spatial scales. For example, my application of subjective classification techniques, such as using equal intervals when assigning frequency classes, would be more informative if threat-specific threshold data were available. Furthermore, data from additional field surveys and monitoring studies focused on biotic degradation would enhance the reliability of threshold values, but the information gain may not be worth the resources needed to obtain it.

It is not necessarily recommended that every risk-based ranking procedure be studied quite as extensively as the ERI. I do recommend that other researchers using ranking procedures evaluate their methods, and more fully understand the strengths and weaknesses of summary indices to address criticisms related to risk assignment and vulnerability indices. Spatially referenced data and expert judgment are likely candidates for inclusion in ranking protocols designed to inform coarse-scale assignments of conservation priorities (Higgins et al. 2005, Zhang and Chen 2014). Expert judgment is useful in risk-based analyses, and planners who are unsure of how to construct a risk-based tool of their own might apply multiple assessment pathways to confirm results, such as comparing multiple ranking procedures, varying model inputs of a specific approach, or swapping data to compare results (Halpern et al. 2007, Paukert et al. 2011). Rank-based risk assessment tools are valuable to conservation planners and managers because they are a cost-effective and flexible means of identifying areas with the least or greatest potential for supporting a full complement of native biota.

Including quantitative parameters into a conservation planning framework is a challenge in large watersheds containing varying degrees of anthropogenic stressors. Determining the vulnerability of such watersheds to anthropogenic stressors is a priority, and obtaining relevant, high-quality data proves to be a challenge. Easily obtainable data, such as satellite imagery, are a good resource for assessing land use/cover inquiries, whereas additional resources are required for characterizing impacts throughout a catchment. Obviously, having quantitative threshold data for individual threats would decrease the uncertainty associated with the ERI, link threat severity to declines in stream integrity, and contribute to a practical and cost-effective approach to conservation management. Conservation planning frameworks should include methods for summarizing multiple threats to biodiversity or ecosystem processes, but few projects with such broad objectives are manageable without including qualitative data within in the process. A feasible compromise to collecting quantitative data for large catchments is the use of expert opinion in assigning ecological vulnerability within the planning process so that relevant management options may address conservation issues.

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River, Virginia and Tennessee, USA: A featured collection. Journal of the American Water Resources Association, 50, 807-819.

Table 2.1. Chi-square test for homogeneity of Monte Carlo simulation results (10,000 iterations) comparing ERI-C scores computed via the original ERI approach versus a quantile-based approach. Percentages of each risk level for each approach are presented. Observed versus expected risk levels (low, moderate, high) for each approach were tested. Chi-square = 16.88 (2 degrees of freedom, $P<0.001$).

	Risk rankings		
Observed	Low	Moderate	High
ERI	50.41	49.05	0.54
Quantile	32.96	52.37	14.67
Expected	Low	Moderate	High
ERI	41.69	50.71	7.60
Quantile	41.69	50.71	7.60

Table 2.2. Group means for 3 risk levels (low [1], medium [2], high [3]) computed from final ERI-C values for the original ERI and quantile-based approaches. The number of catchments in each risk level is shown in parentheses, and the Wilks-lambda statistic tested for equality of group means using ANOVA. Area-based threats are grouped by percents and point-based threats are grouped as integers. NPDES stands for National Pollutant Discharge Elimination System. Also shown are the number of catchments in which each threat is present.

	Number of catchments	Risk levels					
		Original ERI			Quantile		
		1 (67)	2 (17)	3 (23)	1 (55)	2 (27)	3 (25)
<u>Major Threats</u>							
Row crops	47	6.90	4.94	4.87	29.02	20.22	11.76
Pasturelands	91	16.42	10.35	12.91	24.40	18.74	15.40
Urbainzed areas	44	11.28	0.00	0.00	33.09	9.33	9.52
Industrial areas	16	2.87	0.00	0.00	30.98	0.00	0.00
Mining sites	106	18.81	0.00	0.00	15.71	0.00	8.16
Waste facilities	67	25.43	0.00	5.74	21.60	4.44	10.56
Stream crossings	102	26.33	13.41	22.96	24.87	15.56	25.92
Impoundments	43	15.22	0.00	7.17	14.18	0.00	16.20
Manufacturing sites	86	26.10	0.00	13.87	23.60	5.70	16.72
NPDES sites	85	28.48	6.35	15.13	24.22	6.22	17.76
Railroad crossings	79	7.00	5.35	4.26	13.36	6.74	7.84
Road density	107	17.46	20.65	17.22	17.67	19.33	15.48
Wilks' lambda		0.21 (df = 12, 2, 104)			0.07 (df = 12, 2,104)		
F- statistic		9.24 (df = 24, 186)		*P* <0.001	22.74 (df = 24, 186)		*P* <0.001

Table 2.3. Results of backward-stepwise discriminant analysis of ERI-C scores, which were computed via the original ERI approach and a quantile-based approach. NPDES stands for National Pollutant Discharge Elimination System.

Original ERI

Step	Number in	Removed	*F* to Remove	Tolerance	Wilks' Lambda	*F*-statistic
0	12	None				
1	11	Pasture	0.12	0.50	0.21	10.16
2	10	Major dams	0.34	0.74	0.21	11.22
3	9	Row crops	0.65	0.84	0.21	12.45
4	8	Road density	0.8	0.92	0.22	13.93
5	7	NPDES sites	1	0.47	0.22	15.78
6	6	Bridges	0.78	0.75	0.23	18.32
7[1]	5	Railroad density	1.07	0.94	0.23	21.38

Quantile-based

Step	Number in	Removed	*F* to remove	Tolerance	Wilks' Lambda	*F*-statistic
0	12	None				
1	11	Pasture	0.14	0.32	0.07	25.02
2	10	Mines	0.25	0.67	0.07	27.72
3	9	Railroad density	0.31	0.90	0.07	30.99
4	8	Urbanized areas	0.5	0.68	0.07	34.98
5	7	NPDES sites	0.28	0.90	0.07	40.10
6	6	Bridges	1.56	0.60	0.07	46.27
7[2]	5	Waste facilities	1.77	0.41	0.07	54.76

1. Variables remaining are: urbanized areas, industrial areas, mines, manufacturing sites, and waste facilities.
2. Variables remaining are: row crops, industrial areas, dams, manufacturing sites, and road density.

Table 2.4. Canonical discriminant functions for ERI-C scores computed via the original ERI and quantile-based approaches. Threat comparisons are made within risk groups (low, moderate, high). NPDES stands for National Pollutant Discharge Elimination System.

	Original ERI		Quantile	
	Function 1	Function 2	Function 1	Function 2
Constant	2.766	0.347	1.869	0.309
Major threat				
Row crops	0.022	0.02	0.004	0.04
Pasturelands	-0.006	-0.016	0.008	-0.012
Urbanized areas	-0.064	0.059	-0.013	0.005
Industrial areas	0.036	-0.098	-0.185	0.013
Mining sites	-0.017	0.051	0.006	-0.008
Waste facilities	-0.01	0.065	0.002	0.057
Bridges	0.032	-0.024	0.007	-0.027
Impoundments	-0.008	0.001	-0.015	-0.02
Manufacturing sites	-0.097	-0.142	-0.027	-0.073
NPDES sites	-0.023	0.015	0.002	-0.026
Railroad crossings	0.038	0.063	0.008	-0.014
Road density	0.019	0.025	0.027	0.037
Eigenvalues	9.763	0.438	2.513	0.368
Can. correlation	0.952	0.552	0.846	0.519
Cumulative %	0.957	1.0	0.872	1.0

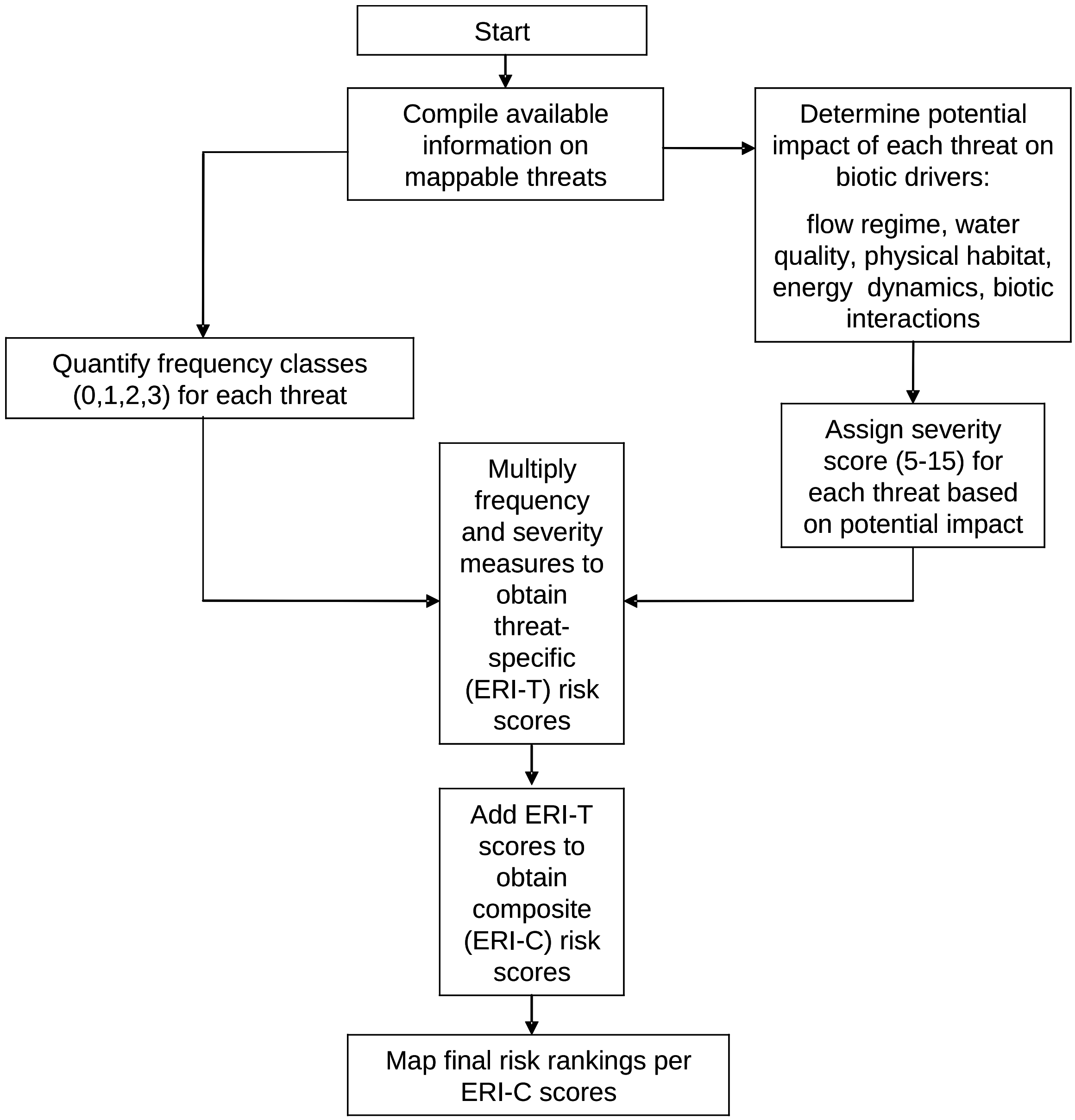


Figure 2.1. Flowchart summarizing the Ecological Risk Index (ERI) protocol. Threat data may originate from various sources, including land use and land cover databases, point- and non-point pollution data, and other local and regional sources of anthropogenic stressors.

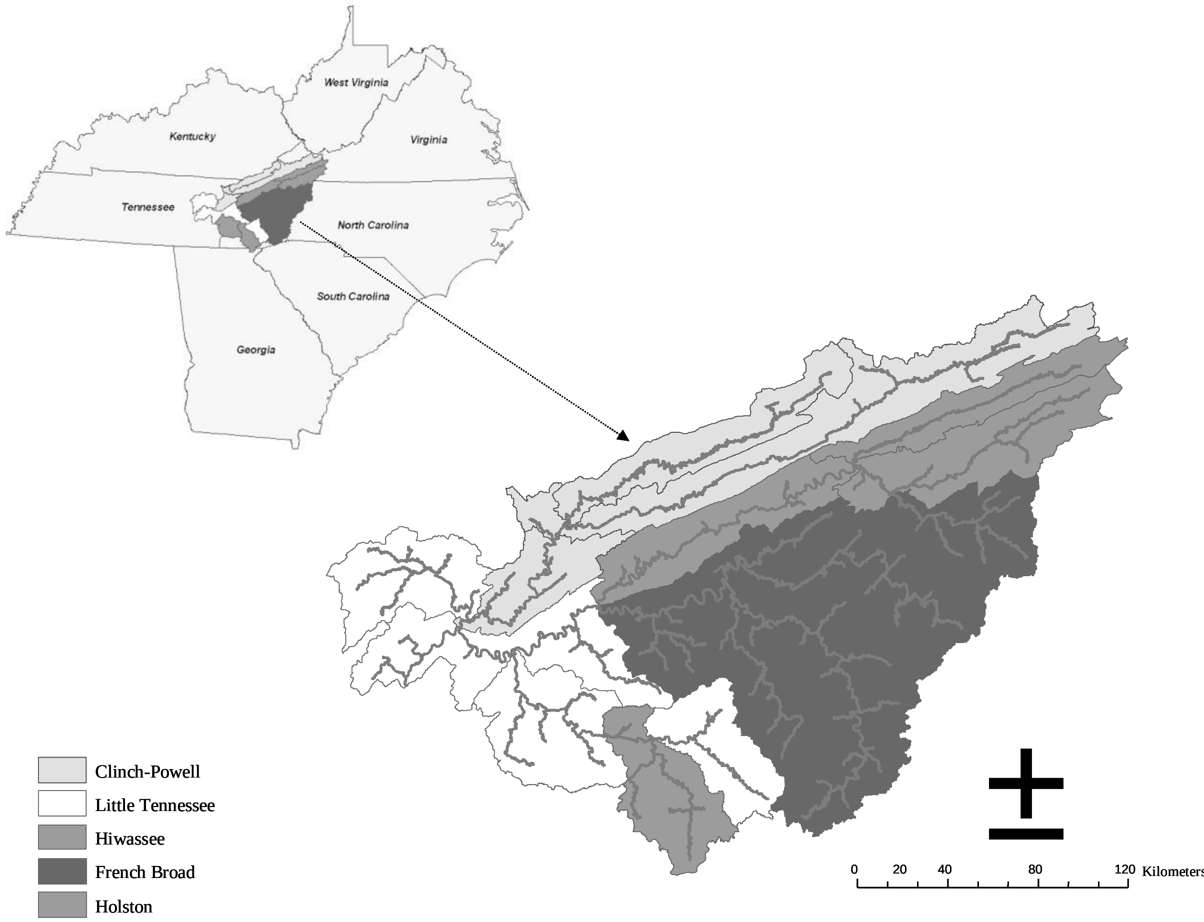


Figure 2.2. Map of the Upper Tennessee River Basin (UTRB) located in the southeastern USA. Catchments depicted are based on 4^{th}-order streams and above (Strahler method). Only major streams are shown. The five major drainages within the UTRB are shown.

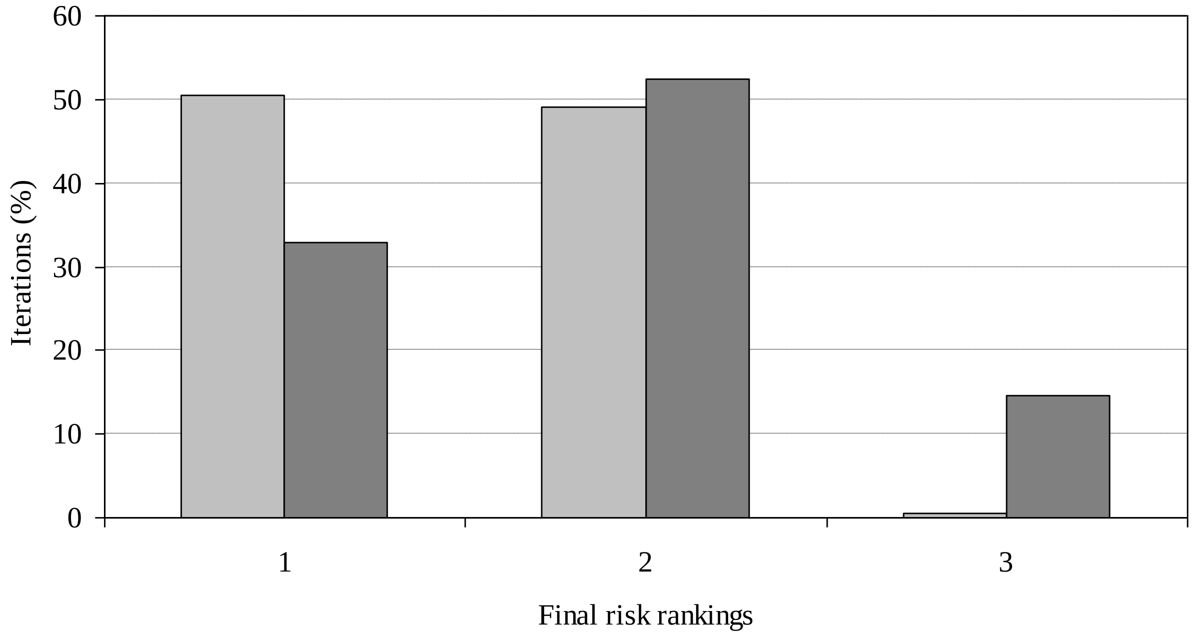


Figure 2.3. Summary of a Monte Carlo simulation (10,000 iterations) of 107 ERI-C scores computed from the original ERI approach (light grey) and a quantile-based approach (dark grey). Percentages of scores in low- (1), moderate- (2), and high-(3) risk levels are shown. One-way ANOVA tests were significant ($P < 0.001$) for each risk level.

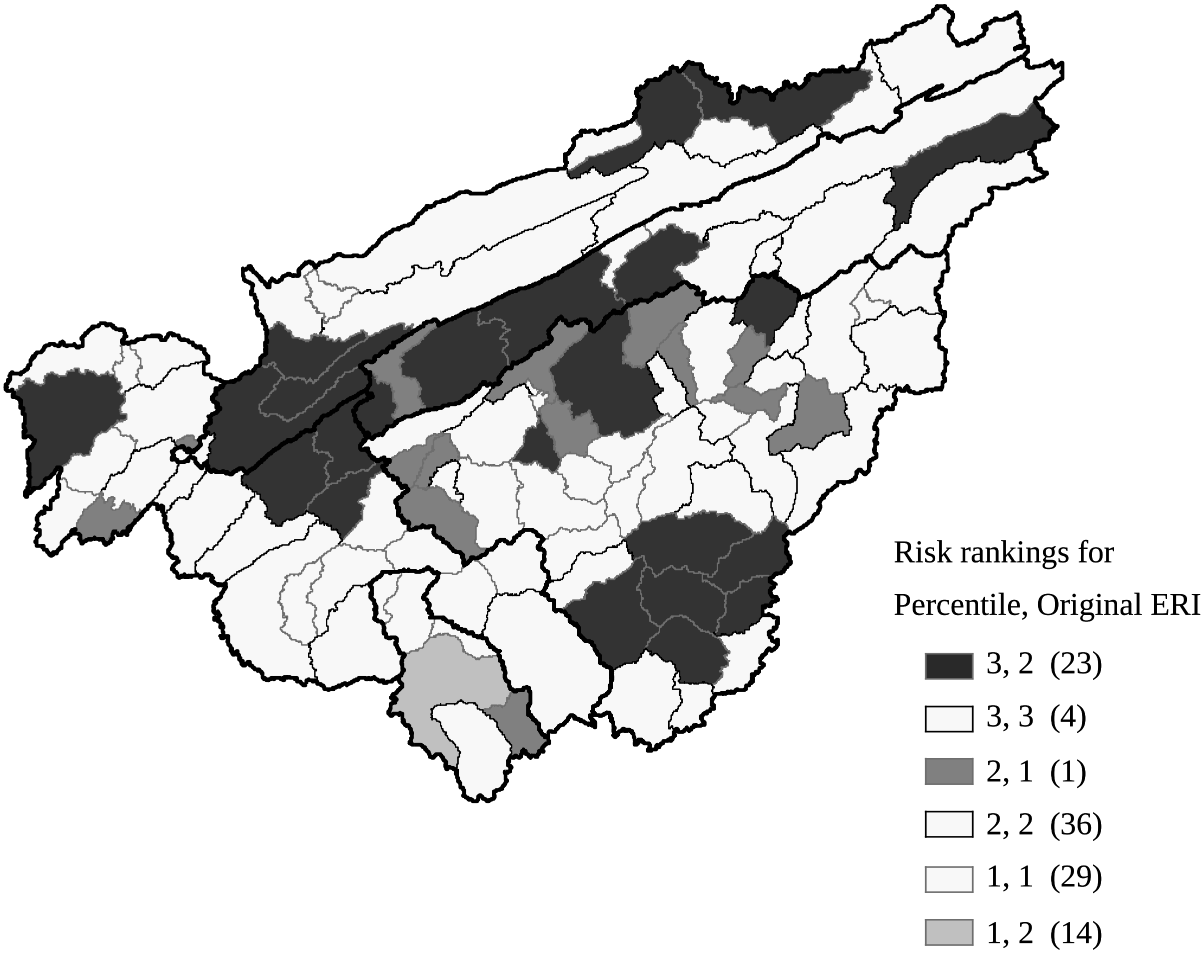


Figure 2.4. Comparison of final risk rankings derived from ERI-C scores for the original ERI approach versus a quantile-based approach for computing threat-frequency classes. ERI-C scores are aggregated into low-risk (1), moderate-risk (2), and high-risk (3) levels to facilitate mapping. Number of catchments in each risk level is in parentheses.

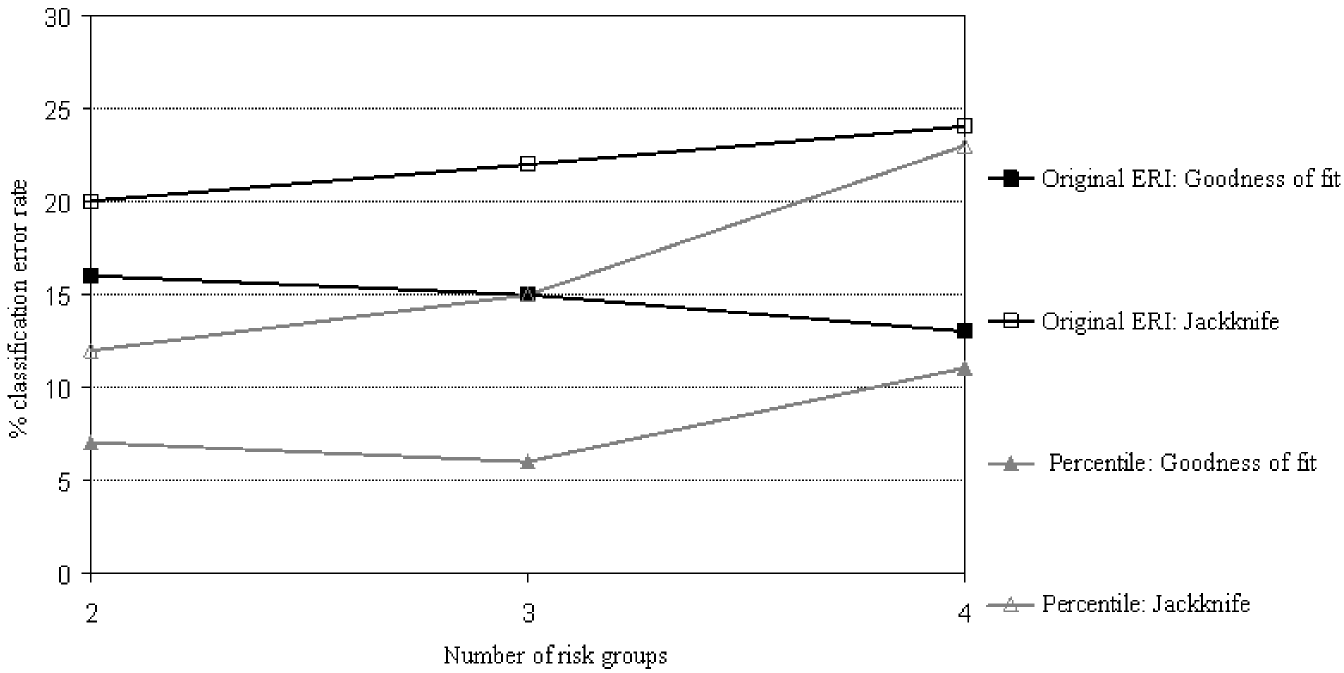
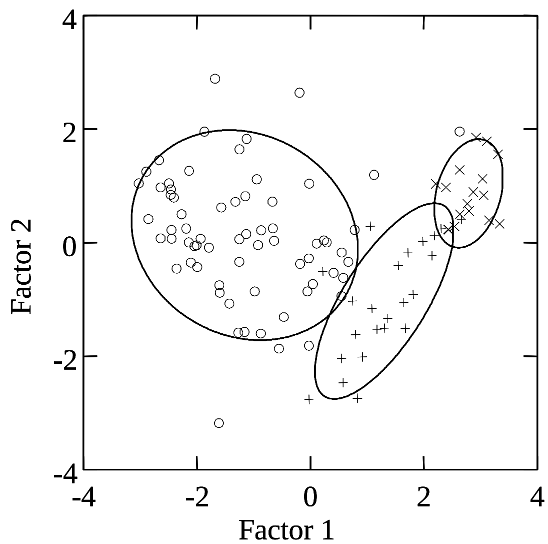


Figure 2.5. Classification of risk groups using goodness-of-fit and jackknife results for the original ERI and quantile-based approaches. Risk groups refer to the number of clusters obtained from *K*-means cluster analysis.

a.



b.

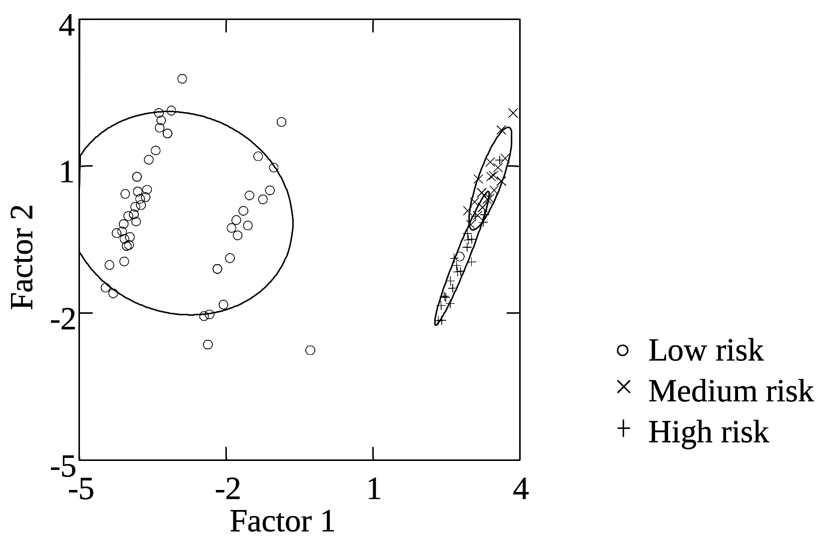
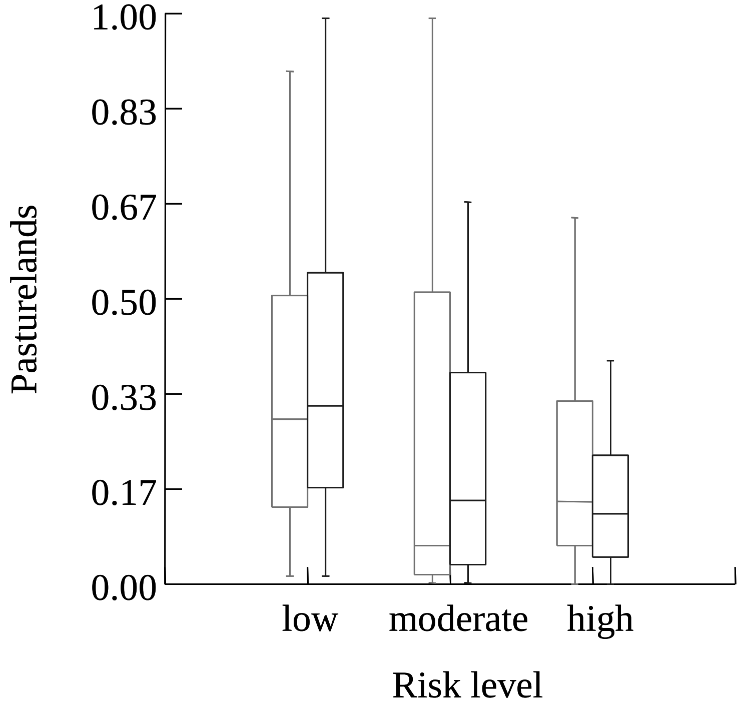


Figure 2.6. Plots of canonical variable scores from discriminant analysis for 107 ERI-C scores computed from (a) the original ERI approach, and b) a quantile-based approach. 95% confidence ellipses for scores are aggregated by low-, moderate-, and high-risk levels. For a), factor 1 represents a continuum from urban-related threats to bridges and road density (from left to right) and factor 2 represents industrial-related activity to railroad density and waste treatment facilities. Factor 1 in the quantile approach (b) represents developed areas to road density and pasturelands, and factor 2 is influenced by site-specific data such as impoundments and manufacturing sites to area-based threats such as agriculture and road density.


1.00
0.83
0.67
0.50
0.33
0.17
0.00
Pasturelands
low
moderate
high
Risk level

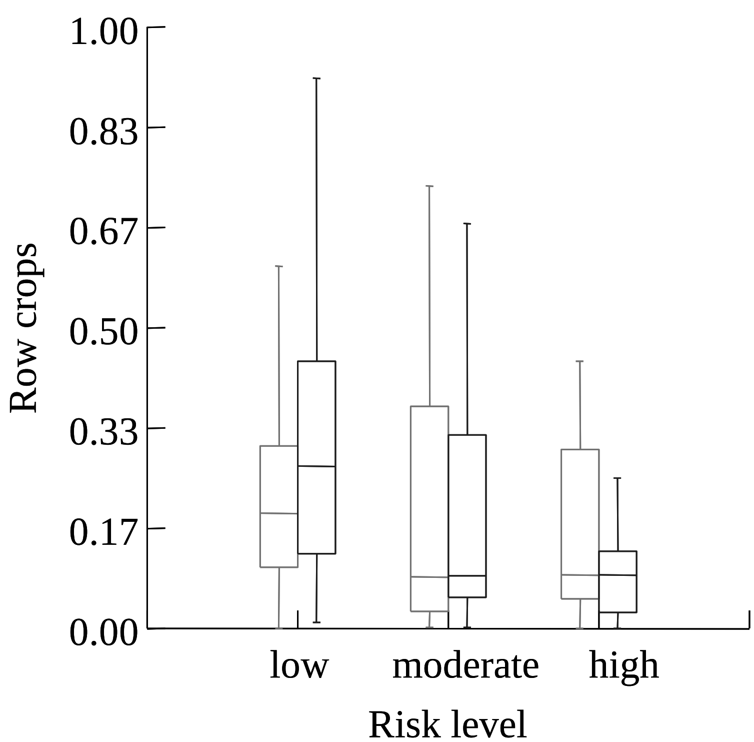
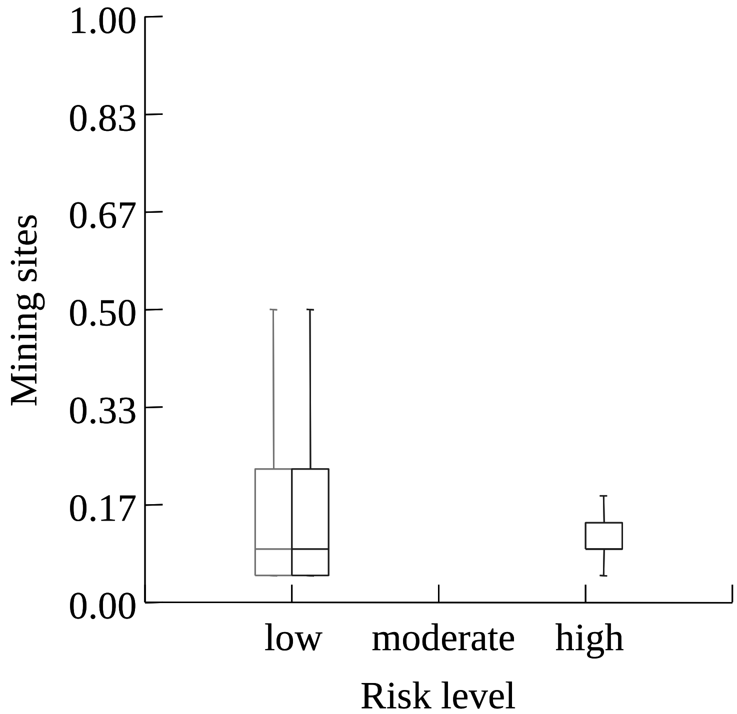

1.00
0.83
0.67
0.50
0.33
0.17
0.00
Row crops
low
moderate
high
Risk level

Figure 2.7. cont'd



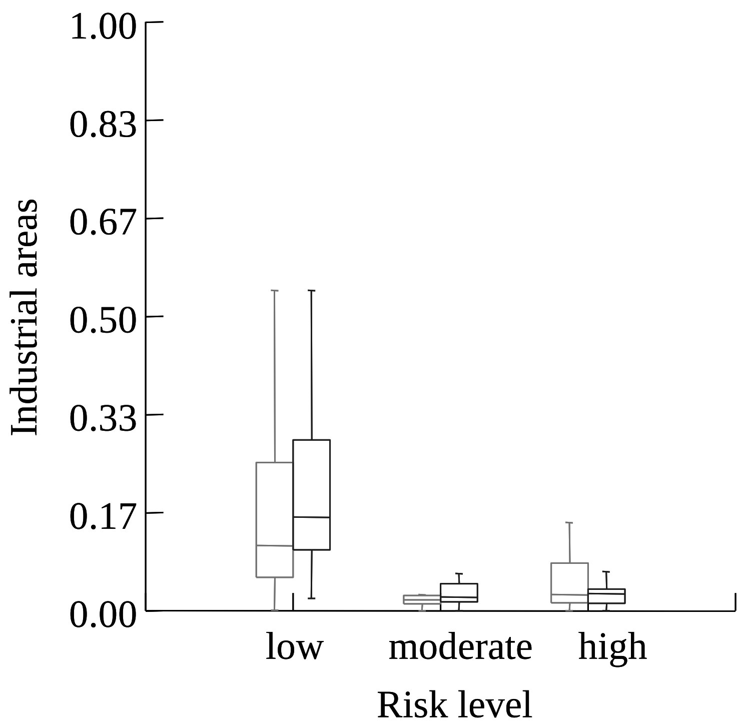
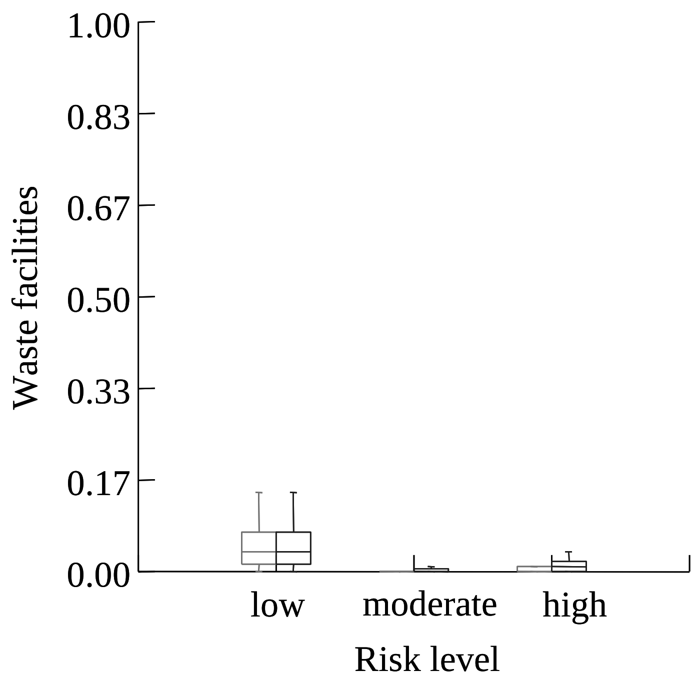


Figure 2.7. cont'd



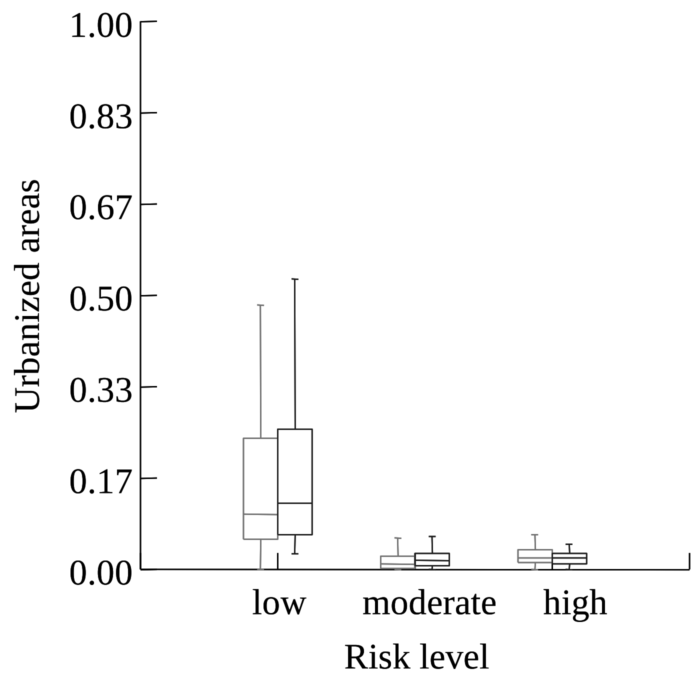
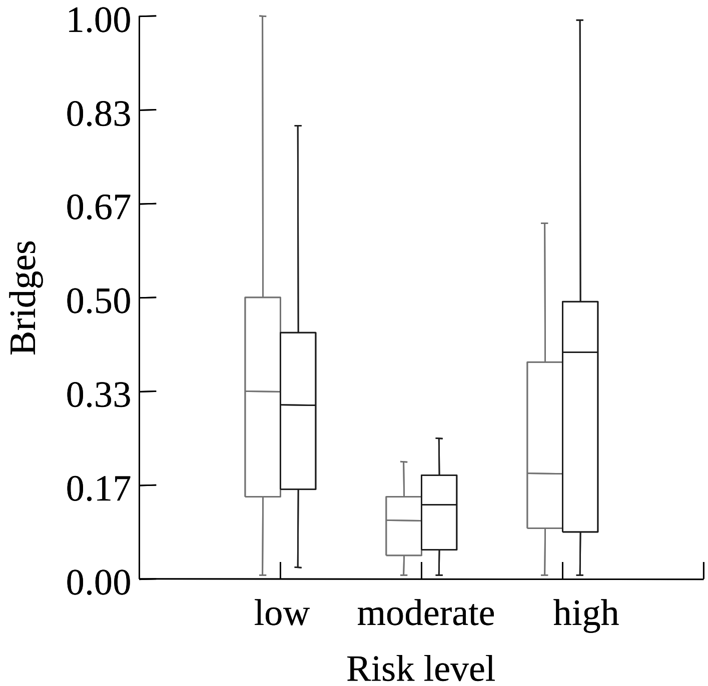


Figure 2.7. cont'd



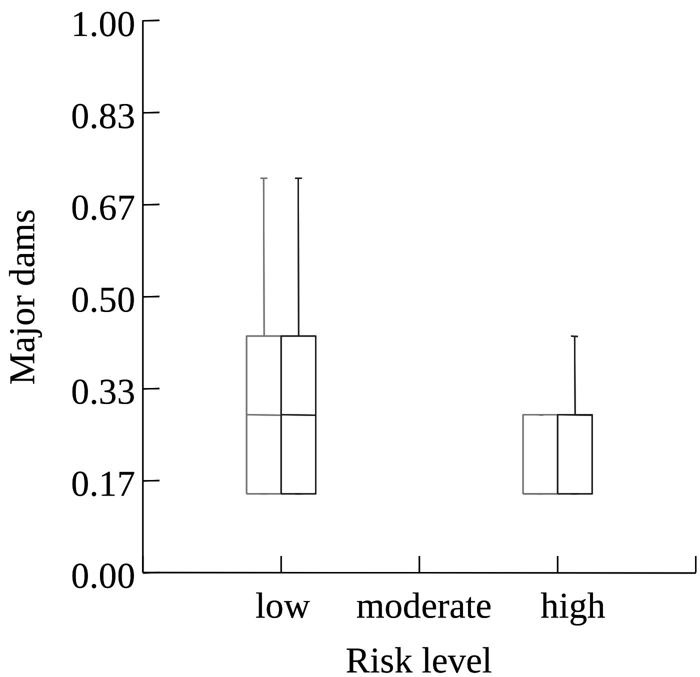
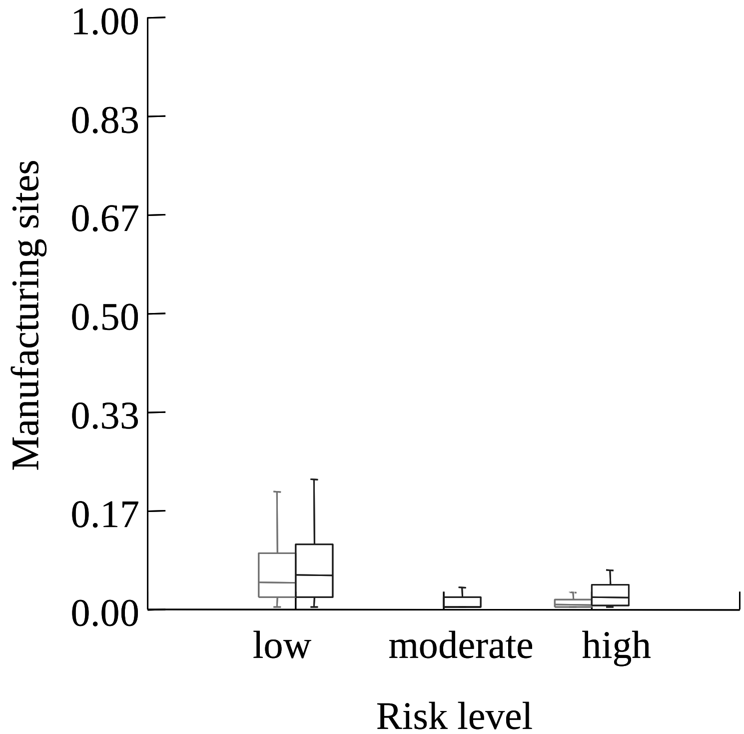
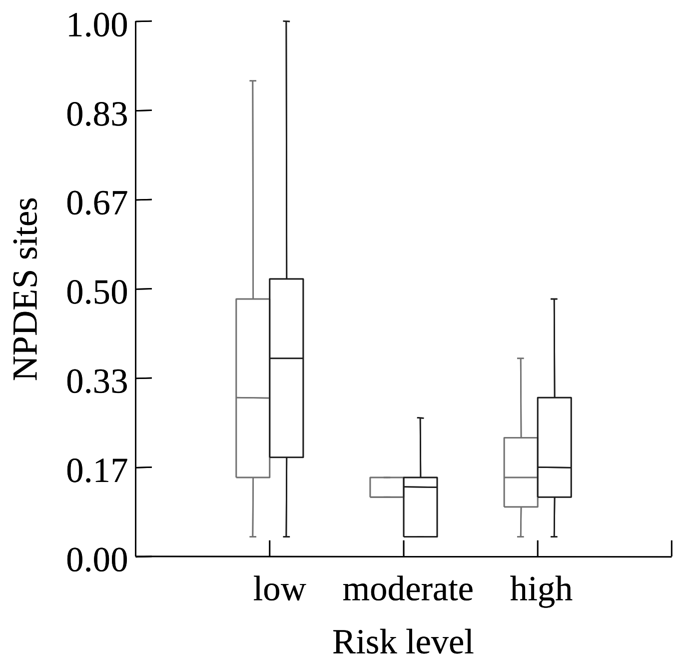


Figure 2.7. cont'd





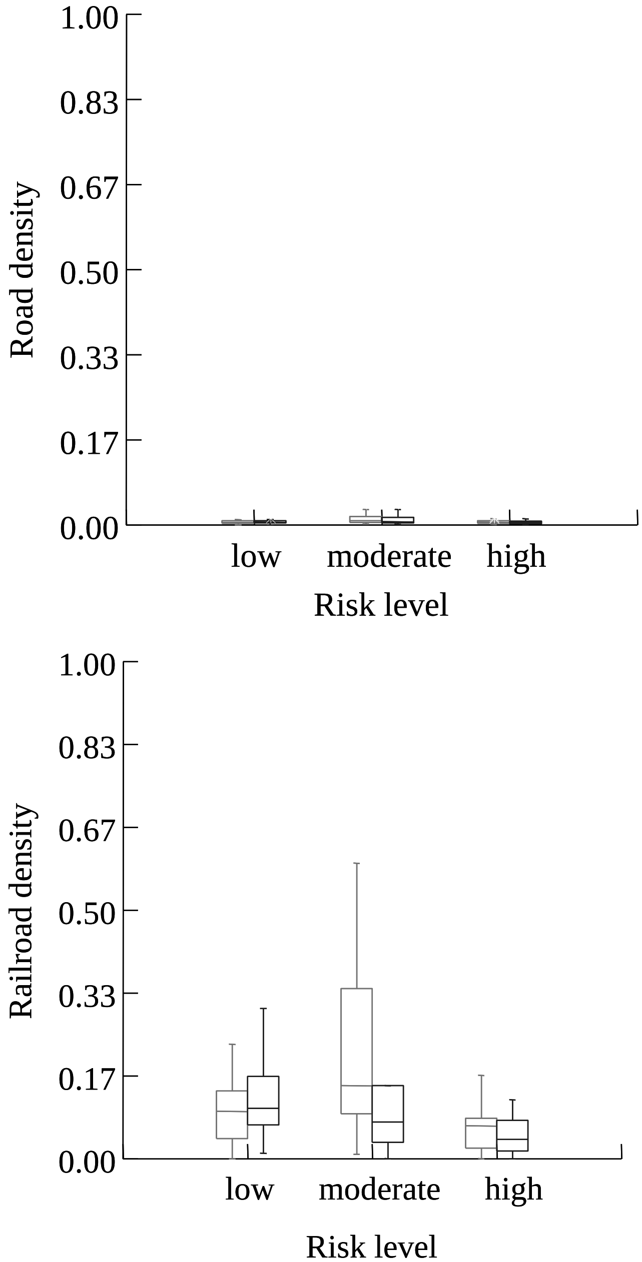


Figure 2.7. Boxplots from *K*-means cluster analysis of final risk rankings for twelve threat types in 107 catchments of the upper Tennessee River basin. Each graph shows one threat; the *x*-axis refers to the low-, moderate-, and high-risk levels for the original ERI (left panels) and quantile-based approach (right panels), and the *y*-axis depicts the standardized frequency scores for each individual threat. Notches indicate medians, and box lengths are confidence intervals (95^{th} quantile). NPDES stands for National Pollutant Discharge Elimination System.

Appendix 2.A. Ranges of scores used in Monte Carlo simulations to assess effects of using severity measures based on expert judgment in the Ecological Risk Index. A triangular distribution was used. The severity score was the apex, and the minimum and maximum scores were the low and high ends of the triangular distribution. The smallest and largest scores among the experts are the minimum and maximum values, respectively. Severity scores were published previously (Mattson and Angermeier 2007) and may range from 5 to 15. NPDES stands for National Pollutant Discharge Elimination System.

Major threat	Apex Severity score	Minimum Score	Maximum Score
Row crops	14	6	14
Pasturelands	11	5	13
Urbanized areas	14	8	15
Industrial areas	12	8	14
Mining sites	12	6	14
Waste facilities	12	6	14
Stream crossings	12	5	13
Impoundments	15	9	15
Manufacturing sites	11	5	12
NPDES sites	12	6	14
Railroad crossings	7	5	10
Road density	9	6	12

CHAPTER 3: A freshwater mollusk database to facilitate research and conservation: knowing the past to plan the future

ABSTRACT

Despite having the most diverse freshwater mussel fauna worldwide, the continued imperilment of freshwater mussels within the southeastern United States has prompted the need for a comprehensive review of their historical ranges and population numbers to facilitate conservation planning and restoration. One of the most diverse arrays of mussel species is found within the upper Tennessee River basin (UTRB), which is widely known for its varied but imperiled mussel fauna. Despite a broad interest in restoring freshwater mussels to their native streams, no regional central archive exists for tracking field surveys or recording conservation and management efforts. The collection of historical species distribution and abundance data within the UTRB prompted the construction of the Upper Tennessee River Mussel Database (UTRMD), from which researchers might gather pertinent information to construct conservation plans for imperiled mussel species throughout the basin. This chapter focuses on a discussion of the database, including its construction and content. Objectives of this chapter include: 1) collating historical freshwater mussel data from across the UTRB; 2) constructing a comprehensive freshwater mussel species database, and; 3) discussing issues pertaining to the constructon and use of a central archive such as the UTRMD. My goal was to construct a secure, comprehensive, online database to organize information on historical and current mussel species locations and conditions. The UTRMD contains >47,400 species records distributed across nearly 2,100 sampling sites within the UTRB. The database is pertinent to current research and accessible to researchers as well as other users interested in regional, national, and global biodiversity patterns. Through its mapping capabilities and accurate record of historical mussel

populations within the UTRB, the database also assists managers with reaching critical goals such as ecosystem restoration and informs management actions such as land-use planning.

INTRODUCTION

Freshwater ecosystems worldwide are threatened by numerous anthropogenic stressors, and measures to conserve ecological processes and aquatic communities are at the forefront of conservation planning. The conservation status of freshwater streams is a major concern among researchers and fisheries management agencies throughout the United States, and portions of freshwater ecoregions supporting major ecological processes that maintain target species have been identified as biologically important (Abell et al. 2000, Baron et al. 2002). Studies have focused on preventing or minimizing species and habitat loss with a goal of impeding a decline in the function and integrity of ecosystems, including changes in energy sources, water quality, habitat quality, flow regime, and biotic interactions, in order to develop a protection plan for aquatic biodiversity (Baron et al. 2002, Dudgeon et al. 2006). The cumulative impacts of anthropogenic stressors within freshwater ecosystems are difficult to measure, and long-term assessments of impacts to aquatic species are difficult to assess, so conservation planning should include both threat reduction and stream monitoring (Pringle 2001). Current, reliable information on the distribution and status of conservation targets is crucial to effective planning.

Although conservation measures are shifting focus from targeting specific aquatic species to conserving ecosystem processes (Geist 2010, Linke et al. 2010), there is much value in retaining species data for biodiversity assessments, monitoring conservation status, and as a means for identifying successful management strategies (Guisan et al. 2013). Species occurrence data are increasingly being linked to larger datasets, and much of these data are being used for conservation planning (Jetz et al. 2012, Margules and Pressey 2000). Freshwater mussels are ideal candidates for biomonitoring programs because they have historically been a large part of the biomass of river systems and play important ecological roles. Documented declines in mussel condition may be attributed to human actions that alter water chemistry, physical habitat, and fish host availability (Neves et al. 1997). Additionally, freshwater mussels may live for many decades, so the presence of a mussel bed indicates that a system that has retained its biological integrity on the time scale of decades and may be a suitable target for conservation planning.

The United States contains about a third of the known mussel species worldwide, but about 70% of its 297 species are considered threatened or endangered (Turgeon 1998). This imperilment is attributed to human activities, mainly land uses that alter the ecological conditions of the streams and rivers in which mussels subsist. Being sessile filter feeders, mussels improve water quality, but also depend upon flowing waters with minimal fine sediment (Strayer et al. 2004). They are considered indicator species because they are a food source to other species, and because they play a critical role in transferring energy and nutrients from the water column to the benthic community (Vaughn et al. 2008). This energy transformation is essential for the ecology and sustainability of other aquatic species (Vaughn and Hakenkamp 2001). Due to their unique characteristics and conservation status, watersheds containing freshwater mussels are considered priority areas, and conservation planning often includes monitoring mussel species recruitment, identifying potential restoration sites, as well as species augmentation, habitat expansion, and species reintroduction actions (Abell et al. 2000, VDGIF 2010).

I constructed a regional database, called the Upper Tennessee River Mussel Database (UTRMD), for freshwater mussel species located within watersheds of the southeastern United States. My work began within the upper Tennessee River basin (UTRB), which is considered one of the most diverse freshwater ecosystems in the United States (Abell et al. 2000).

Freshwater mussel species were historically numerous within the basin. Today, there are 60 recorded mussel species, and most of those are state-listed as species of conservation concern, along with 30 species listed as federally protected (Hampson et al. 2000, VDGIF 2010). This chapter is an account of the process of developing a database, a description of the database itself, and a discussion of the issues involved in making it pertinent to conservation planning.

Database questions

Mussel species occurrences had been recorded throughout the southeastern United States. for many decades, and historical data collections are an important resource for mapping species assemblages and identifying mussel beds with active recruitment. Since species assemblages have been altered by human activities throughout the region, understanding the consequences of acute and chronic impacts on species distributions supports restoration efforts by providing a historical perspective of population dynamics (Neves et. al.1997). Furthermore, a pilot study found that the number of field survey sites has declined throughout the region, so a record of past field survey sites may provide a template for additional survey locations. With this in mind, I collected mussel data from various state and local agencies with the purpose of mapping historical trends in species distributions. A product of this inquiry was the UTRMD, which includes extensive information about the condition, distribution, and recruitment status of freshwater mussels throughout the UTRB.

The UTRMD has several advantages over individual-based datasets that are commonly used in agencies and universities. First, most databases, such as those offered by natural heritage sites as well as state and local agencies, often record presence/absence of threatened and endangered species. My database includes this information and extends the scope of data coverage to other assemblages, including those species considered “currently stable” in conservation assessments. Second, the UTRMD is configured to accommodate data collected by various sampling methods, thus providing a comprehensive collection history. Third, it is imperative for researchers and managers who address endangered species conservation to have access to ecologically relevant data from all essential parties (Guralnick et al. 2007). Finally, the UTRMD interface is user-friendly, web-based, and has essential safeguards to prevent corruption and inappropriate access (i.e., passwords and security levels).

My goal was to construct a secure, online database that would provide information relating to historical and current mussel species locations and conditions. Objectives of this effort included: 1) collating historical freshwater mollusk data from across the UTRB; 2) constructing a comprehensive freshwater mussel species database, and 3) discussing issues pertaining to constructing and using a central archive such as the UTRMD. Ultimately, the database would be accessible to regional mussel researchers as well as users who may be interested in linking regional, national, and global biodiversity patterns (Darwall et al. 2008).

STUDY AREA

There is a high level of aquatic species diversity in the southeastern United States. The river systems in this area drain landscapes with diverse geologic, physiographic, and climatic elements, and the area is rich in history of the zoogeographic and evolutionary processes shaping aquatic fauna (Abell et al. 2000, Smith et al. 2002). High levels of endemism and a unique natural history followed by anthropogenic impacts have rendered the UTRB one of the most biologically threatened river systems in the nation (Neves and Angermeier 1990, Hampson et al. 2000, Diamond et al. 2002). There are 28 families of native fishes within the Tennessee system, with five specious families: minnows, suckers, catfishes, sunfishes, and perches (Abell et al.

2000). There are also many endemic species of freshwater mussels and crayfishes (Winston and Neves 1997).

The UTRB encompasses parts of four states: Virginia (8107 km^2), Tennessee (29785 km^2), North Carolina (14193 km^2), and Georgia (3315 km^2), and includes the entire drainage of the Tennessee River upstream of Chattanooga, TN (55 400 km^2) (USGS 2001). Three physiographic provinces, the Cumberland Plateau, Valley and Ridge, and Blue Ridge, are represented. The Cumberland Plateau is composed mainly of Pennsylvanian sandstone and conglomerate. Ordovician and Cambrian limestone, shale, and sandstone compose the Valley and Ridge province, and Pre-Cambrian and Paleozoic igneous and metamorphic rock comprise the Blue Ridge province. Elevations range from 189 meters above sea level at Chattanooga to 2037 meters at Mt. Mitchell, the highest point in the eastern United States. The terrain on the Tennessee-North Carolina border within the Great Smoky Mountains National Park is extremely rugged, with 16 peaks over 1829 m (6000 ft) and 55 km of crests exceeding 1524 m (5000 ft) in elevation.

Most of the basin is forested (65%), mainly contained within the Jefferson, Pisgah, Cherokee, Nantahala, and Chattahoochee national forests. Agricultural lands, located mostly within the Valley and Ridge province, make up about 25% of the land use. Pastureland accounts for most of the agricultural areas, whereas row crops represent less than 3% of the study area (Hampson et al. 2000). The remaining 10% includes impervious surfaces (6%), barren lands (mainly inactive and active mining facilities (3%), and open water (1%) (USGS 2001). The Tennessee Valley Authority (TVA) has constructed four mainstem impoundments (holding 3.8 billion m^3 of water), as well as 17 reservoirs on the tributaries (with a combined storage capacity of 12.3 billion m^3). There are also 17 private reservoirs within the basin (740 million m^3) (USGS 2001).

There are five major drainage systems throughout the UTRB (Figure 3.1), all with unique climatic, physiographic, and hydrologic conditions. The Clinch-Powell (11430 km^2) and Holston (9780 km^2) rivers flow through southwestern Virginia and form the northern portion of the basin. The French Broad (13271 km^2) and Little Tennessee (6804 km^2) rivers begin in the Smoky Mountains in North Carolina and flow west with large drops in elevation. The Hiwassee River (6993 km^2) joins the mainstem Tennessee River just east of Chattanooga after flowing down the mountains of northern Georgia, southwestern North Carolina, and southeastern Tennessee.

There are 102 mussel species found within the entire Tennessee River basin of Tennessee, Alabama, Georgia, North Carolina, and Virginia (Master et al. 1998, Price et al. 2014). The UTRB currently has 60 extant species, with >45 species in the upper Tennessee River tributaries of the Clinch, Powell, and Holston rivers (Diamond et al. 2002, VDGIF 2005). Based on similar-sized watersheds throughout the country, the Clinch-Powell river basin is ranked first for the greatest number of at-risk fish and mussel species (Master et al. 1998). Acute and chronic impacts from human actions within the Tennessee River basin have been associated with regional declines in biodiversity and biological integrity, including 15 of 174 fish species listed as federally endangered, 50 fish species having state-specific conservation concern, and over half of the 60 extant endemic mussel species receiving state and/or federal protection within the UTRB (Neves and Angermeier 1990, Hampson et al. 2000, Diamond and Serveiss 2001, Diamond et al. 2002).

METHODS

Database Structure

The UTRMD was intended as a central archive for field-collected mussel data. The database was constructed as an online source for data entry and access, and the only requirements for a user are an internet connection, a Web browser, and a user account login. It allows multiple users to access the system at the same time. The database was written in Visual Basic (,Net in Visual Studio 2003, Microsoft.NET 1.1 framework), and uses the Microsoft IIS6 web server (Conservation Management Institute, Virginia Tech). The UTRMD data are stored in a relational MySQL database comprising 20 tables that track over 90 data points for accuracy. The database consists of 55 variables summarized into four data input categories: citation sources, field survey data, sampling technique (quantitative/qualitative), and species information (Figure 3.1). Data input varies based on citation type and survey method (Table 3.1). The following is a description of the input categories, including details on data requirements.

Primary data sources such as a report, journal article, or book, are recorded to track primary resources (Figure 3.2). The database was formatted to compare new entries to previously entered citations to prevent duplicates in the system, and only primary peer-reviewed sources are allowed. This not only prevents duplicate records, but also ensures data quality by minimizing the use of secondary data such as natural history or distribution data. Citation entries require specific fields depending on citation type. For example, reports and journal entries require title, funding source (e.g. agency, university), and publisher information, while book citations require title and publisher. There is also an option to label a citation source as having sensitive species data, which restricts general access to the data associated with that citation.

Field survey data are input using a combination of drop-down menus and spatially referenced site coordinates. This data category summarizes the location and date(s) of the field survey, and requires specific sampling location coordinates. Data required include state, county, map quadrant, river basin, and river names, which are selected from drop-down menus. The user must input specific latitude/longitude and Universal Transverse Mercator (UTM) coordinates. The database requires proper combinations of UTM information and latitude/longitude coordinates prior to accepting data. Users may also enter site descriptions and other text into a comment field.

Mussel field surveys may be performed using either qualitative or quantitative sampling techniques (Strayer and Smith 2003). Both quantitative (number of individuals per species) and qualitative (species presence) surveys are common in mussel research, and the UTRMD accommodates both types of data sets. The database requires particular information depending upon survey method, and there are several data requirements for each method to assure data quality. Although species information is important, it is not a required field for either technique. Necessary data for qualitative surveys include survey method (e.g., visual scope, scuba, snorkel, or visual search) and catch per unit effort (CPUE). Similarly, records for quantitative surveys require CPUE, number of quadrants surveyed, and the number of passes within each quadrat for each survey.

Lastly, the database requires specific species data based on whether the field survey was qualitative or quantitative. Mussel occurrences are entered either as individuals or assemblages, and abundances must be quantified even for surveys in which no mussels were found. Species identities must also be input, even for fresh-dead and relic shells. More detailed data are required for quantitative surveys, including the number of individuals found per each species and individual shell lengths. Data from qualitative surveys may be entered as an assemblage, indicating species presence. Both sampling techniques require data on species condition (e.g., live animal, recently dead, or shell only). The database provides a drop-down menu for scientific names, and common names may also be entered.

Several functions within the UTRMD are reserved for authorized users only so that database integrity is maintained. Most importantly, field survey locations containing threatened and endangered species are protected within the database. Only authorized users may view or export these data so that species locations remain protected. There is also a hierarchy of authorization such that the system may restrict access to viewing, editing, and import/export functions based on a user’s authorization level. Lastly, a user must be authorized to perform import and export functions. These functions require spreadsheet-type formatting, and the export function allows the user to choose data via major data-type or individual field(s). These functions facilitate data import and export functions using an error detection system to ensure data quality.

A caveat to the construction of the UTRMD is that it also contains historical mollusk species records. Many of the older data records (pre-1980) contain mollusk occurrence data along with mussel survey data. Since mussel surveys no longer include extensive data on other mollusk species, these historic data provide a snapshot of mollusk occurrences that may not otherwise be noted and were included in the database. The UTRMD includes abundance data, habitat descriptions, and field collection techniques on 1,775 snail and clam species records.

RESULTS

The UTRMD was constructed to retain freshwater mollusk data from both historical records and current collections. The database contains information on >3000 sampling sites from nearly 100 studies and reports, and also includes data from smaller, institution-based databases. The results described here refer to collections from 1963-2008 to coincide with another project, and mussel surveys are still occurring throughout the basin. For my study years, there are >47,400 species records, with nearly 2,100 sampling sites within the UTRB. More specifically, the database contains 579 sampling sites and 32,928 species records within 49 12-digit hydrologic units within the Clinch-Powell river basin (Figure 3.3), a catchment known historically for its threatened and endangered freshwater mussels. Additionally, the UTRMD indicates that with some of the subwatersheds, known viable mussel populations have active recruitment. According to the database, the remaining subwatersheds with live mussel beds do not have obvious signs of juveniles or active reproduction.

The database contains life history data on 51 mussel species and two species complexes within the Clinch River basin (Appendix 3.A). There were more individuals found in recent surveys than previously, which may be related to the increase in the catch per unit effort since 2004. The most abundant species recorded in the UTRMD were *Actinonaias ligamentina, Actinonaias pectorosa, Epioblasma capsaeformis, Medionidus conradicus,* and *Villosa iris*. Alternatively, *Epioblasma florentina walkeri*, *Epioblasma haysiana*, *Lampsilis abrupta, Leptodea fragilis, Pleurobema cordatum, Toxolasma lividus,* and *Villosa fabalis* were recorded once, and only in the early study period. The combined species counts, along with mussel survey locations over time, suggest that some streams have thriving mussel populations. Therefore, either streams have had fewer surveys or their mussel populations have not been as well monitored.

DISCUSSION

The UTRMD began as an effort to develop and maintain a web-based interface of mussel data for use by universities, management agencies, and consultants involved in mussel research and management within the UTRB. An extensive amount of mussel species data from the past century was collected, and life history data were assessed for their usefulness to mussel conservation planning. The database is now a central archive of field-collected mussel data and was designed such that it can be extended to store data from regions other than the UTRB.

The creation of the UTRMD presented a few challenges. First, it was challenging simply to find archival data because most were not in an electronic format. For example, state agencies have conducted mussel surveys for decades, and most of these data are summarized in reports. Species data are spread among non-governmental organizations and government agencies, with no integrated accounting of field studies or restoration efforts, which supports the need for a central archive such as the UTRMD. The second challenge was determining whether a complete database is more valuable than having partial information relating to species assemblage accounts (i.e. specific species data collected instead of all species present at a site). Third, the database had to accommodate multiple field survey methods and pertinent data had to be input consistently. Fourth, even primary data sources did not have complete biological records (i.e., sex and age data), and the database had to indicate that information from each journal citation was entered completely. Lastly, databases are effective only if they are usable and used. Successful long-term database maintenance requires quality control guidelines, data transferability, and custodial responsibility (Costello 2009). These factors were addressed within the UTRMD by having a user identification system in which data access is controlled by system authorization.

In the process of completing the UTRMD, several issues such as data quality, database security, and propriety concerns were obstacles to retaining and recording data online. These issues were solved by having all input data verified by a second researcher, and by including specific access codes into the user security system to protect sensitive species data. I encountered problems similar to those reported by Costello (2009) with regard to motivating researchers to provide data and to assuring that the data would be used appropriately. I also encountered proprietary issues, so that author permissions were often required before use (Guralnick et al. 2007). As development of the database progressed, there were opportunities for improving data entry pathways and determining methods for applying modern spatial references to historical data records.

Preservation of data on environmentally sensitive species, such as freshwater mollusks, requires obtaining original data sources, which helps prevent the duplication of records and the exclusion of data missing from published reports. This effort generally necessitates cooperation among various local, state, and regional agencies and university researchers. Preserving the data and mapping species assemblage data over time provides valuable population information that is not possible to obtain otherwise (Balian et al. 2008), and changes our understanding of species distribution over time and space. Although researching primary data sources is time consuming, advantages include a complete count of all of the field surveys for species that may now be threatened, endangered, or even extinct. Such summaries provide a foundation useful for research and educational purposes (Whitlock 2011).

Although there are similarly derived freshwater mussel data available online, most of these sources focus on natural history data provided by state and federal agencies. Open access to publicly-funded research results is increasingly common. For example, commonly referenced data such as satellite imagery and other databases covering topics as diverse as land cover, impervious surfaces, watershed health, and natural history information are the results of data collected by agencies for specific purposes. Many of these data are open access, and have a primary data source that is not publicly available. While these online sources offer important information, there is a need to preserve historic data of an environmentally sensitive species group, which informs population managers of population demographics over time and space and facilitates conservation planning (Berman and Cerf 2013).

The online publication of species occurrence data may be accomplished using an approach that considers data quality, web-based security, and propriety issues at the forefront (Graf and Cummings 2014). Ecological studies would be enhanced if all species occurrence data were included in online databases so that ecological research, such as evaluating relationships between species distributions and land use disturbances, may be enhanced. Other advantages of an online database include updating endangered species information, collating species and individual viability assessments, promoting better mussel species propagation, and integrating various formats into a single secure usable data structure, thus allowing approved database users to download data to create maps, perform statistical analysis, and assess future conservation actions. For example, my research group is currently using the UTRMD to map species life history traits, including bed activity, recruitment status, and species turnover. These assessments have informed mussel conservation planners and identified research needs.

MANAGEMENT IMPLICATIONS

The notion of a central database for use in future research on mussels in the UTRB is appealing for planning purposes, and has the potential to streamline field collection (Appendix 3.B). Species data are currently collected in a similar manner by the numerous agencies and researchers involved in freshwater mussel research. There were indications while collating the UTRMD that some data collection was replicated unbeknownst to those collecting the data. Additionally, several parallel databases had been created unbeknownst to the researchers involved. With access to an online, central database such as the UTRMD, I expect more opportunities for collaborative conservation planning to arise, which would enhance opportunities for additional research and perhaps catalyze funding for more extensive conservation actions.

Give that protocols and locations of field surveys have varied over time, an analysis of how changes in survey methods (i.e., qualitative versus quantitative) and site locations have influenced population assessments may be a high priority for managers. For example, altering survey methods or harvest regulations have been shown to influence estimates of wildlife populations in subsequent years (Mattson and Moritz 2008). Although mussel surveys have been collected for many years, no comprehensive analysis of the sensitivity of population estimates to the availability of field survey data has been published. The purpose of such an analysis would not be to question conservation status, but to quantify the uncertainty associated with such assessments and to influence where and when future field surveys occur.

Some data collection programs, such as those using citizen scientists, are meant for open access with regard to data entry and use. This type of database, in which the general public provides environmental data and/or species occurrence data, has proven useful in supplying information leading to better monitoring of species assemblages, enhanced scientific understanding, and better strategies for conservation (Bonney et al. 2009, Dickinson et al. 2012). A similar viewpoint is shared by authors who recognize the need for broader access to species data to better fulfill professional duties and to enhance our understanding of biodiversity shifts

(Whitlock 2011). The UTRMD will serve a similar informational purpose for researchers, and as citizen scientists are recruited into formal mussel field surveys, these data may be included into databases such as the UTRMD.

The UTRMD could become accessible to national and global research and conservation communities (Darwall et al. 2008, Nobles and Zhang 2011), and information housed in the database may contribute to research and ongoing land use/restoration planning. I anticipate that the usefulness of a database such as the UTRMD is recognized as more than just a historical record of mussel bed activity, and that the research community recognizes that species databases are useful for identifying areas appropriate for restoration, and the availability of a central archive to house species data provides a foundation for focused and effective management in continuing conservation efforts.

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Table 3.1. Field codes and descriptors compiled in the upper Tennessee River Mussel Database (UTRMD). Field codes refer to the main data categories within the UTRMD, and individual variables associated with each category are listed below them. UTM stands for universal transverse mercator. CPUE is catch per unit effort.

Field Code	**Data Required**	**Input Method**
Citation data		
Type	Journal/Report/Book	User input
Author		User input
Publication Date		User input
Field Collection data		
Species Data	Scientific name	Drop-down menu
	Common name	Drop-down menu
Endangered species data		Yes/No
Mortality	Alive/Relic/Shell	Drop-down menu
Shell length (mm)		User input
Number of individuals		User input
Juvenile/Adult		Drop-down menu
Location Data		
State		Drop-down menu
County		Drop-down menu
Quadrant		Drop-down menu
Basin		Drop-down menu
River		Drop-down menu
Latitude/Longitude		User input
UTM coordinates		User input
Collection Method		
Qualitative	Survey type	Drop-down menu
	Scube, Scope, Snorkel	
	CPUE	User input
Quantitative	Quandrants	User input
	Number of passes	User input
	CPUE	User input

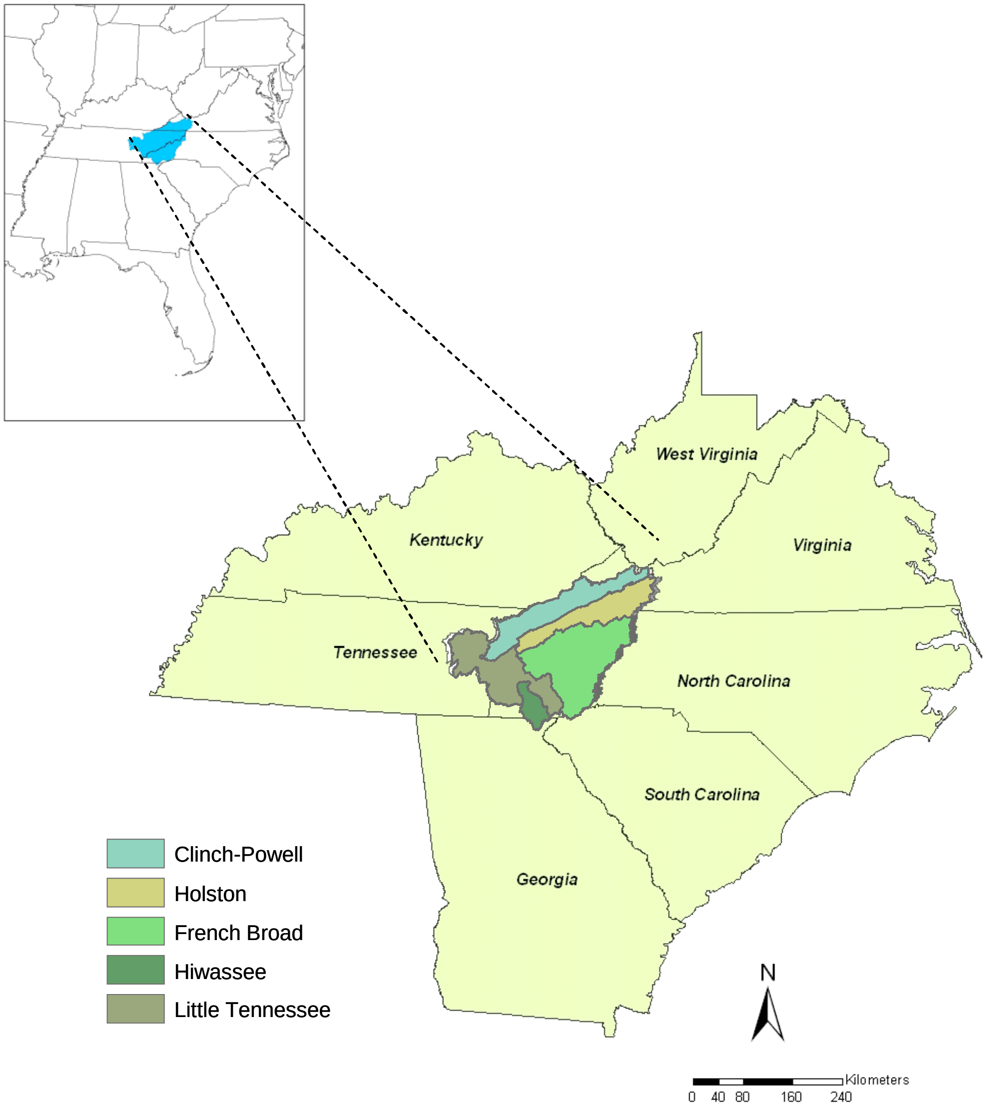

Figure 3.1. The five major watersheds of the upper Tennessee River system, USA.

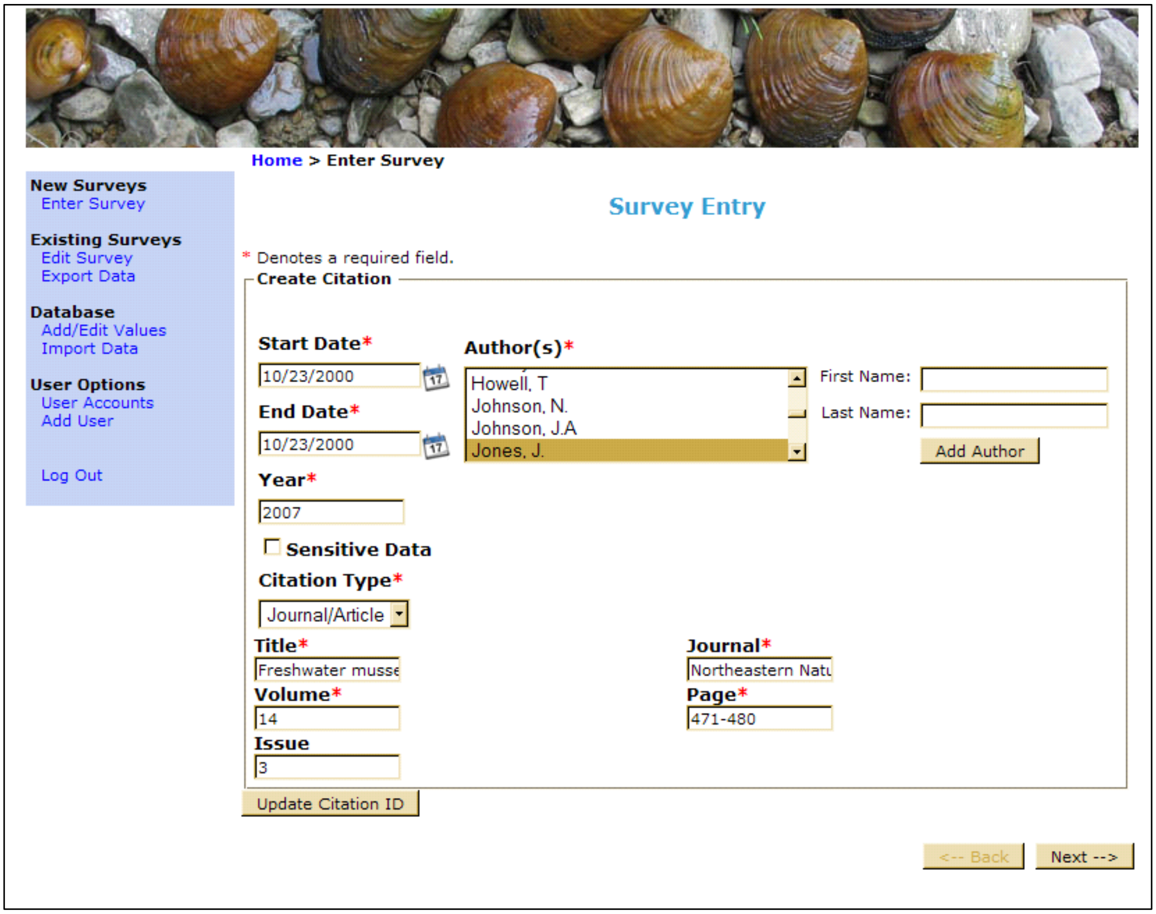

Figure 3.2. Initial screen of the Upper Tennessee River Mussel Database (UTRMD). This image shows the drop-down format for author(s), a format consistent throughout the input screens. The start and end dates of a field survey, publication year, and citation information are all required fields. The user may also enter a new author name using the “Add Author” user input tabs (First name, Last name). Also shown on the left-side menu are options for entering a new survey, editing a previously entered survey, importing/exporting data, and user information.

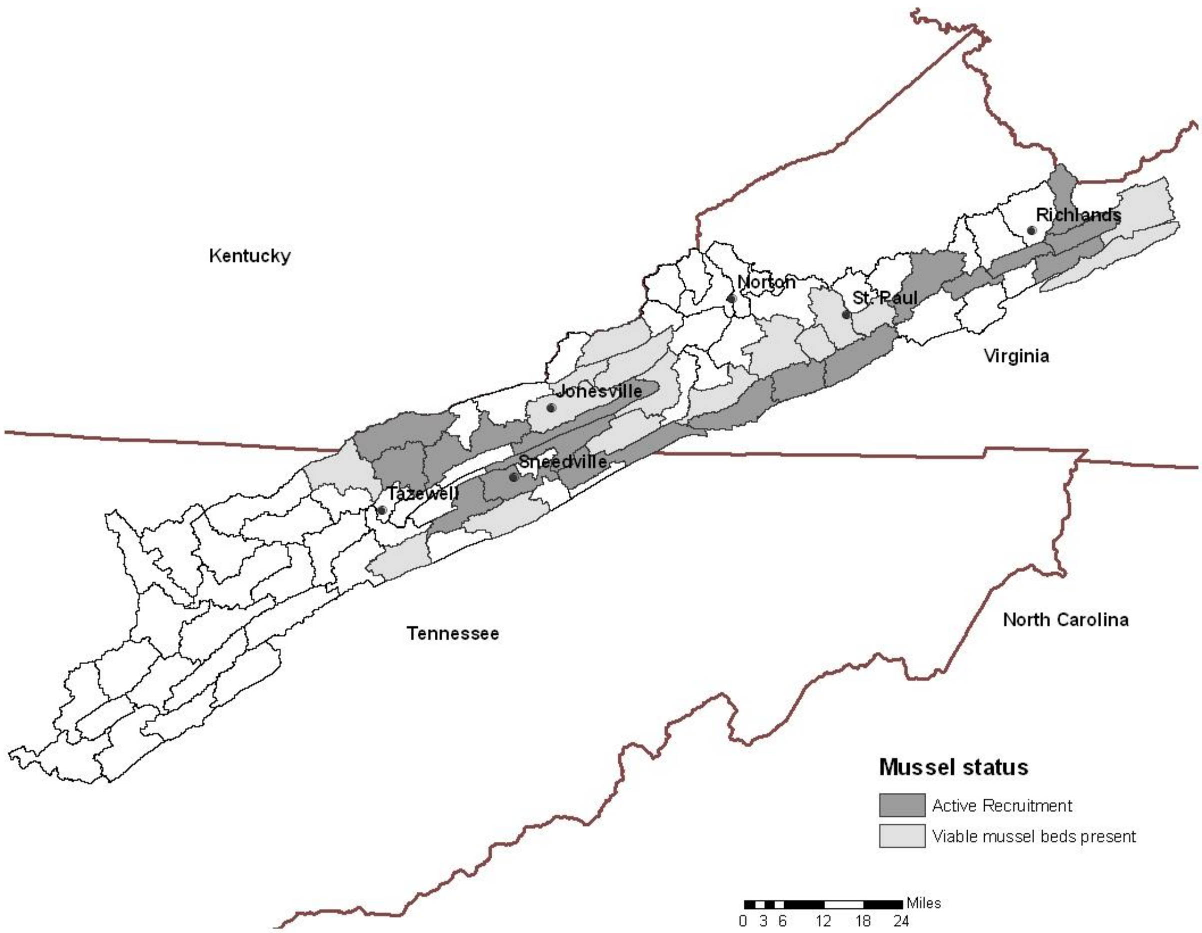

Figure 3.3. Freshwater mussel status within 87 12-digit hydrologic units across the Clinch-Powell River basin, USA. Those areas with active recruitment are also considered to have viable mussel beds, but the presence of mussel beds does not necessarily indicate active recruitment. Historical mussel surveys were collected from 1963-2008, and active recruitment refers to those mussel beds with juveniles present within 1998-2008. Field survey sites correspond to those hydrologic units containing mussel beds.

Appendix 3.A. Freshwater mussel counts for mussels found within the Clinch River basin, USA from 1963-2008. Blank spaces mean that no individuals were found for that particular species in a study period.

ScientificName	1963-1985	1986-1997	1998-2008
Actinonaias ligamentina	3805	874	2312
Actinonaias pectorosa	3564	2201	922
Alasmidonta marginata	90		2
Alasmidonta viridis	11		2
Amblema plicata	919	420	5
Cumberlandia monodonta	55	2	
Cyclonaias tuberculata	494	27	51
Cyprogenia stegaria	132		19
Dromus dromas	91	9	98
Elliptio crassidens	24	1	
Elliptio dilatata	1056	317	270
Epioblasma brevidens	198	14	96
Epioblasma capsaeformis	192	6	1075
Epioblasma florentina walkeri	6		
Epioblasma haysiana	5		
Epioblasma triquetra	142	1	21
Fusconaia barnesiana	650	772	434
Fusconaia barnesiana/Pleurobema oviforme Complex		1	
Fusconaia cor	243	4	8

Fusconaia cuneolus	601	18	21
Fusconaia subrotunda	1216	267	50
Fusconaia/Lexingtonia/ Pleurobema Complex	37		
Hemistena lata	72		35
Lampsilis abrupta	1		
Lampsilis fasciola	525	299	275
Lampsilis ovata	440	61	23
Lasmigona costata	2185	207	7
Lasmigona holstonia	3		188
Lemiox rimosus	38	3	21
Leptodea fragilis	147		
Lexingtonia dolabelloides	68	2	8
Ligumia recta	72	3	2
Medionidus conradicus	882	725	2517
Pegias fabula	3	3	1
Plethobasus cyphyus	117	6	9
Pleurobema cordatum	18		
Pleurobema oviforme	581	748	908
Pleurobema plenum	8		20
Pleurobema rubrum	20		1
Potamilus alatus	431	53	1
Ptychobranchus fasciolaris	425	284	242
Ptychobranchus subtentum	423	107	810

Quadrula cylindrica	247	11	12
Quadrula intermedia	34	13	47
Quadrula pustulosa	325	1	6
Quadrula sparsa	13	2	
Strophitus undulatus	34	1	3
Toxolasma lividus	1		
Truncilla truncata	178		1
Unidentified Unionid Juveniles	1		2
Villosa fabalis	3		
Villosa iris	2179	7656	7267
Villosa perpurpurea	89	1	23
Villosa vanuxemensis	356	5	22
Total	23448	15125	17790

Appendix 3.B. Sample of the agencies and organizations contacted, description of available data, and issues related to individual datasets that were identified as eligible for import into the Upper Tennessee River Mussel Database.

Dataset	**Description**	**Availability**	**Issues**
Virginia Department of Game and Inland Fisheries (VDGIF)	Based on permits issued	spatially referenced data	Some data sensitive
Virginia Tech surveys for the Virginia Department of Transportation	Surveys in proximity to bridge and road construction	Open access	Limited survey area
Virginia Natural Heritage Database	Comprehensive (presence/absence (P/A)) database used for regulation	Restricted access	Some data sensitive
Virginia Department of Conservation & Recreation	Comprehensive (presence/absence (P/A)) database used for regulation	Restricted access	Some data sensitive
North Carolina Wildlife Resources Commission	Detailed dataset provided by state malacologist	Open access	Formatting
Tennessee Wildlife Resources Agency	unknown format	Restricted access	Little detail
Published literature	Survey information	Open access	Time consuming
Environmental Consulting Groups	Survey information	Open access	Little detail
Tennessee Valley Authority (TVA)	Comprehensive Qualitative (presence/absence) database	Open access	Qualitative data only
U. S. Fish and Wildlife Service	Comprehensive (presence/absence) data	Data for parts of UTRB	Qualitative data only
Jones and Neves (2008)	Published account of North Fork Holston River	Open access	Limited survey areas
Petty et al. (2007)	Dataset from Copper Creek Study	Open access	Limited survey area

Tennessee Valley Authority (TVA)	Comprehensive Qualitative (presence/absence) database	Open access	Qualitative data only
U.S. Fish and Wildlife Service	Comprehensive (presence/absence) data	Data for parts of UTRB	Qualitative data only
Jones and Neves (2007)	Published account of North Fork Holston River	Open access	Limited survey areas
Clinch-Powell River basin dataset	25-year quantitative samples	Reports and datasheets	Limited survey areas
TVA Cumberland Mussel Conservation Program	Historical surveys conducted within the Tennessee River basin	Reports	Limited survey areas
Robert Dillon Snail Database	Snail database maintained by College of Charleston	Open access	Historical surveys only
Henley (1996)	Site-specific study within Holston River	Reports	Limited survey areas
Winston and Neves (1997)	Surveys of smaller upper Tennessee River basin streams	Journal	Limited survey areas
Church (1991)	Survey of the Little River, Major tributary of the Upper Clinch	Report	Limited survey areas
Wolcott and Neves (1994)	Surveys in the Powell River	Journal	Limited survey areas

CHAPTER 4: Streams, spatial scale, and human impacts: how data availability impedes freshwater mussel conservation

ABSTRACT

The influences of modern society on stream conditions and habitat quality are concerns to freshwater mussel conservation. The goal of this study was to evaluate how historical land use patterns, a major driver of habitat quality, relate to decades of freshwater mussel distributions within the Clinch River basin, USA, a basin historically containing about 10% of the world's freshwater mussel species. The basin has had extensive mining activity over the past 150 years, but is now mostly forested (61%), along with agricultural (16.7%), developed (9.7%), grassland (9.4%), wetlands/open water (2.3%), barren (0.7%), and shrub/scrub (0.3%) land covers. Study objectives were to: 1) describe historical land cover patterns at three spatial grains: 12-digit hydrologic units (HUC12s), riparian buffers within each HUC12, and riparian buffers within 2 km of mussel collection sites; 2) relate spatiotemporal patterns in freshwater mussel species via species turnover, recruitment changes, and current population status to historic patterns in riparian land cover; 3) evaluate the historical role of human activities on species assemblages within existing mussel beds, and 4) discuss the conservation value of decades of freshwater mussel data.

Mussel presence was clustered throughout the basin, with species sampling occurring in 40 of the 86 HUC12s from 1963-2008. An increase in quantitative collections since 2001 has resulted in a 60% increase in the number of juveniles identified since 1963, despite the overall mussel population declining by >75% over the past 50 years. Additionally, shifts in sampling technique over time made it difficult to determine mussel population status. Minor land cover changes have occurred basin-wide, with increases in barren lands (0.4%) and grasslands (0.4%)

in areas with continual and intense land use. Forested areas decreased by 1.2% basin-wide, while urban areas, mostly within the southern portion of the basin, increased by <0.3%. The majority of watersheds (90%) have experienced <5% absolute change in their riparian land cover. Streams with the most riparian disturbances since 1980 have had continued major disturbances, such as mining, and mussel surveys have ceased. HUC12s with mussel data from the early and late study periods show significant increases in urban (t = 3.944, P = 0.003) and agricultural (t = 3.227, P = 0.010) land cover, but a decline in riparian forest (t = -2.127, *d.f* = 9, P = 0.062).

Results of analysis of variance tests suggested that the current adult mussel abundance has been impacted by changes in land use (F = 9.752, *d.f.* = 9, P = 0.096), while juvenile abundance has not (F = 6.063, *d.f.* = 9, P = 0.149). The results suggest that streams with repeated mussel surveys are supporting increasing populations, including active recruitment in several beds. An increase in quantitative surveys since 2004 most likely had provided more accurate species and population counts, although actual population sizes are still uncertain. Land use within a 2-km radius of repeated quantitative surveys has been stable over time, even though mussel abundances have declined. Furthermore, variation in riparian land cover has been minimal over time and space, supporting the hypothesis that land use has not been a major contributor to population fluctuations. Overall, the study suggests that land cover change over the past 30 years does not explain the observed declines in freshwater mussel populations, but the effects of land conversion may need to be monitored. Quantitative surveys should continue to be conducted throughout the basin to provide more accurate information about mussel distribution and abundance, which can inform conservation strategies.

INTRODUCTION

Freshwater mussels (Mollusca: Unionoida: Unionidae) are an important part of the freshwater fauna in eastern North America, where they have a center of biodiversity, but where their numbers have also declined in recent decades (Williams et al. 1993). The degree to which freshwater mussels are impacted by modern society depends on the duration, extent, and intensity of human activities within a watershed (Haag and Williams 2013). It is well established that events such as toxic spills (Sheehan et al. 1989) and riparian disturbances (Diamond et al. 2002) directly impact mussel recruitment and species persistence, but the impacts of land cover changes on established mussel beds have not been thoroughly examined. This study focused on the relationship between historical land use and decades of freshwater mussel species distributions to determine whether land cover patterns over time and space are related to mussel population status. Freshwater mussel species are good candidates for studying the effects of land use changes on stream integrity because many of these species are long-lived and have lifestage-related sensitivities to water quality and physical habitat conditions (Neves et al. 1997).

Human activities impact freshwater ecosystems at multiple spatial scales, and these impacts may be evaluated across different spatial (e.g., riparian versus watershed) and temporal (e.g., recent versus historical) frames. Watersheds are typically characterized by their degree of human dominance when land cover data are considered, with forested riparian areas and undeveloped floodplains ranking as least impacted (Omernik et al. 1981). Additionally, human-dominated areas with impervious surfaces and industrial uses are commonly considered as having poor conservation value (Wang et al. 2001). Freshwater mussels are especially impacted by agricultural practices within riparian zones because of the potential for sedimentation and pesticide leaching (Arbuckle and Downing 2002). Human activities within riparian zones are specifically cited as influencing mussel bed productivity. Anthropogenic stressors that disrupt the physical habitat or water turbidity such as pasturelands, industrial sites, and poorly managed forest practices are especially harmful (Box and Mossa 1992, Diamond and Serveiss 2001, Neves et al. 1997, Strayer 2006). Stream reaches with successfully reproducing mussel beds typically have fewer major stressors within 2 km upstream than their failing counterparts (Diamond and Serveiss 2001), but the influence of human-dominated stressors, whether stable or dynamic, on long-term mussel viability is uncertain (Vaughn 1997). Since land use is such an important factor to freshwater mussel conservation (Hopkins 2008), the role of land use change on recruitment status and species distribution is an open question.

Despite best efforts in stream monitoring and assessment, and water quality restoration, it is unknown why freshwater mussel populations are declining nationwide (Regnier et al. 2009). Freshwater mussels are especially stressed by land and water uses that alter local habitat, water chemistry, or sediment loads (McRae and Allan 2004). Chronic threats such as industrial operations, coal mining and processing, impoundments, and agriculture have been implicated regarding reduced reproductive success and juvenile mortality. Acute threats, such as chemical spills and mine spillage, contribute to mussel absences from previously inhabited sites (Diamond et al. 2002). Both chronic and acute stressors influence individual mussel beds. Although researchers have quantified species declines associated with local pollutants and activities such as toxic chemical spills (Naimo 1995), studies relating regional and local influences to freshwater mussel status are unresolved (Allan 2004, Utz et al. 2009).

Land use and land cover (LULC) data are generally accepted indicators of disturbance within a landscape (Freeman et al. 2003), and may also be used to explore the complex relationship of historical land use and modern impacts within streams (Allan 2004). The value of

LULC data in assessing freshwater mussel populations is still being explored (Hopkins and Whiles 2010). Besides the obvious observation that landscapes with fewer acute threats support stronger mussel populations, the relationship(s) between historical land use, local impacts, and mussel populations are unclear (Arbuckle and Downing 2002, Soucek et al. 2003, Hopkins and Whiles 2010). My goal was to determine whether observed historical changes in spatiotemporal land cover patterns are linked to decades of mussel species distribution and recruitment status.

Factors contributing to mussel absence from a site previously known to support them include both short-term stochastic events (natural and anthropogenic disturbances) and chronic conditions, such as impacts stemming from land uses (Haag and Williams 2013). Depending on the frequency and severity of pollution events, impacted stream reaches may still be considered for mussel bed restoration (Sheehan et al. 1989). For example, there may be short-term acute stressors in some areas, such as spills from a bridge, that might impact immediate mussel recruitment, but recruitment eventually recovers. Other areas, in contrast, might feature chronic conditions that reduce mussel survival/recruitment, promote fish-host loss, or continuously expose biota to toxicants, thereby impairing mussel bed viability. Under such conditions, adult mussels may be viable, but not reproductively successful (Strayer et al. 2012). Ultimately, restoration efforts would include managing current sites, and identifying additional stream sites with less risk potential (Haag and Williams 2013).

A key goal of freshwater mussel conservation planning is to restore and maintain the physical, chemical, and biological properties of a watershed that facilitate regular recruitment. Juvenile mussels are more sensitive to environmental factors than adults, so maintaining natural properties, functions, and interactions is likely to retain key ecological functions and biodiversity (Dudgeon et al. 2006, Strayer 2006) over multiple levels of biological organization (Angermeier and Schlosser 1995, Neves et al. 1997, Vaughn 1997). The historical impacts of local and regional land uses on freshwater mussels must be understood for conservation planning to move forward to protect species assemblages and retain biological integrity (Diamond et al. 2002). Both local and regional land uses can influence aquatic macroinvertebrate populations (Allan and Johnson 1997, Lammert and Allan 1999); the ways in which aquatic systems react to land cover changes depend upon stream size, biotic community present at the time of an impact, severity of land cover changes, resiliency of the system, and permanence of human activities within a watershed (Allan 2004, Hynes 1975).

Against this background, I took a multi-scale spatial approach to study the relationship between historical land cover and decades of mussel distributions within a watershed containing a long history of mussel surveys. The Clinch River basin, USA, contains about 10% of the world's freshwater mussel species, and has one of the most diverse mussel assemblages. Unfortunately, more than 30% of the mussel species are imperiled (Williams et al. 1993), and, among similar-sized watersheds in the United States, the Clinch River is ranked first for the greatest number of at-risk fish and mussel species (VDGIF 2010). Mussel population surveys have shown a continual decline in recruitment for many species, and human activities have altered water quality and reshaped the landscape (Neves and Angermeier 1990, Hampson et al. 2000, Diamond et al. 2002). Hence, my study objectives included: 1) describing historical land cover patterns within watersheds of the Clinch River basin; 2) relating spatiotemporal patterns in current freshwater mussel distributions to patterns in riparian land cover; 3) evaluating the impacts of historical human activities on decades of species assemblages within existing mussel beds, and 4) discussing the conservation value of decades of freshwater mussel data.

My goal was to evaluate long-term trends in freshwater mussel distributions on the basis of historical land use changes as delineated by hydrologic units (as defined by the U.S. Geological Survey) in a freshwater ecosystem for which I have over 100 years of mussel survey data. Not only is an analysis of historical land cover and mussel data long overdue, but understanding how land use changes have impacted mussel recruitment and persistence will inform conservation efforts, such as identifying stream segments with the greatest potential for juvenile success and, subsequently, successful species reintroductions. Lastly, an evaluation of nearly 50 years of mussel data may provide an overview of data quality and purpose, and aid in improving designs of future mussel surveys.

METHODS

Study Area

The Clinch River basin is located within the Appalachian Mountains of the southeastern United States in an area unglaciated in recent geologic time (Figure 4.1). The basin drains both the Clinch (7,542-km^2) and Powell (2,429 km^2) river systems, and forms the northern-most portion of the upper Tennessee River basin (UTRB). The region is characterized by a mountainous Ridge-and-Valley terrain with a high percentage of karst features. It has a temperate climate, with an average rainfall of 114 cm per year (Hampson et al. 2000).

The Clinch River basin has historically contained one of the most diverse aquatic faunas in the United States, and is considered a freshwater conservation priority area (Abell et al. 2000). Of the 1,000 freshwater mussel species identified worldwide, nearly 125 species have been found within the greater UTRB watershed (Neves et al. 1997). The Clinch basin historically had 79 mussel species, and currently contains about 52 mussel species, including 28 that are federally protected and 38 with state protection (Hampson et al. 2000). According to The Nature

Conservancy (Smith et al. 2002), primary conservation targets in the Clinch River valley include the lower-river, transitional, and headwater mussel assemblages.

Major threats to aquatic resources in the Clinch basin include toxic spills, sedimentation from agricultural and logging practices, pesticide toxicity, nutrient inputs from agriculture, and contaminants associated with coal mining (Abell et al. 2000, Smith et al. 2002). The Clinch basin contains two major dams, Melton Hill and Norris, but is the only section of the UTRB with a large portion of its waters remaining free-flowing. There are polluted, impaired stream segments throughout the middle and upper reaches of the Clinch River basin (VA DEQ 2012). Human activities such as agriculture and timber harvest have contributed to these impairments, but more important are chemical contamination and mining (Hampson et al. 2000).

Freshwater mussel species data

Freshwater mussel data for the Clinch River basin were obtained from mussel surveys dating back to the late 1800s and collated into a mollusk database (the Upper Tennessee River Mollusk Database ((UTRMD), see Chapter 3). The UTRMD archives field survey data collected using either quantitative (visual and tactile surveys for obtaining site-specific mussel density) or qualitative (visual survey focused on mussel richness and presence) methods. The database is the most comprehensive compilation of species presence over time and space within the UTRB. Data entries require spatial references, and the database automatically checks for duplicate records. There are >32,000 species records at >550 sites within the Clinch River basin spanning five decades. I collated and mapped mussel species records (n = 10,498) located within the Clinch River basin by year (1963-2008), life-stage (adult or juvenile), and recruitment status (juvenile presence versus absence). A caveat to recruitment status is that translocation of glochidia from laboratory to stream may have been occurring in the latter years of the study.

Unless noted in the field report, these translocated juveniles would be counted as being part of an actively recruiting mussel bed. However, since there was no indication of translocation in the database citations, all juveniles were identified as recruiting from stream-based mussel beds. Species data were refined and grouped by period based on land use analysis (described in next section): 1963-1984 (n = 3,731) and 1998-2008 (n = 6,307), respectively. Results of mussel surveys (n = 460) between the two time periods of interest (1985-1997) were removed from analysis so that field-survey data corresponded to land cover time periods, and multiple mussel generations were more likely to be represented. HUC12s (n = 10) containing collection sites in both time periods were retained for further analysis.

Land cover change

Land cover patterns were analyzed using LULC data (USGS 1990, Kearns et al. 2005) from the early 1980s to 2001 and National Land Cover Data (NLCD) from 1992 and 2001 (Homer et al. 2007). All data were converted on a pixel-by-pixel basis ($30m^2$) into one of eight land uses: open water, developed (both low and high), barren (including active and inactive mining sites), forest, shrub/scrublands, grasslands, pasture/cultivated, and wetlands (Anderson et al. 1976, Wickham et al. 2013)). LULC (1980-1984, 1990-1992, 2001-2004) and NLCD (1992 and 2001) data were summarized by decade, and a series of decision-support rules and classification and regression tree (CART) models were used to characterize temporal changes in land cover. Producer's accuracy rates were consistent between datasets (Wickham et al. 2013).

The basin was divided into 86 HUC12s and land use changes were mapped to identify areas with more or less change by decade. A pilot study found that land use between the middle (1990s) and early (1980s) years and between the middle (1990s) and late (2001) years did not change significantly, so further analysis included land use for the 1980s and 2001. These time periods also likely represent different mussel generations since mussels are long-lived. Due to the minimal changes in pixel values, I did not estimate changes in basic landscape metrics such as fragmentation and contagion (O'Neill 1988). Percentage change in land use in each HUC12 during 1980-2001 was computed as the difference between 2001 and 1980 land cover types per pixel, divided by the total number of pixels.

Two additional spatial grains were analyzed for patterns of land use change. First, since freshwater mussels are sensitive to local disturbances (Diamond et al. 2002), I applied a 100-m riparian buffer on both sides of streams basin-wide. Streamside buffers were applied to 3rd -5th Strahler stream orders, which are those most likely to contain mussel beds within the Clinch River basin (VDGIF 2010). Land cover within riparian buffers was summarized for each HUC12. Lastly, I considered the potential impacts of local land uses on mussel bed condition. I identified ten mussel beds (via collection site data) surveyed in both study periods, and defined a 2-km riparian buffer around each bed (Diamond and Serveiss 2001). Then, land use changes between periods within those buffers were analyzed with a paired t-test (n = 10).

Impacts of land use on mussel populations

I evaluated the relationship between land use and the presence of mussel recruits by examining land cover near known mussel beds from the early and late study periods. I identified ten HUC12s with mussel data for early and late study periods, and then summarized their land uses within a 2-km riparian buffer around specific field-survey collection sites. Mussel data were collated, including species name, life stage (juvenile or adult), and number counted at site. Mussel bed status was assessed by examining changes in mussel counts, number of species present, and presence/absence of juveniles (i.e., recruitment). Geomorphological features were assumed constant. One-way analysis of variance (ANOVA) tests were used to determine whether land cover changes accounted for mussel abundance change between the early and late time frames.

RESULTS

Mussel distribution and abundance

Historical collection effort for mussels and the apparent distributions of the respective species varied greatly across space and time in the Clinch River basin. Mussels were collected in 40 of the 86 HUC12s during 1963-2008. Mussel presence was clustered throughout the basin, and 30 HUC12s contained mussels in either the early (1963-1985) or late (1998-2008) time periods; 10 HUC12s containing mussels were surveyed in both study periods (Table 4.1). Field surveys shifted from qualitative surveys in the early time period that encompassed most of the Clinch River basin, including the Powell River, to more spatially limited, quantitative surveys that focused on Copper Creek, but also included portions of the lower and upper Clinch River. Ten fewer mussel species were identified in the late (n = 41) versus early (n = 51) time periods, but none of these were exclusive to streams surveyed only in the early time period, such as the Powell River. However, the number of mussels counted per HUC12 increased in the later time period, likely a result of more intensive survey effort within a geographically narrow zone.

Specifically, there were collections from 24 HUC12s in the early, pre-1985 time period, with 23,448 mussels counted, including two HUC12s containing >3,200 individual mussels. In the post-1998 time period, 13 HUC12s had 17,790 mussels counted. Distribution of mussels was uneven across the HUC12s, and included one site in Wise County with a single specimen (*Villosa iris*), and two sites having >3000 mussels each. Copper Creek, sampled in both time periods, exhibited a 3-fold increase in the number of mussels sampled, with >5100 mussels counted in the late time period. The number of mussel collection sites per HUC12 ranged from 1 to 58 (Figure 4.1), indicating clumped collections over the study period. This outcome was expected because known mussel beds are often revisited to note bed condition, while new mussel beds are rare.

Those streams with greater mussel catch-per-unit-effort historically did not necessarily have similar collection efforts in more recent years, so absolute counts of mussels are not comparable (Figure 4.2). Qualitative samples were collected almost exclusively over the years until 2004, when the ratio of quantitative to qualitative samples increased nearly 3-fold (Figure 4.3). For example, field surveys since 1998 have been more focused, encompassing 11 fewer HUC12s, but more than doubling the number of collection sites relative to historical surveys. Also, the increase in quantitative collections since 2001 has resulted in a 60% increase in the number of juveniles identified since 1998 (Figure 4.4), despite overall mussel counts being 24% lower than those observed 50 years ago, declining from 23,448 to 17,790.

Aggregate counts suggest that mussel abundance was highest in the early years, dropped to its lowest in the 1990s, and is somewhat greater today (Figure 4.5). These demographic trends are further supported by the numbers of juveniles found throughout the basin today compared to the early time period. Fewer than 20 juveniles were collected prior to 2004, whereas 596 juveniles have since been counted. Although this trend may be a result of the shift to quantitative sampling and translocation programs that release laboratory-raised glochidia, the inference that recruitment is ongoing in several sites throughout the basin suggests that physical habitat conditions are suitable, and at least some mussel species may be demographically viable.

Since both the number of sites visited and the collection methods applied at each site were not consistent throughout the study period, no unbiased index of mussel relative abundance or catch per unit effort can be computed; however, there were a few noteworthy abundance trends. Of the 54 species found throughout the Clinch River basin in the 1960s, counts for 35 species have declined, counts for 8 species have increased, 9 species had but single-year detections, and counts for two species exhibited no trend. Within those HUC12s sampled in both the early and late study periods, there were more individuals and more species found since 1998 than prior to 1985 overall, but counts were mixed for individual HUC12s (Appendix 4.A). Those species currently assessed as demographically stable, such as *Villosa iris* (counts in 8 out of 10 HUC12s), were observed consistently in both study periods. Conversely, imperiled species, such as *Pleurobema oviforme* and *Amblicata pectorosa*, had smaller spatial distributions with very few individuals observed. Most imperiled species counted in the late study period had juveniles present. Additionally, there were 24 species in the early and five species in the late study periods labeled as "unique," defined as species found only within that study period (Table 4.1). Additional surveys are required to determine the current status of these species.

The recent increase in quantitative sampling has offered new insight into mussel recruitment throughout the basin. For example, more species were counted in the late study period using quantitative versus qualitative sampling techniques (Table 4.1). Of those species with active recruitment, seven are classified as critically imperiled, five are of special concern, and 11 have secure populations (NatureServe 2013). Six species exhibit greater juvenile counts today than in previous periods, suggesting either that habitat conditions favor recruitment or that earlier qualitative sampling had overlooked juveniles (Table 4.1, Appendix 4.A).

Land cover change

Land cover in the Clinch River basin has changed little during the study period. Today, the basin is mostly forested (61%, including riparian areas), with other land covers including agriculture (16.7%), developed (9.7%), grassland (9.4%), wetlands/open water (2.3%), barren

(0.7%), and shrub/scrub (0.3%). A paired-sample *t*-test showed no change in HUC12 land cover basin-wide (t=0.0, df=7, p=1.0) between the early and late study periods. The largest shifts involved barren lands (+0.4%), grasslands (+0.4%), and forested areas (-1.2%) basin-wide, indicating that forested land is being converted to other uses. The increases in barren and grassland land covers are located in HUC12s with previous mining activities, although my study did not verify ongoing mining activity.

Apparent changes in land cover varied with the spatial grain used for quantitative assessment. Land cover within HUC12s changed more than land cover in larger hydrologic units (Figure 4.1), indicating that land cover changes were local, and that no basin-wide patterns were detected. Several HUC12s had >15% of their land converted, but most experienced <5% converted (Figure 4.6). The most dynamic HUC12s, located within the upper Clinch basin, have changed continually over several decades, and contain both inactive and active mining sites along with companion land uses such as deforestation (Hampson et al. 2000). These areas historically have not contained viable mussel beds and, according to my mollusk database, were last surveyed prior to 1963. Although not a formal analysis within my study, the current land uses within these HUC12s indicate that there is little potential habitat for mussels, and that field surveys would, therefore, not be recommended.

Riparian land use trends paralleled those for entire HUC12s. Most watersheds (90%) experienced <5% change within their riparian areas during the study period. HUC12s with the most riparian change have experienced continual disturbance, such as mining, with few mussel species present historically. In fact, mussels have not been collected within these HUC12s for at least 50 years. In contrast, significant increases in development (t = 3.944, P = 0.003) and agriculture (t = 3.227, P = 0.010) have occurred within riparian buffers of HUC12s with repeated mussel collections. Some riparian forest has been lost (t = -2.127, P = 0.062) over the past 30 years (Table 4.2). These are ecologically sensitive areas for mussel conservation (Diamond et al. 2002), so increasing landscape disturbance poses impacts to mussel bed viability. Although land cover changed in riparian areas of some HUC12s, land cover within 2 km of repeatedly collected mussel beds did not change significantly (Wilcoxon signed rank tests, P >0.10 for all land uses). These results suggest, along with trends in mussel counts, that the impacts associated with land cover change near actively recruiting mussel beds are not detectable using the landscape-level methods employed in my study.

Impacts of land cover change on mussels

Land cover was relatively stable over time throughout the Clinch River basin and in HUC12s with mussel observations, suggesting that land cover change is not directly linked with mussel decline at the landscape level. The basin has remained mostly forested for the past 30 years, including critical riparian zones, with the exception of hydrologic units historically and currently associated with mining. Mining is generally limited to the western edge of the basin, and, if there were mussels present there in the past, my database indicates that they were extirpated before 1900. All mussel observations in both time periods of my study were >2km downstream of permanent land conversions, suggesting that land conversion in the most disturbed areas of the basin has itself had little impact on mussel counts (Diamond and Serveiss 2001). Acute pollution events associated with mining and industry, such as toxic spills (e.g., coal and slurry spills from electric plants), would negatively impact mussel beds downstream of stream crossings or industrial sites, but such events were not captured by the methods of my study.

Mussel counts within those HUC12s sampled in both the early and late years appear to be increasing over time (Table 4.3). In contrast to basin-wide abundance trends (see above), results for HUC12s sampled in both study periods suggest that numbers of adults and recruits are increasing (Table 4.4). However, these patterns are confounded with recent shifts in collection methods. As reported previously (Kovalak et al. 1986, Miller and Payne 1993), I expected to see increases in both species numbers and juvenile numbers as methods became more quantitative, and this expectation was consistent with observed abundances (Table 4.3).

Results of past research have suggested that threats within 2 km of a mussel bed have the largest impact on mussel population stability and recruitment (Diamond and Serveiss 2001), so I evaluated the relationship between abundance and land use for those beds with both early and late collection sites. Neither current adult (ANOVA: $F = 9.752$, d.f.= 9, $P = 0.096$) nor juvenile ($F = 6.063$, d.f. = 9, $P = 0.149$) mussel abundances were strongly impacted by changes in land use. Instead, total counts in beds indicated that mussels are responding positively to their environment today (Figure 4.4). This result may be confounded by changes in sampling design between study periods, but mussels seem to be responding positively to minimal recent shifts in land cover. Other factors also support this finding. First, riparian land uses within these areas showed but minimal changes over time and space, supporting the hypothesis that land use has not been a major contributor to population fluctuations. Second, regional land cover outside the 2-km buffer zones has been stable, so that impact from upstream disturbances is likely minimal. Lastly, methodologies have changed such that more intense, quantitative collections in recent years have most likely provided more accurate species and abundance counts than ealier collections (Miller and Payne 1993).

DISCUSSION

Mussel distribution and abundance

Data from field collections indicated that mussel populations have declined over the past 50 years throughout the Clinch River basin, despite increased field surveys and improved recent recruitment. Field surveys have not been spatially consistent over the past 30 years, but in those areas with repeated mussel surveys, counts of both adults and juveniles have increased. For example, there was a 24% decline in total numbers of mussels observed since 1963 basin-wide; however, counts have increased since reaching a record low in the 1990s. This dynamic suggests that either habitat conditions have improved enough that more juveniles are surviving to adulthood, or increased sampling intensity has artificially increased observed recruitment. Hence, although mussel counts are below those of 50 years ago, recent field surveys suggest that the assemblages are rebounding in more recent times.

Observed mussel patterns and data quality

Sampling technique influenced how mussel demographic trends were evaluated, as both qualitative and quantitative surveys were used at various times and places. Consistent with Miller and Payne (1993), qualitative surveys within those HUC12s containing early and late surveys identified greater species diversity and richness historically than did more recent quantitative surveys in those same areas (Table 4.1). Conversely, quantitative sampling revealed much more recruitment, although more recent surveys did produce observations of a number of species previously unseen locally. For instance, there has been an increase in the number of juveniles observed over time, and there are more species known to occur today than 50 years ago. On the other hand, there are fewer adults today, and some species have not been identified in areas where they once were found. These results may simply reflect inconsistent sampling efforts, suggesting that more focused sampling designs and widespread use of standardized protocols are needed to accurately characterize mussel population patterns.

Since mussels have patchy distributions even where they are plentiful, both survey techniques have their utility. Even when both survey types are used, determining mussel population size and assemblage composition may be difficult (Miller and Payne 1993, Obermeyer 1998). Inconsistent sampling techniques, as used over the course of my study period, present a challenge for evaluating how fluctuations in counts are related to local and regional abiotic and biotic conditions. For example, the impacts related to an acute stressor such as a spill is easily determined, but impacts associated with chronic stressors may take long-term, intensive site-specific research. However, the fact that collection methods are inconsistent across time, space, and collectors does not negate the importance of evaluating results of qualitative and quantitative surveys together. For example, my database has been useful in identifying key overall patterns in freshwater mussel abundance within the Clinch River basin. A more consistent sampling design is recommended, such as using quantitative surveys in beds where recruitment is uncertain, would help distinguish beds with active recruitment from those that merely hold adults. Additionally, specific long-term study objectives such as determining recruitment over time within a specific bed or observing species assemblages within a stream provides a context for purposely choosing appropriate survey methods (Hornbeck and Deneka 1996, Strayer et al. 1997, Vaughn et al. 1997). The use of both sampling approaches may support assessment of potential conservation actions and direct biologists toward potential restoration sites.

Influences of spatial grain on observed land cover patterns

Although land cover within the Clinch River basin has not remained static, changes have been few within those HUC12s and riparian areas containing mussel beds. The majority of land conversions took place within HUC12s containing well-established industrial or urban areas in which mussels had been extirpated or never occurred. There have been no published mussel surveys for these streams for >50 years. Consequently, there is a spatial mismatch between land cover changes and mussel-bed hotspots.

Land conversions were minimal throughout the Clinch River basin, with regional land cover changes occurring within HUC12s that contained no notable mussel populations. Even with internal errors in producing land cover maps (Wickham et al. 2013), the land cover patterns remained similar as the spatial grain increased. Results of the land cover analysis suggest that those portions of the Clinch River basin in which mussel sampling regularly occurs have not had significant land cover changes within the past 50 years. For example, forested areas, which are important for mussel habitat stability (Diamond and Serveiss 2001), have remained stable along riparian buffers. Instead, other impacts from stochastic events, such as toxic spills or sediment loading events may better explain recent population declines.

Spatial grain was important when evaluating changes in land cover. Land conversions were minimal throughout the basin, as well as for individual HUC12s. The amount of forest cover did decline in riparian zones within HUC12s, which may compromise future mussel conservation efforts by restricting the number of potential restoration sites (VDGIF 2010). This finding is in contrast to that for the smallest spatial grain assessed, which were buffered areas around mussel beds with repeated mussel surveys, in which land cover, including forests, was not altered significantly. My results suggest that riparian areas basin-wide are especially vulnerable to land conversion, but that such impacts are presently distant from mussel beds with active recruitment.

Impacts of land and water use on mussels

Most of the North American freshwater mussel species have been impacted by anthropogenic stressors over the past century (Box and Mossa 1999). Human activities have negatively impacted much of the Clinch River basin, especially in the upper reaches where industrial operations and mining are active (Sheehan et al. 1989, Diamond et al. 2002). Agriculture is more common in the middle reaches of the basin, where most mussel recruitment occurs. Although most of the basin's riparian areas remain forested, forest cover has declined the most when compared to other land cover types. My analysis concurs with Price et al. (2014) that adult mussels are present throughout the basin, but in smaller numbers in the Clinch River basin than historically. The UTRMD also indicates that there is little recruitment within most of the recently surveyed sites. This result has puzzled researchers because no obvious biotic or abiotic factors appears to be causing the decline (Jones et al. 2014).

Some stream reaches with seemingly healthy adult populations are located in disturbed or chronically polluted areas. These reaches have little or no recruitment, and hold demographically senescing mussel communities. Adult mussels tend to be more tolerant of stressors that produce heavy metals, increase sediment, and alter water conditions (Neves and Angermeier 1990, Hampson et al. 2000). Such impacts often harm juveniles without permanently reducing adult mussel populations (Strayer and Malcom 2012). While land cover has remained relatively stable within those reaches having recruitment, without adequate protection from chronic and acute stressors, recruitment may potentially decline even in these reaches, posing declines in mussel population viability.

Those streams containing mussel beds with active recruitment have had few land conversions over time, so land cover change did not explain mussel patterns at this spatial grain. Riparian zones surrounding mussel beds within the Clinch River basin are mostly forested, which supports channel structure and minimizes erosion (Lammert and Allan 1999). This maintains biotic condition and provides, along with a sub-critical number of stochastic pollution events, opportunity for active recruitment (Hopkins and Whiles 2010). Some riparian zones within HUC12s did experience significant land use changes, but many of these areas have had no mussel surveys within the past decade. Thus, if the basin retains its current mix of land cover, additional recruitment is plausible. A survey of potential land cover changes within riparian zones may be appropriate so that those beds with active recruitment may be adequately protected. Continuing anthropogenic stressors occurring upstream may potentially compromise mussel conservation efforts. Similar conclusions have been put forth by Lammert and Allan (1999), Roth et al. (1996), and Stewart et al. (2005), among others, who have suggested that land use throughout a catchment impacts aquatic communities.

Conservation implications

Results of my study attest to the importance of analyzing long-term datasets in the context of setting conservation priorities. I analyzed the impacts of land cover change on mussel community diversity and abundance within the Clinch River basin, a mussel conservation hotspot. Analysis of field survey data suggests that mussel species abundances and recruitment have increased within the Clinch River basin over the past 20 years, but the number of sampling sites has declined, and changes in sampling techniques may be skewing results. Sampling is more focused than in the past, which has provided a useful assessment of selected streams within the basin, but it has also limited the information available about other mussel beds.

Consequently, mussel population trends are difficult to assess, and it is possible that some of the perceived population declines simply reflect lack of unified sampling design such as stratified random sampling.

The mussel database contained collections that were mainly qualitative. This precluded any rigorous analysis of population status, or even of determining actual species absence. It would be helpful to have consistency in field survey data so that population trends could be determined. This objective may come to realization if quantitative surveys are continued in future collections. Sites identified by the database as having active mussel recruitment have also had few land cover changes, so surrounding stream reaches may be surveyed as potential restoration sites (VDGIF 2010). Alternatively, increases in active recruitment noted in the database may be a reflection of translocation efforts of laboratory-raised glochidia to mussel beds with healthy adult populations. Unless noted during a field survey, these mussel beds would presume to have active recruitment. Currently, there is no way to determine from the database which scenario is accurate. Additionally, other locations with adult mussels may need further field surveys focusing on fish host viability, risks of pollution events, substrate suitability, and potential land conversions prior to being deemed appropriate for conservation actions.

The use of long-term datasets is not novel, and there is much to be learned from long-term data when considering management actions (Dodd et al. 2012, Mattson and Moritz 2008). In my study, the mussel data were collected under various funding sources and for a multitude of reasons; however, it was possible to provide an overview of mussel trends. One caveat is that future population assessments should use consistent methods and take into account historic surveys. Managers would benefit from directing field surveys to focus on specific objectives, such as total population numbers or recruitment, and to identify an appropriate spatial grain for further field surveys. The importance of using good science cannot be overstated, especially when data analysis depends on results of application of consistent, well-planned methods.

The conservation status of freshwater species is often linked to the degree to which humans influence the surrounding landscape (Jackson et al. 2004). Freshwater mussel species are at risk globally (Regnier et al. 2009), and human activities have long been implicated as major causes of decline (Williams et al. 1993). Many of the land uses within the Clinch River basin associated with increases in nutrient loads or sediments, water temperature fluctuations, and agriculture remain (Saunders et al. 2002), and may make some mussel populations vulnerable to extirpation (Jackson et al. 2004). Alternatively, few changes in land cover since the early 1980s, along with documented increases in mussel abundance, indicate that portions of the basin have thriving populations. However, forested land has declined within riparian zones, and further land conversions may compromise restoration and recovery potential (Hopkins and Whiles 2010, VDGIF 2010).

Pollution events may ultimately impact healthy mussel populations if additional riparian areas are disturbed or acute pollution events occur upstream, so management of the entire watershed upstream from mussel beds is often recommended (Saunders et al. 2002, Utz et al. 2009). Protecting upstream resources is important, because bed reestablishment of beds is unpredictable. For example, sections of the Powell River that were previously well-populated were decimated by mining pollution in the early 20th century, and individuals translocated to previously populated beds have failed to thrive (Sheehan et al. 1989). Such events, including mining spills and bridge-related accidents, are likely to recur, so locating potential mussel habitats is an important research opportunity.

Mussel conservation efforts require an understanding of how mussels are distributed within a watershed, including the role of human impacts on species longevity, reproduction, and recruitment, and hence, distribution. The conservation status of freshwater aquatic species is often linked to the degree to which humans influence the surrounding area (Jackson et al. 2004). Some land uses contribute higher sediment loads to nearby streams than others, but there are few studies linking long-term sediment-loading to mussel health (Arbuckle and Downing 2002, Box and Mossa 1999). This type of study often includes relating landscape changes to population status over space and time. The situation was much more complicated for freshwater mussels. Consistent with past research suggesting that human impacts are most threatening to aquatic species when they occur closer to physical habitats (Allan 2004, Diamond and Serveiss 2001), my results suggest that neither basin-wide nor riparian-zone land cover changes explained mussel declines. Other long-lasting demographic impacts of infrequent toxic events may be most important.

Conservation of freshwater mussels is a high priority within the Clinch River basin. Much of the basin is undeveloped, which has provided opportunities for restoration. Although mussel abundances appear to be increasing, it is paramount to continue conducting both quantitative and qualitative surveys basin-wide so that population trends may be rigorously assessed. Additionally, I suggest protecting natural land cover within riparian zones upstream of mussel beds, which will provide a natural buffer for land use changes, although mussel beds would be vulnerable to stochastic pollution events such as toxic spills (Atkinson et al. 2012). Additionally, streambank preservation retains key habitats for aquatic communities, protects stream integrity, and ultimately retains a healthy ecosystem for protecting threatened species (Angermeier and Bailey 1992, Dudgeon et al. 2006). The situation that land cover has remained stable within the basin is uncommon when compared to other watersheds throughout the United States (Drummond and Loveland 2010), so protecting its unique qualities is a conservation priority.

CONCLUSIONS AND RECOMMENDATIONS

The spatiotemporal relationship between land use and freshwater mussel distributions of multiple spatial grains and a mollusk database spanning five decades was evaluated. My research was originally meant to predict future ecological risk to freshwater mussel distributions within the Clinch-Powell River basin, but land cover changes were minimal over the past 40 years, which obfuscated risk projections. Furthermore, human land and water uses have been consistent throughout the basin. Despite these minimal changes, mussel populations have fluctuated both stream-wide and within individual beds. Variations in sampling techniques may have influenced population estimates, but there likely have been environmental or land and water use influences.

Researchers should be aware of the general types of information obtained from quantitative and qualitative sampling methods and how these data are most appropriately used in studying population dynamics. Delineating long-term goals for bed-specific and stream-wide research should provide context for adopting consistent site-specific sampling methods. My study addressed the relationship between changes in land use and mussel assemblages. There may be value in broadening the research scope to relating stream characteristics such as flow, temperature, and chemical content to spatiotemporal changes in recruitment and species assemblages.

Lastly, there is little doubt that freshwater mussel populations have declined over the past 40 years while land uses have remained relatively stable. With this in mind, mussel conservation plans would benefit from continuing long-term field surveys in those mussel beds that have repeated quantitative sampling. It is recommended that additional environmental surveys within 2-km of those sites, such as stream characteristics, water chemistry, temperature, and flow regime, as well as biotic surveys including fish host and predator presence, in order to obtain a better understanding of how nearby environmental conditions may influence mussel bed composition.

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Table 4.1. HUC12s containing freshwater mussel collection sites in both early (1963-1985) and late (1998-2008) time periods, within the Clinch River, USA. “Unique” refers to those mussel species found in only one study period. A distance of 2km from repeated collection sites was used for analyzing land cover data.

	Number of Sites		Number of species		Number of Unique Species		Sites within 2km		Number of adults		Number of juveniles		Number of quantitative samples	
HUC12	Early	Late	Early	Late	Early	Late	Early	Late	Early	Late	Early	Late	Early	Late
060102050101	2	17	3	5	0	2	2	4	3	36	0	3	0	0
060102050102	9	23	8	7	2	2	9	20	26	95	0	2	0	0
060102050401	20	1	29	8	21	0	3	1	245	75	0	1	0	0
060102050403	6	3	28	14	15	1	1	2	41	296	0	8	0	41
060102050701	5	11	4	7	0	3	5	8	17	42	0	0	0	0
060102050702	8	9	6	5	2	1	8	9	30	32	0	0	0	0
060102050703	23	34	30	27	3	0	23	34	31	205	0	0	0	0
060102050803	29	3	39	26	14	1	10	3	501	1471	0	145	46	1616
060102050808	20	8	40	32	9	1	20	8	326	2705	0	345	0	3050
060102051002	14	1	32	22	12	2	4	1	149	761	0	48	0	809

Table 4.2. Statistics of comparing land cover between 1980 and 2001 within riparian areas of ten HUC12s containing freshwater mussel collection sites. Standard deviations and means of land coverages are shown.

Landcover Type	Paired Differences Mean	Standard Deviation	Standard Error Mean	95% Confidence Lower	Upper	t	Degrees of freedom	P-Value (2-tailed)
Open water	0.027	0.183	0.058	-0.104	0.158	0.465	9	0.653
Developed	0.316	0.253	0.080	0.135	0.497	3.944	9	0.003
Barren/ Open Rock	0.690	1.602	0.507	-0.457	1.836	1.361	9	0.207
Forests	-1.842	2.738	0.866	-3.801	0.117	-2.127	9	0.062
Shrubland/ Orchards	0.007	0.030	0.010	-0.015	0.028	0.711	9	0.495
Agriculture	0.800	0.784	0.248	0.239	1.361	3.227	9	0.010
Wetlands	0.003	0.028	0.009	-0.017	0.023	0.350	9	0.735

Table 4.3. Wilcoxon signed rank test comparing mean mussel counts between early (1963-1984) and late (1998-2008) study periods for riparian areas within 2 km of freshwater mussel collection sites.

	Number of individuals			
	Early	Late	*Z*-statistic	*P*-Value (2-tailed)
Adults	1369	5718	-2.293	.022
Juveniles	0	552	-2.366	.018
Total counts	1369	6270	-2.293	.022

Table 4.4. Mussel species with active recruitment within the Clinch River basin since 2004. Data are from quantitative and qualitative surveys throughout the basin. Conservation status (G1 = critically imperiled, G2 = imperiled, G3 = vulnerable, G4 = apparently secure, G5 = secure) was obtained from the NatureServe database (http://www.natureserve.org/explorer/index.htm). Species in bold font have greater counts today than in the early (1963-1984) sampling period.

Scientific Name	Observed Number of Adults	Observed Number of Juveniles	HUC12s	Conservation Status
Actinonaias ligamentina	567	11	33, 38, 43	G5
Actinonaias pectorosa	891	30	33, 38, 43	G4
Cyclonaias tuberculata	50	1	33	G5
Cyprogenia stegaria	14	5	33, 38, 43	G1
Dromus dromas	94	4	43	G1
Elliptio dilatata	221	10	33, 38	G5
Epioblasma brevidens	76	20	33, 38, 43	G1
Epioblasma capsaeformis	752	323	33, 38, 43	G1
Epioblasma triquetra	17	4		G3
Fusconaia cor	7	1	38, 43	G1
Fusconaia cuneolus	19	2	33, 38	G1
Fusconaia subrotunda	49	1	38	G3
Lampsilis fasciola	103	6	33, 38, 43	G5
Lampsilis ovata	22	1	38	G5
Lemiox rimosus	18	3	33, 38	G1
Medionidus conradicus	1386	77	16, 33, 38, 43	G3, G4
Pleurobema oviforme	78	1	2	G2, G3
Pleurobema plenum	18	2	38	G1
Ptychobranchus fasciolaris	233	7	33, 38, 43	G4, G5
Ptychobranchus subtentum	704	31	33, 38, 43	G2
Quadrula pustulosa	4	2	38	G5
Truncilla truncata	0	1	33	G5
Villosa iris	149	4	1, 2, 16, 33, 38	G5
Villosa vanuxemensis	12	1	14	G4

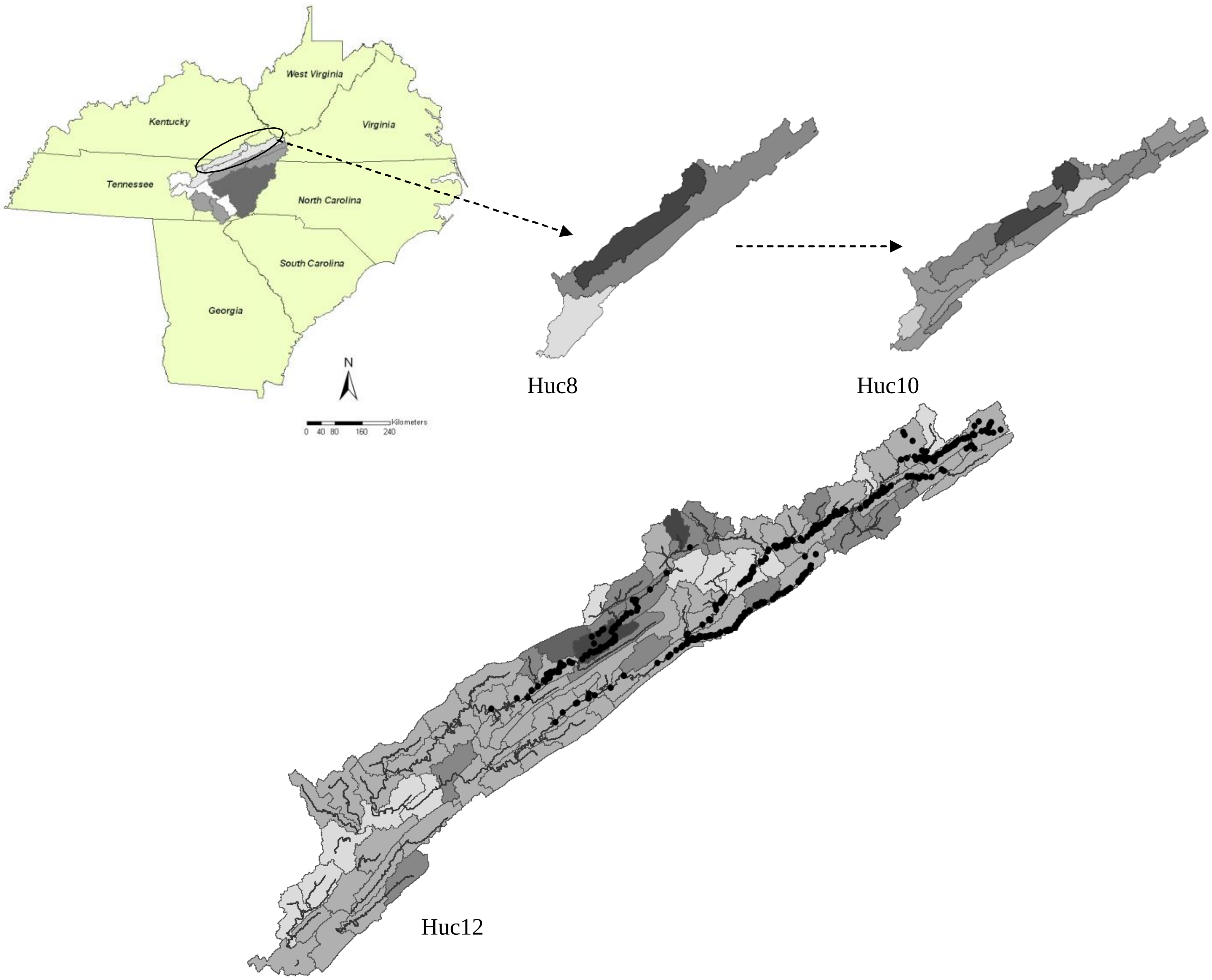


Figure 4.1. The Clinch River basin is located in southeastern United States, and composes part of the upper Tennessee River basin. Three hydrological-unit spatial grains are shown (HUC8, HUC10, and HUC12). Strahler 3rd-5th stream orders are shown at the HUC12 level, and mussel survey sites from 1963-2008 are indicated (black circles). Land use changes (1980-2001) are shown at each hydrologic unit level as varying shades of gray corresponding to no land conversion (no shading) to maximum land conversion (darkest shade: HUC8=5.6%, HUC10=5-8.75%, HUC12=15-20%).

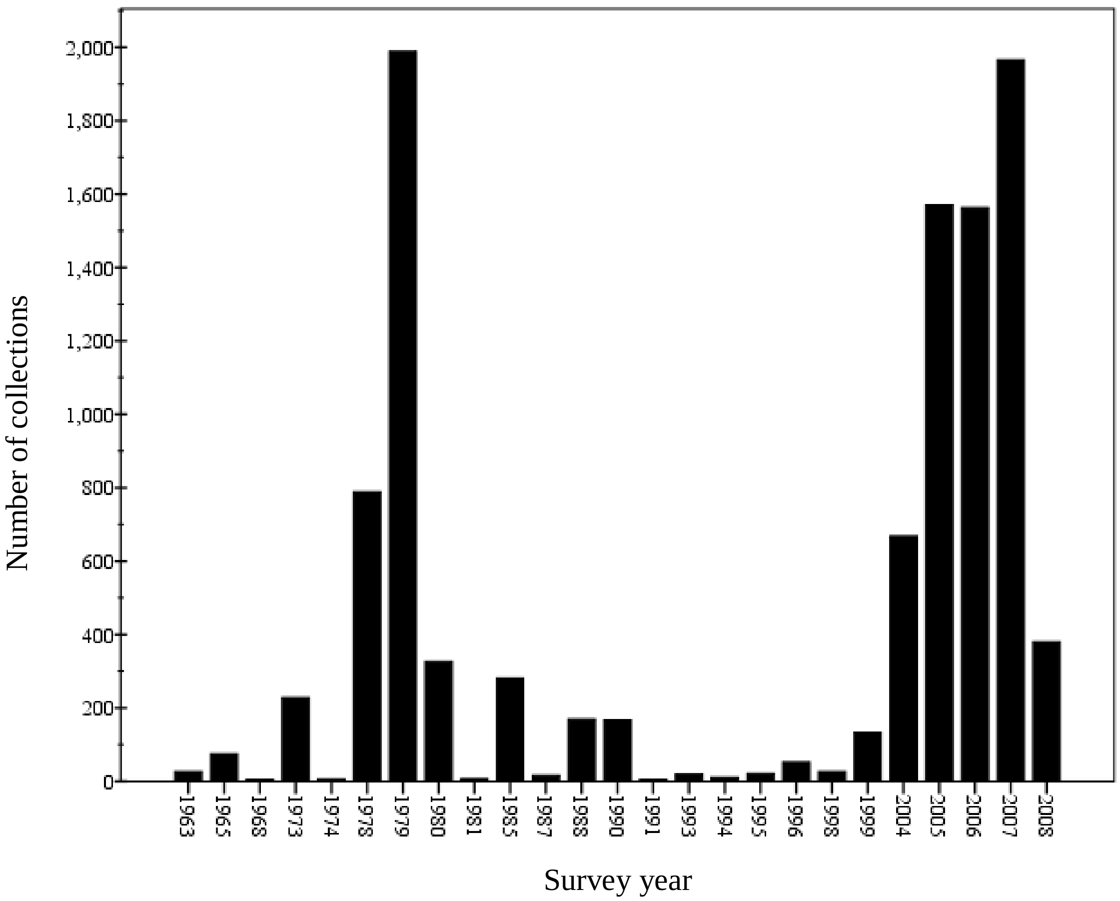


Figure 4.2. Numbers of freshwater mussel field surveys during 1963-2008 within the Clinch River basin, USA.

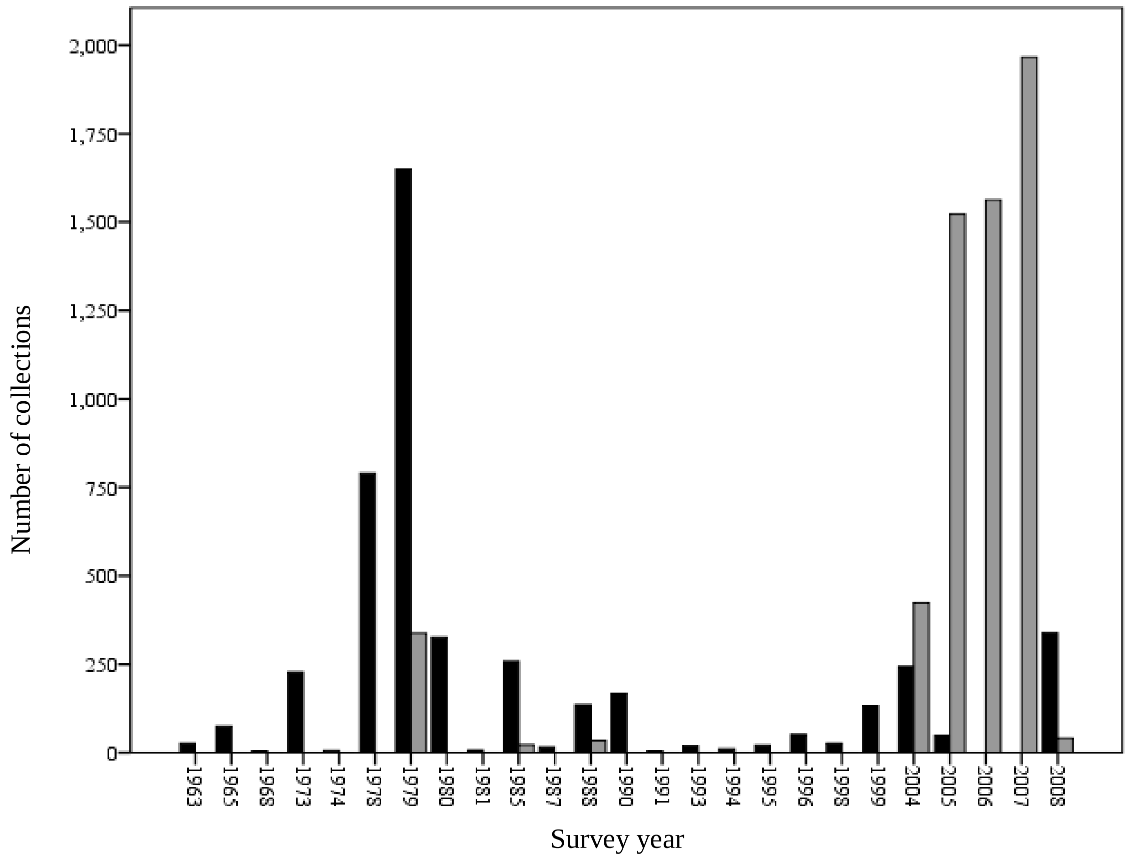


Figure 4.3. Frequencies of qualitative (black) and quantitative (gray) collection methods used at freshwater collection sites throughout the Clinch River basin during 1963-2008.

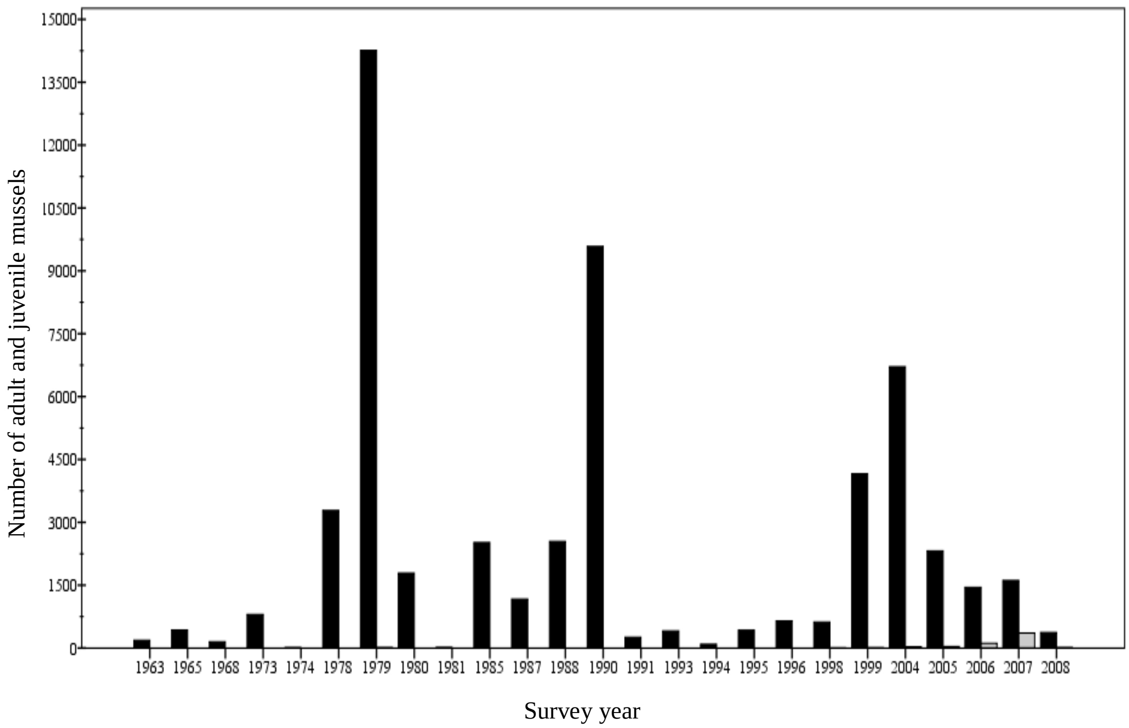


Figure 4.4. Numbers of adult (black) and juvenile (gray) freshwater mussels collected during 1963-2008 within the Clinch River basin, USA. Data are based on the numbers of live adult and juvenile mussels found in both qualitative and quantitative collections. Adult mussels were found in every collection year, ranging from 2 mussels in 1998 to 14,270 in 1979. Number of juveniles ranged from 0 in 19 of the 25 years, with collections of up to 356 in 2007.

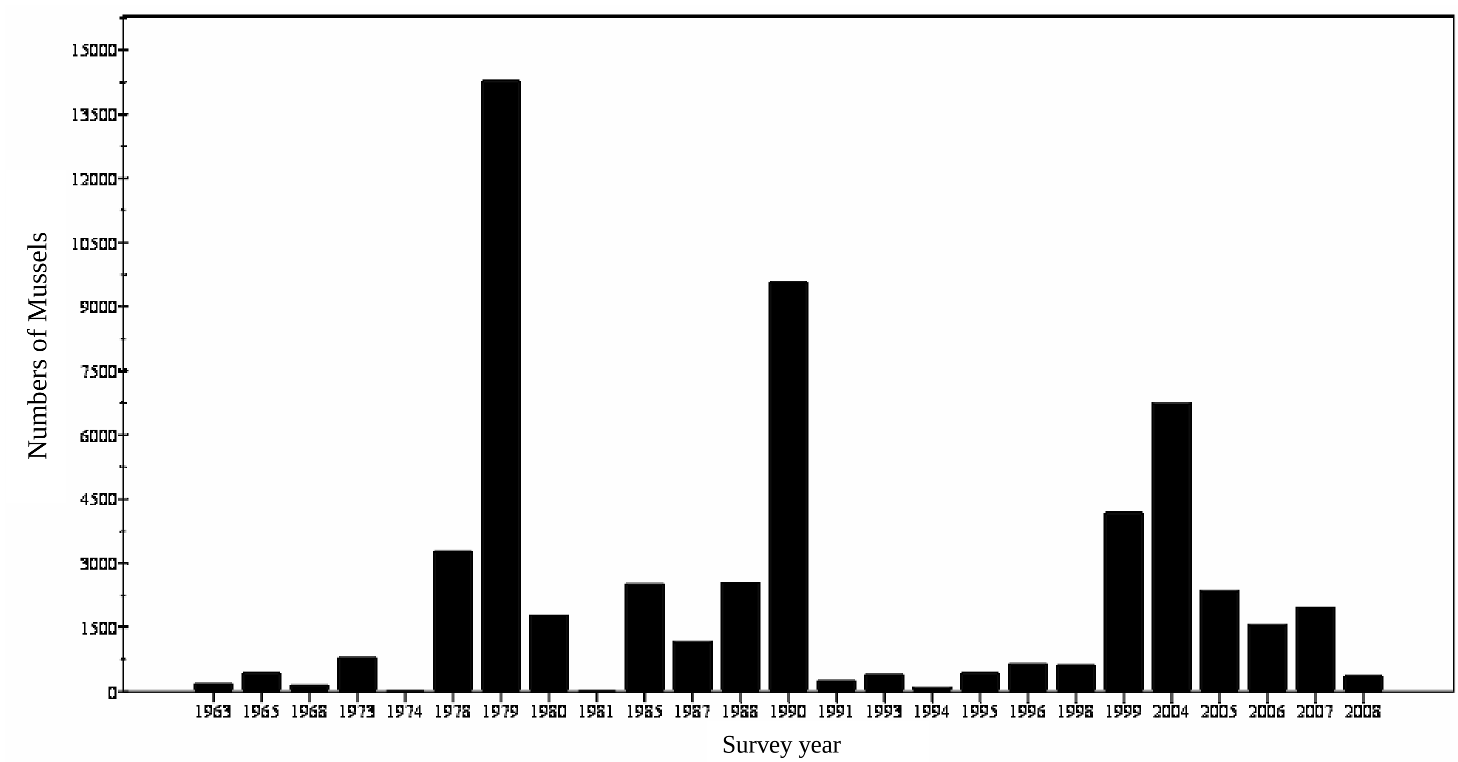


Figure 4.5. Estimated freshwater mussel population (all species) from 1963-2008 within the Clinch River Basin, USA. Population trend is based on the number of live adult and juvenile mussels found using both qualitative and quantitative collection methods.

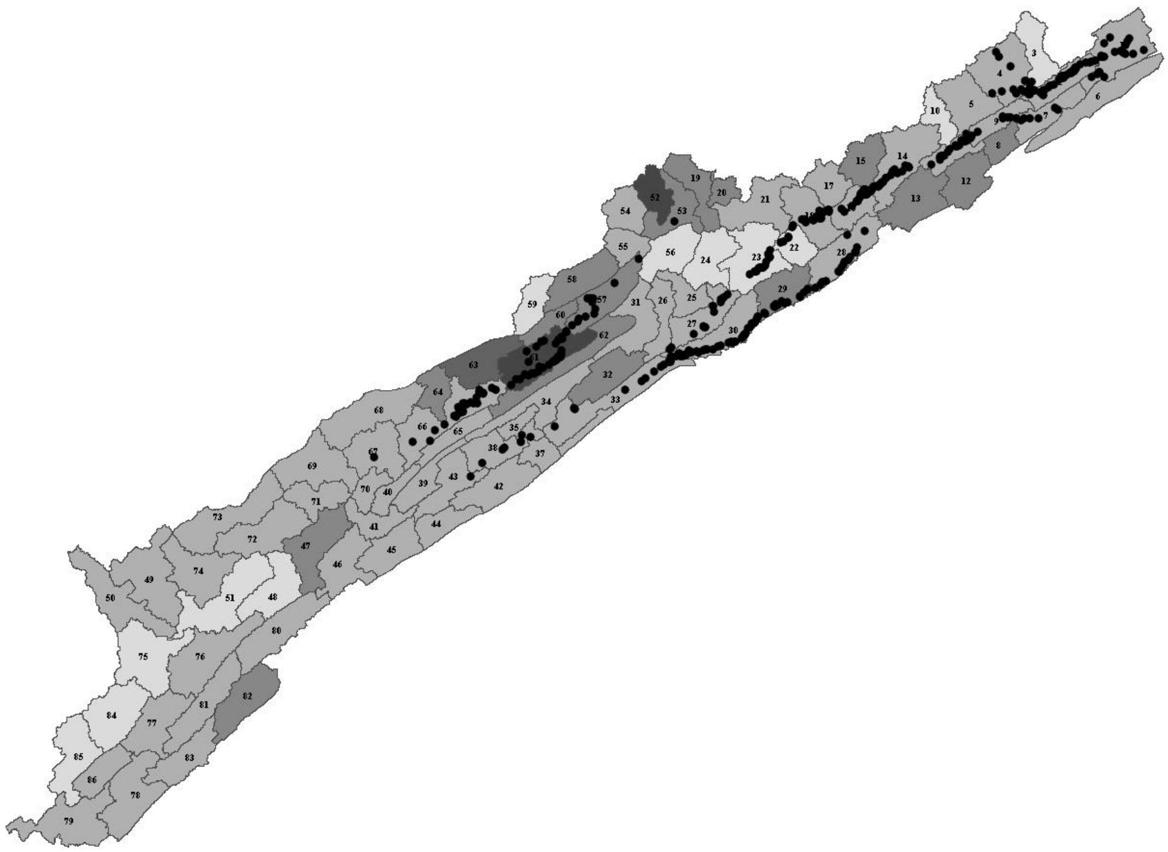
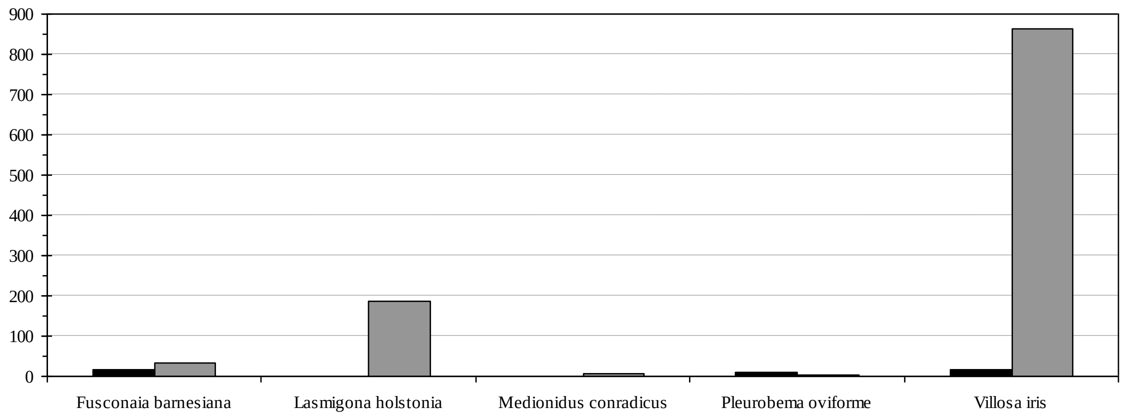


Figure 4.6. Land cover changes throughout the Clinch-Powell River basin, USA. Changes between 1982 and 2001 are represented as percentage of land cover differences on a pixel-by-pixel basis. Land cover change is represented by white through dark grey: <2% (white), 2- 5% (light grey), 6-10% (medium light grey), 11-15% (medium dark grey), and 16-20% (dark grey). Freshwater mussel collection sites are shown as dots. HUC12 numbers are shown for reference.

Appendix 4.A. Demographic trends for particular freshwater mussel species within 10 HUC12s of the Clinch River basin, USA, with field collections from both 1963-1984 (black) and 1998-2008 (gray). Total counts (vertical axis) are included for both time periods as long as individuals of a species were found in at least one time period. Watershed (W) number corresponds to the other tables and figures in this study. 12-digit hydrologic units are in parentheses.



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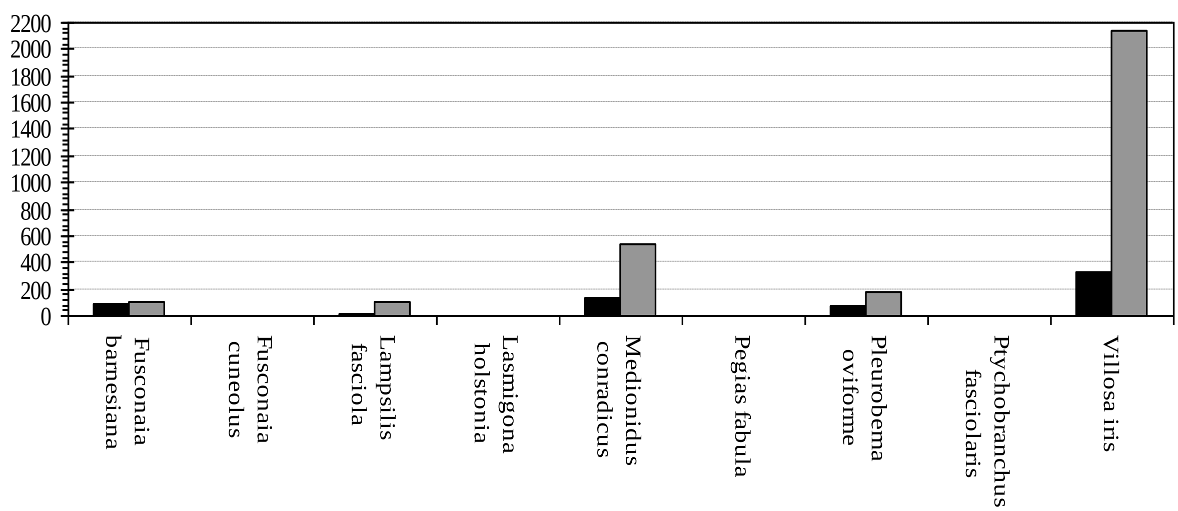

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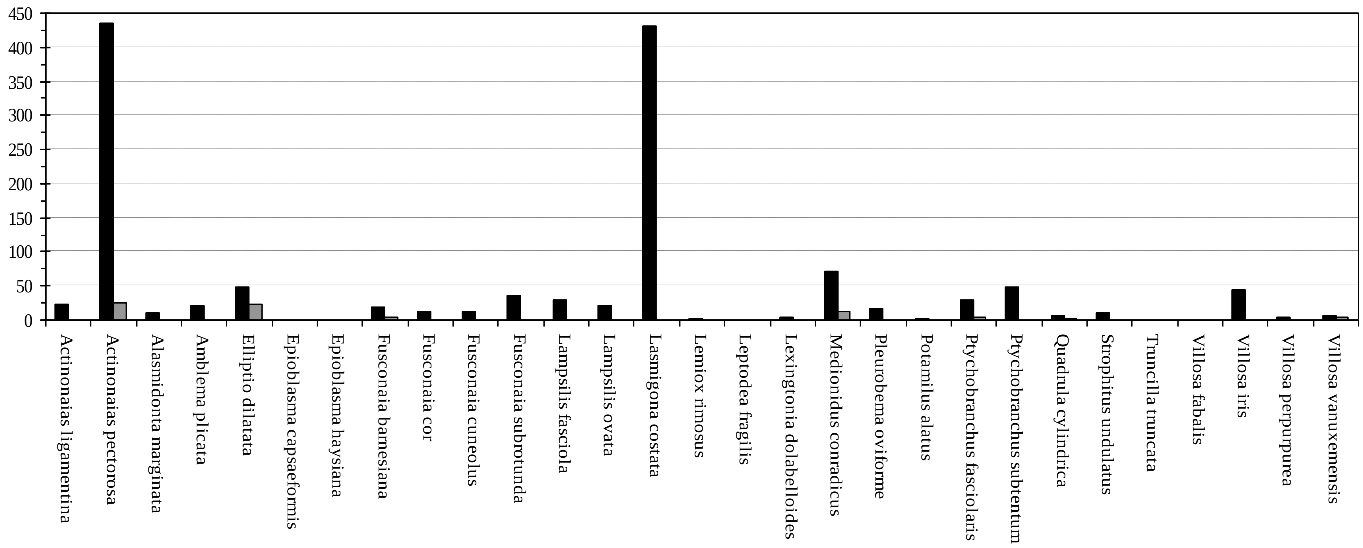

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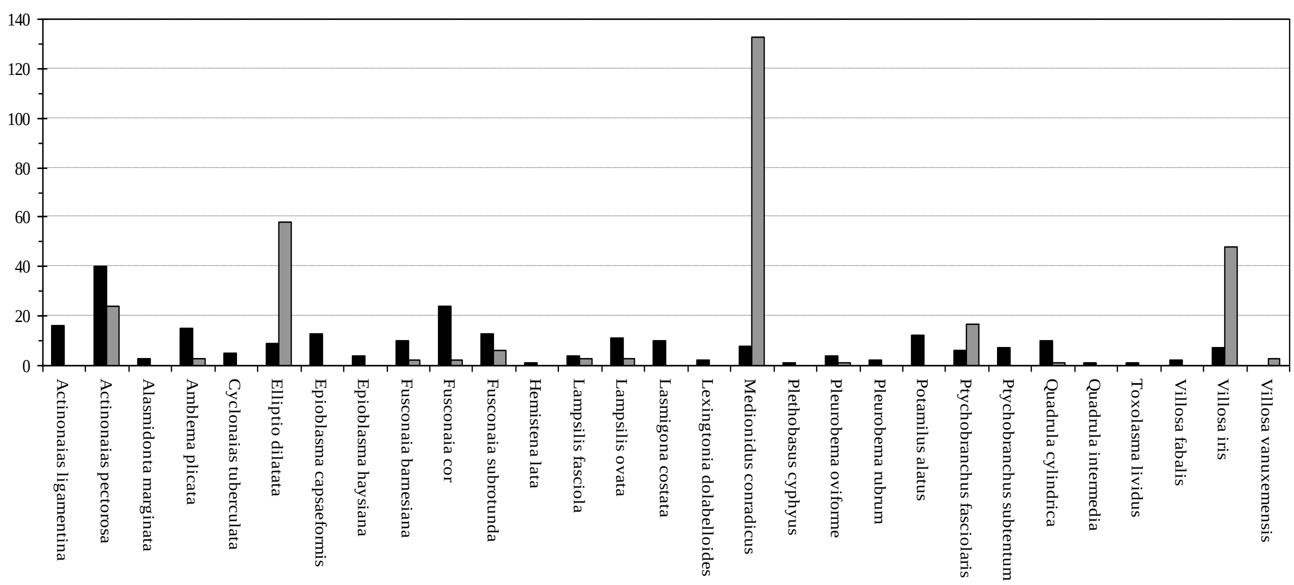

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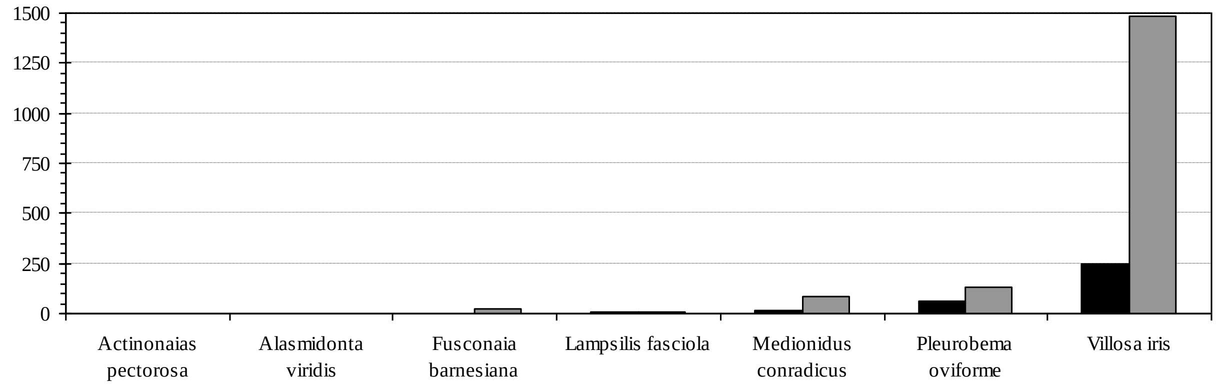

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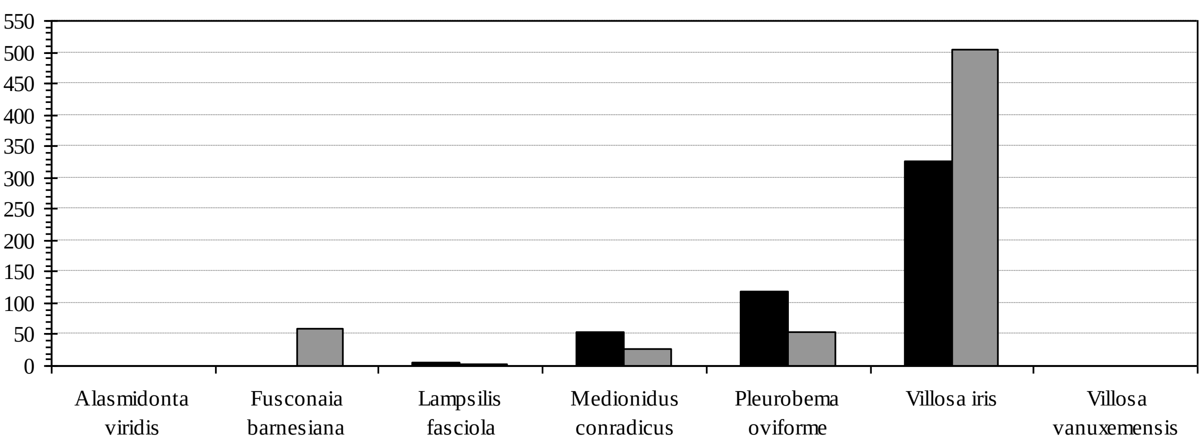

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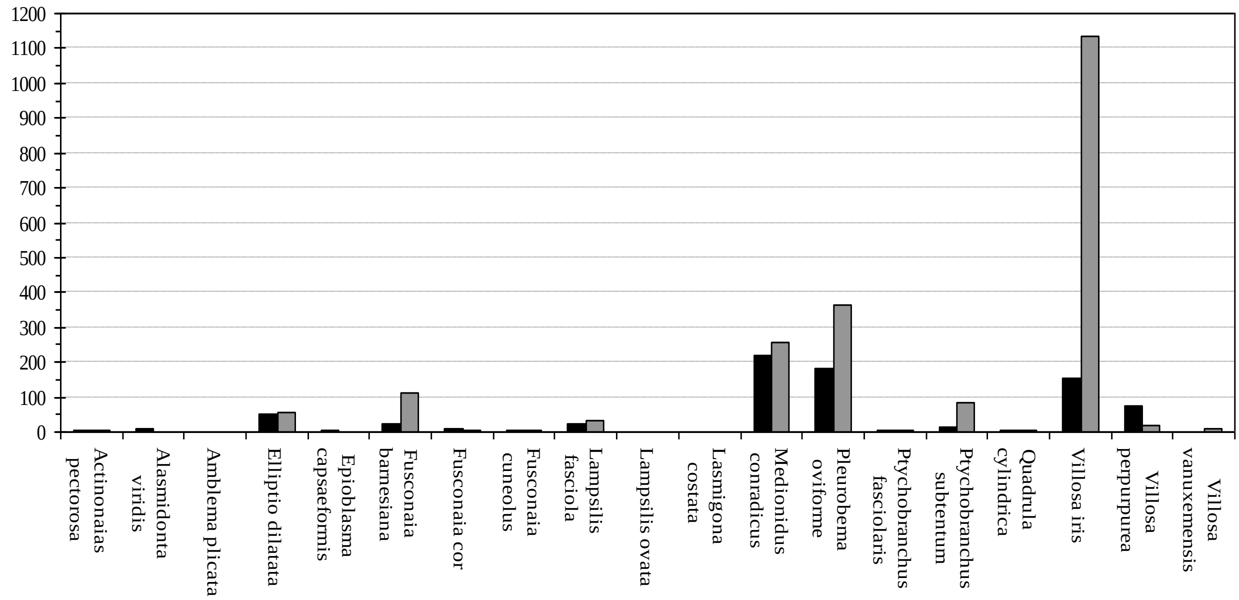

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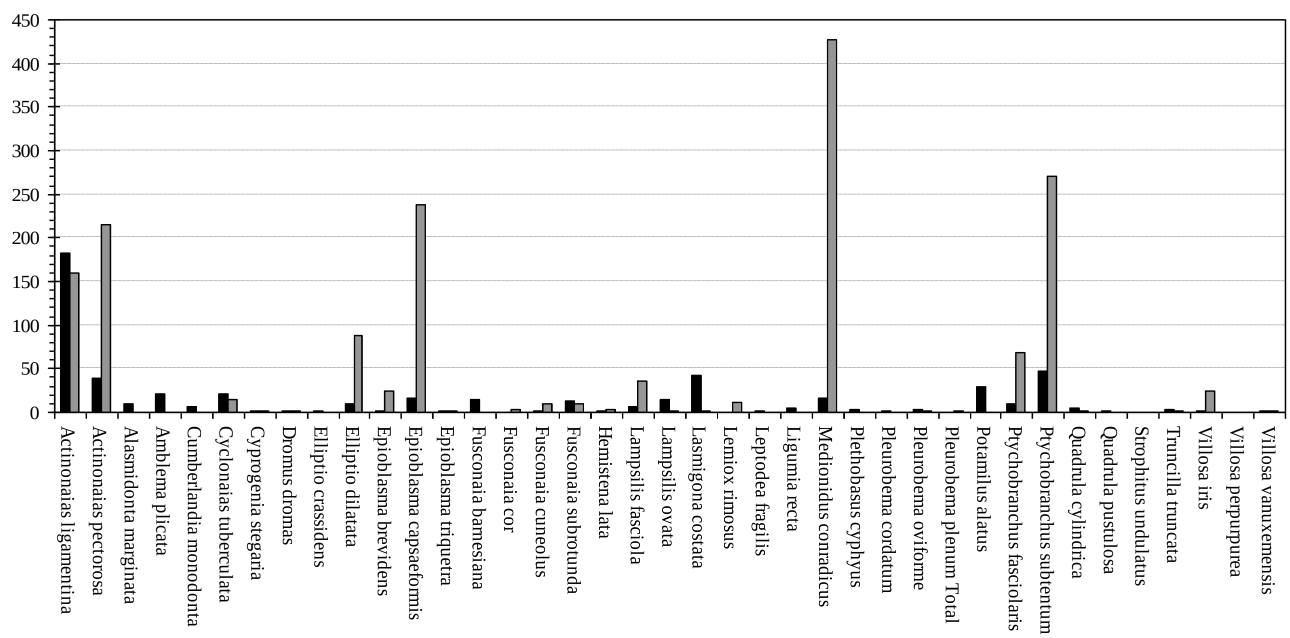

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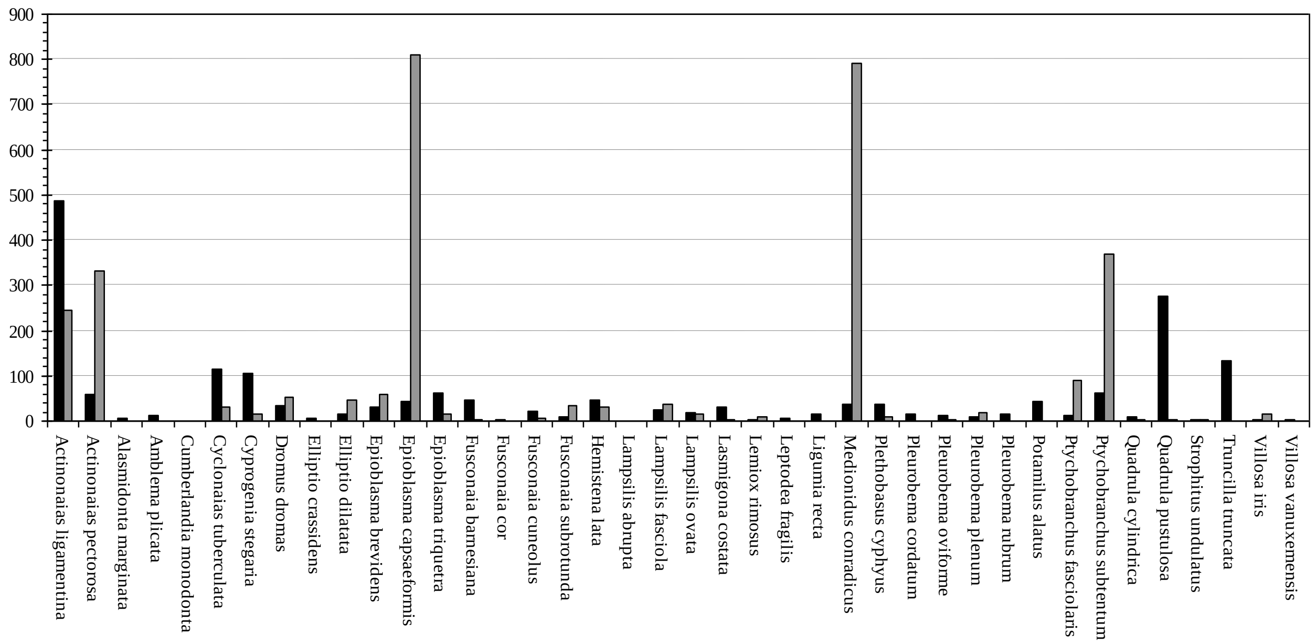

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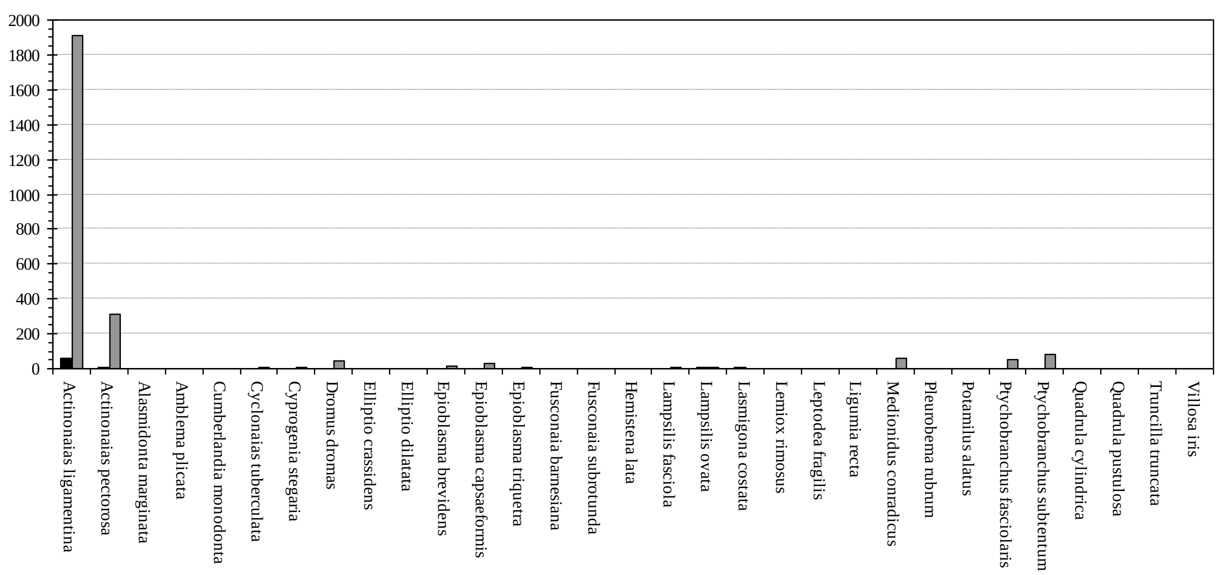

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GENERAL CONCLUSIONS

My dissertation tells a story about the challenges associated with conserving freshwater ecosystems, including how to incorporate risks into the planning process when impacts are difficult to measure, determining the effectiveness of risk models when data are scarce, and studying the impacts of land use change on aquatic species assemblages. While each chapter focuses on a particular aspect of freshwater conservation planning, there are still many challenges to address.

Ecological and evolutionary processes ultimately are as much of a concern in a biological diversity conservation strategy as are species diversity and composition. The Ecological Risk Index (ERI) uses this perspective in a proactive approach to risk assessment by identifying regions with gains/losses of ecological integrity as its endpoint. As human-related threats fluctuate throughout a region, so may the negative effects on biotic drivers, and parts of a watershed may be more impacted than others. I found that there were a greater number of threats within smaller streams, and fewer mussels within those areas today than historically. In fact, aquatic diversity has declined historically throughout the UTRB, including in those catchments considered to have moderate risk, because many of the streams are impacted by land uses in headwater areas (Neves and Angermeier 1990).

Improving the conservation planning process does not require a reinvention of techniques and concepts. Risk-based assessments provide an adequate basis for characterizing the impacts of human activities on conservation targets. Given that all applications and techniques have limitations, borrowing a framework and tools from an established field is often advantageous to developing a new approach (Stem et al. 2005). I focused on explicitly addressing the risks to biotic drivers to inform conservation planners of threats to conservation targets, which afforded a cost-effective and holistic view of the impacts of human activities on both terrestrial and aquatic systems. The ERI is a practical approach to summarizing anthropogenic impacts and ranking the conservation potential of freshwater ecosystems. The index is unique because it combines qualitative data from biologists with information about threat frequency and literature-based degradation thresholds for estimating the degree to which individual threats are likely to impact catchments.

Land use was a major factor in my dissertation for two reasons. First, land use data was a surrogate for field survey data within the ERI construct. Quantitative data relating to the impacts associated with individual threats throughout the upper Tennessee River basin would have been nearly impossible to collect in a timely manner, and using readily available spatial data made the ERI a reasonable approach to risk assessment, both in my work and in later analyses by others (Paukert et al. 2011, Zhang and Chen 2014). Second, the conservation status of freshwater species is often linked to the degree to which humans influence the surrounding landscape (Jackson et al. 2004). For example, many of the land uses within the Clinch River basin are associated with increases in nutrient loads or sediments, water temperature fluctuations, and agriculture (Saunders et al. 2002), and may make some mussel populations vulnerable to extirpation (Jackson et al. 2004).

My dissertation results attest to the importance of analyzing long-term datasets in the context of setting conservation priorities. Freshwater mussel species are at risk globally (Regnier et al. 2009), and human activities have long been implicated as major causes of decline (Williams et al. 1993). I related landscape changes to population status over space and time, and found the situation to be very complicated. Consistent with past research suggesting that human impacts are most threatening to aquatic species when they occur closer to physical habitats

(Allan 2004, Diamond and Serveiss 2001), my results suggest that neither basin-wide nor riparian-zone land cover changes explained mussel declines. Rather, long-lasting demographic impacts of infrequent toxic events may be most important.

Ecological Risk Index

The ERI is an assessment tool for evaluating the frequency and severity of threats to ecological integrity. It is meant to be a coarse filter for identifying patterns of regional land uses and impacts and might be used in conjunction with higher-resolution data for local planning. Readily available data (i.e. satellite imagery and previously collected point data) were adequate for providing an overview of current threats within the UTRB, and differences in risk patterns among drainages reflect predominant land uses.

The ERI protocol comprises five main steps to identify threats, determine their impacts within a watershed, and map risk regions for use in conservation planning. First, readily available and mappable land and water uses, termed threats to ecological integrity, are summarized and mapped. Second, expert opinion and journal publications are used to assign severity scores based on potential impacts of each threat to ecological integrity. Third, the frequencies of each threat within predefined spatial subunits are estimated. Fourth, a threat-specific index of ecological risk is computed for each subunit. Lastly, composite index of ecological risk is computed over all threats for each subunit.

The ERI-C scores use expert judgment in two ways. First, severity scores are assigned by inquiring about local impacts to stream systems. Severity was scored for each threat independently, and synergistic or cumulative effects from multiple threats were not considered in severity scores. Second, expert judgment was combined with equal intervals of threat frequencies and scientific literature to define degradation thresholds for frequency classifications. Threats with direct or continual influences on streams generally exhibited higher severity scores than threats located farther from streams or with intermittent effects. Watersheds with greater frequencies of intensive land use had higher ERI-C scores, and threats with high severity scores (i.e., magnitude) affected risk rankings independently of their frequency.

The ERI may be improved as more reliable threshold relationships are identified for larger spatial scales. For example, our application of subjective classification techniques, such as using equal intervals when assigning frequency classes, would be more informative if threat-specific threshold data were available. Furthermore, additional field surveys and monitoring studies focused on biotic degradation would enhance the reliability of threshold values, but the information gain may not be worth the resources needed to obtain it.

Testing the Ecological Risk Index

The ERI is subject to the general criticisms voiced about other risk-based planning tools (e.g. Wolman 2006), and the ERI is similarly sensitive to the methods used to classify risk within its framework. The purpose of this chapter was to compare the results of quantitative and qualitative data in ERI model outputs to better inform conservation planners of the risks and benefits of using a rank-based risk planning tool. The analysis was used to 1) individually address the frequency and severity components and 2) compare the outcome of my methods to that of a systematic approach of assigning frequency and severity classifications. This chapter considered three aspects of the ERI: 1) threat-frequency classes, 2) the number of final risk rankings used to group ERI-C scores, and 3) the role of expert judgment in a rank-based risk model.

I found that the ERI output is more sensitive to the process used to classify threat frequencies than to differences in how experts judge threat severity. Monte Carlo simulations of the average severity score distributions indicated that the ERI was sensitive to the risk ranking methods used and that expert judgment affected the distribution of catchments among risk levels. Overall, experts assessed risk to be lower than risk rankings derived from a quantile-based approach. Additionally, cluster analysis indicated that the ERI consistently distinguished among multiple risk levels. The results for the quantile-based approach indicated that three risk levels, as in the original ERI, were most appropriate and informative, and we retained these classifications. Clusters associated with the final risk rankings indicated that individual threats had unique patterns among risk ranking levels, suggesting that risk rankings might distinguish among the degrees of impact related to individual threats.

Variations in severity score assignment indicated that an expert-derived scoring system is a key factor in addressing the degree to which threats impact individual catchments. My investigation of severity score assignments also suggests that expert judgment was more conservative in assigning risk than was an assigned classification system such as the quantile-based approach. Additionally, expert judgment produced more conservative and biologically sound risk rankings than did randomly assigned severity scores. Several threats were especially influential in determining ERI-C scores and risk rankings for each analytical approach. Lastly, three final risk ranking levels provided the most useful information for mapping and management purposes.

In conclusion, expert judgment is useful in risk-based analyses, and planners who are unsure of how to construct a risk-based tool of their own might apply multiple assessment pathways to confirm results such as comparing multiple ranking procedures, varying model inputs of a specific approach, or swapping data to compare results (Halpern et al. 2007, Paukert et al. 2011). Rank-based risk assessment tools, such as the ERI, are valuable to conservation planners and managers because they are a cost-effective and flexible means of identifying areas with the least or most potential for supporting a full complement of native biota.

Database considerations

The purpose of the next two chapters was to evaluate how changes in land cover, which is a surrogate for the various land and water uses occurring throughout the Clinch River basin, have altered the biological integrity and species diversity within a freshwater ecosystem. This evaluation of the impacts of humans on aquatic systems had two parts. First, a database, the Upper Tennessee River Mussel Database (UTRMD), was constructed to complete the objectives of Chapter 3. The UTRMD was compiled from over 50 years worth of mussel surveys conducted throughout the upper Tennessee River basin, including the Clinch River basin. Second, the database was used in Chapter 4 to determine the long-term impacts of human activities within a region with a historically diverse number of indicator species. The conclusions from Chapter 4 are discussed in the next section.

The UTRMD is the most comprehensive compilation of species presence over time and place within the UTRB. The database comprises >3000 independent sampling sites from nearly 100 studies, reports, and other databases from 1963-2008. This translates to >47,400 species records, with nearly 2,100 sampling sites within the UTRB. More specifically, the database contains 579 sampling sites and 32,928 species records for the Clinch-Powell river basin. Aggregate counts suggest that mussel abundance was highest in the early years, dropped to its lowest in the 1990s, and is somewhat greater today. Recruitment is ongoing in several sites throughout the basin, suggesting that physical habitat conditions are suitable, and at least some mussel species may be demographically viable.

Field surveys have not been spatially consistent over the past 30 years, but in those areas with repeated mussel surveys, counts of both adults and juveniles have increased. For example, there was a 24% decline in total numbers of mussels found since 1963 basin-wide; however, counts have increased since reaching a record low in the 1990s. This suggests that either habitat conditions have improved enough that more juveniles are surviving to adulthood, or increased sampling intensity has artificially increased observed recruitment. Hence, although mussel counts are below those of 50 years ago, recent field surveys suggest that the assemblages are rebounding.

Inconsistent sampling techniques, as used over the course of my study period, present a challenge for evaluating how fluctuations in counts are related to local and regional abiotic and biotic conditions. The fact that collection methods are inconsistent across time and space does not negate the importance of evaluating results of qualitative and quantitative surveys together. Despite these limitations, my results have been useful in identifying key overall patterns in freshwater mussel abundance within the Clinch River basin, and relating land cover changes to mussel species distributions.

Species data are currently collected in a similar manner by the numerous agencies and other researchers involved in freshwater mussel research, which makes the UTRMD a relevant addition to conservation planning. With access to an online, central database such as the UTRMD, opportunities arise for additional research and may provide funding for more extensive national and global conservation actions (Darwall et al. 2008, Nobles and Zhang 2011). In the long-term, information housed in the database may be used to achieve critical research goals such as ongoing land use/restoration planning. Additionally, a historical record of mussel bed activity is paramount for identifying areas appropriate for future mussel restoration, and the availability of a central archive to house species data provides a foundation for focused and effective management in continuing conservation efforts.

Land cover change and freshwater mussels

This chapter focused on the relationship between historical land use and freshwater mussel species distributions to determine whether land cover patterns over time and space are related to mussel population status. Freshwater mussel species are good candidates for studying the impacts of land use changes on stream integrity because many of these species are long-lived and have lifestage-related sensitivities to water quality and physical habitat conditions (Neves et al. 1997). I took a multi-scale spatial approach to study the relationship between land cover and mussel distributions within a watershed containing a long history of mussel surveys. My study objectives included: 1) describing historical land cover patterns within watersheds of the Clinch River basin; 2) relating spatiotemporal patterns in freshwater mussel distributions to patterns in riparian land cover; 3) evaluating the impacts of human activities on species assemblages within existing mussel beds, and 4) discussing the conservation value of decades of freshwater mussel data.

My land cover analysis suggests that those portions of the Clinch River basin in which mussel sampling regularly occurs have not had significant land cover changes within the past 50 years. Although riparian areas throughout the Clinch River basin are especially vulnerable to land conversion, such impacts are presently distant from mussel beds with active recruitment. Furthermore, the majority of land conversions took place within HUC12s containing well-established industrial or urban areas in which mussels had been extirpated or never occurred. There have been no published mussel surveys for these streams for >50 years. Consequently, there is a spatial mismatch between land cover changes and mussel bed hotspots.

Some stream reaches with seemingly healthy adult populations are located in disturbed or chronically polluted areas. These reaches have little or no recruitment, and hold demographically senescing mussel communities. Adult mussels tend to be more tolerant of stressors that produce heavy metals, increase sediment, and alter water conditions (Neves and Angermeier 1990, Hampson et al. 2000). Such impacts often harm juveniles without permanently reducing adult mussel populations (Strayer and Malcom 2012). While land cover has remained relatively stable within those reaches exhibiting recruitment, without adequate protection from chronic and acute stressors, recruitment may potentially decline even in these reaches, posing implications to mussel population viability.

CONCLUSIONS

The ability of natural communities to continue to provide ecosystem processes and retain species diversity will continue to be a major concern for the conservation community. The ERI was developed to aid researchers with ranking conservation concerns within a landscape so that further research may be directed toward specific issues. Since the ERI uses readily available data and expert judgment to guide the conservation planning process, this is a pliable and simple addition to current risk assessment procedures. Additionally, the results of my analysis on the impacts of land cover changes on aquatic systems indicate that these systems respond to human activities in complex ways and further research is needed to understand how species assemblage patterns relate to current environmental conditions. Lastly, the UTRMD was constructed to assist conservation planners with identifying prospective research objectives by providing a comprehensive analysis of mussel species assemblages. This database contains information on historical collections of all the mussel species within the UTRB, and may be linked to national and global resources.

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