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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 sustainng the natural physical, chemical, and biological processes within representative ecosystems. Often data to measure these components are inadequate or unavailable. The impact of human activities on ecosystem processes complicates integrity assessments and might 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 locally collected data might not accurately reflect potential impacts. We suggest that changes in five major biotic drivers—energy sources, physical habitat, flow regime, water quality, and biotic interactions—might be used as surrogates to inform conservation planners of the ecological integrity of freshwater ecosystems. Threats to freshwater systems might be evaluated based on their impact to these drivers to provide an overview of potential risk to conservation targets. We developed a risk-based protocol, the Ecological Risk Index (ERI), to identify watersheds with least/most risk to conservation targets. Our 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. We illustrate application of our protocol with a case study of the upper Tennessee River basin, USA. 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 our watersheds. We also found that the relative importance of threats is sensitive to the spatial grain of the analysis. Our case study demonstrates that the ERI is useful for evaluating the frequency and severity of ecosystemwide risk, which can inform local and regional conservation planning.

Keywords Freshwater conservation · Conservation planning · Ecological integrity · Biotic driver · Impact assessment · Human disturbance · Land use · Ecological risk

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Introduction

Conservation planning focuses primarily on conserving species and ecosystems of interest within an ecologically sustainable management program (Abell and others 2002; Groves and others 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 representative conservation targets. Planning measures currently identify conservation targets by using abiotic and biotic

properties to represent biodiversity in large geographic areas, whereas 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). Current assessments for conservation planning rarely 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 would 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 in 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 properties and the surrounding landscape, making conservation actions complicated because both the regional context and local disturbances affect ecological integrity (Allan and others 1997; Roth and others 1996). Impacts to ecological integrity correspond to the amount of disturbance within a system, as measured by human-induced drivers that negatively alter ecosystem functions. Initially, we intended to integrate established risk assessment methods into freshwater conservation planning, but we found that these methods did not evaluate ecosystem drivers or use ecological integrity as an end point in assessing risk. With this in mind, we developed a risk-based approach to inform regional planners of localities with least/most potential for protecting conservation targets so that local management decisions might be more informed and effective.

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

Role of Ecological Integrity in Risk-Based Assessments

Human land use affects both local and regional assessments of stream conditions (Lammert and Allan 1999; Roth and others 1996; Wang and others 2001). Impairments to the ecological integrity of streams can be classified using physical, chemical, and biological components collected locally (Detenbeck and others 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 (Hughes and Hunsaker 2002; O'Neill and others 1997); however, specific impairment pathways are often unknown, making causal links difficult to confirm.

Risk assessment measures the exposure of an end point to a stressor (Table 1), and the purpose of most risk studies is the quantitative assessment of alterations in function and condition of selected end points (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 and others 1997). Stressors should be easily identified and impacts should be easily quantifiable if conventional risk-based approaches are applied at regional scales.

Protecting ecological integrity is the ultimate goal of conservation planning, and staging risk-based assessments requires consideration of potential declines in physical, chemical, and biological components. The effects of stressors on ecosystem integrity can 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 and others 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 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.

We suggest that ecological integrity is an appropriate assessment end point for evaluating the risk of human impacts to stream systems, much like biotic integrity is already used as an end point for assessing the impacts themselves. Alterations in the major drivers (i.e., energy sources, physical habitat, flow regime, water quality, and

Table 1 Comparison of terms used in the continuum of risk-based analyses

	Conventional risk assessment	Ecological risk assessment	Threat assessment
Threat	Not applicable	Source of stress	Anthropogenic source of stress
Hazard/ stressor	The source of an adverse effect (e.g., industrial plant)	Act or phenomenon that has the potential to do harm; proximal and distal stressors might be identified	Anthropogenic source of stress
Risk	Probability of occurrence due to exposure to hazard	Probability of occurrence 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 end-point 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 toxin release)	None measured; changes in the natural range of variability could be considered
Impact	Quantitative measure of the amount of harm of a hazard to an end point	Quantitative measure of the amount of harm of a threat/ hazard to an end point	Measure of existing or potential harm of a threat to a study area
(Exposure) Pathway	Route that substance (hazard) takes through system; human health-related (e.g., endocrine-affecting substances travel through water, are ingested, attack liver, pancreas)	Route from a threat/hazard to an end point; ecosystem derived (e.g., stream route of toxin output)	Usually not determined due to system complexity
Assessment end point	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 end point	Quantitative measure of the response to a hazard; human health-related (e.g., amount of damage in liver, pancreas)	Quantitative measure of the response to a threat/hazard; species related (e.g., number and species of fish killed)	Measures system components related to ecosystem condition

biotic interactions) of freshwater systems ultimately affect species distributions and abundances, and such drivers might be used as surrogates to evaluate overall ecological integrity (Karr and others 1986; Poff and others 1997). Stream monitoring at local scales documents changes in biotic drivers, indicating adverse effects from land/water uses. When applied to larger spatial extents, data from local studies might aid in estimating potential stressor impacts (Lammert and Allan 1999). Focusing on biotic drivers during risk assessment emphasizes the importance of sustaining ecosystem functions to minimize loss of valuable populations and communities (Baron and others 2002; Walker 1992). Applying threat evaluation to a framework for freshwater conservation planning, and, specifically, threats to watersheds, might 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 end points, such as shoreline habitat and in-stream condition, that might be affected by a variety of known risks (Norton and others 2002; Wiegers and others 1998). Conventional single stressor versus single end point 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 and others 2002; Norton and others 2002) and might be aggregated to identify multiple end points and summarize risk within watersheds (Suter and Barnthouse 1993).

We reviewed the meanings of terms commonly used in risk-based analyses and looked for shared features that could be applied at regional scales (Table 1). We used accepted definitions in our evaluation to compare similarities and differences between risk and impact assessment (Stem and others 2005). In particular, we defined components appropriate for use in regional studies and identified established methods that might include the evaluation of ecological integrity as an end point. Herein, we describe how study focus, statistical

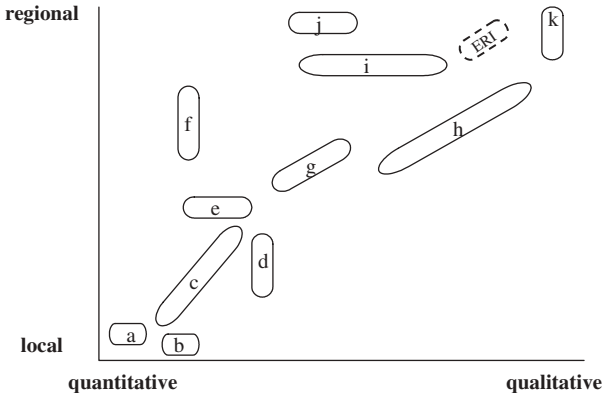


Fig. 1 Schematic representation of risk-based assessments along gradients of spatial scale and methodological rigor. Horizontal position reflects the relative importance of quantitative analysis. Vertical position reflects spatial extent. Each ellipse represents one or more published assessment approaches. Refer to legend for letter explanations

tools, and data collection differ at various spatial extents. We chose representative studies from the literature to summarize approaches used for risk-based assessments over a range of spatial scales (Figure 1).

Methods to assess human-induced disturbances that negatively affect ecosystem functions and processes might be ordered along a quantitative-to-qualitative axis as well as along an axis of spatial extent (Figure 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. 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. We found no studies that focused on a single stressor at large 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 and others 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 and others 1999; Walker and others 2001; Wiegers and others 1998), ranking impacts based on risk classes (Slob 1998), and using ranks in land-use intensity or land-cover change to compare watersheds (Detenbeck and others 2000; Turner and others 1996). 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 (Bryce and others 1999; Muhar and Jungwirth 1998).

In summary, much recent work applies risk-based approaches to conservation planning, but there are no standards yet. Data requirements and analysis 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 lenses of fish and macroinvertebrates, 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 and others 2000; Muhar and others 2000; Slob 1998; Walker and others 2001). We 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. We found that the basic tenets of risk assessment, namely the probability of risk occurring based on threat frequency and severity within the system of interest, were not explicitly applied as the spatial extent increased.

Ecological Risk Index

We gleaned three key concepts from our 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 end point. Next, threats were commonly weighted according to prevalence, and impacts pertained to physical components of ecosystems. Finally, aggregates of locally collected data were useful in identifying regional threat patterns. We used these concepts to build a protocol for assessing the impacts of anthropogenic stressors on the ecological integrity of watersheds.

Our protocol, the Ecological Risk Index (ERI), integrates the frequency of various land uses with estimates of their potential impact on biotic drivers (Table 1). Briefly, the ERI uses a ranking procedure to identify areas of low, moderate, and high risk to stream biota based on the potential harm of identified threats to the flow regime, physical habitat, water quality, energy sources, and biotic interactions of a freshwater

system. We incorporated two aspects of risk assessment—frequency and severity—into our 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 forecast areas that might 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 land 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, unpredictable, occur mainly on or near bridges, and are sometimes unreported. Bridge data, therefore, were adequate 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.

We 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/least likely to perpetuate those species and communities (Scott and others 1993; Stoms 2000). With this in mind, the ERI had to be applicable to all stream biota and compare threats with a common biological currency. The ERI protocol comprises five main steps (Fig. 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 predefined 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 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/least in need of protection.

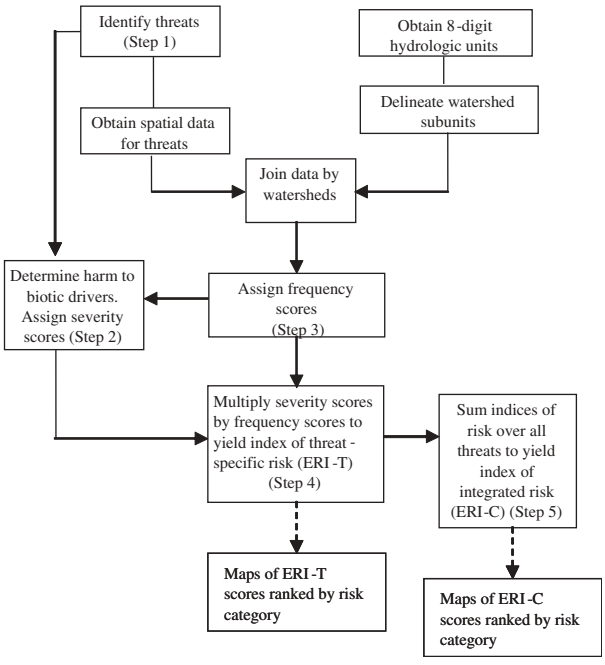


Fig. 2 Flowchart depicting major steps in developing the ERI within a threat assessment framework. All major threats within each subunit are summarized and maps are produced with final ranking scores to allow visual comparisons

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-subunit basis and estimating potential impacts of identified threats. Frequency scores are assigned based on total frequency counts per subunit. We assigned frequency classes at equal intervals of occurrence for lack of ecological data to inform us otherwise. Exceptions, for which empirically derived frequency classes have been referenced frequently in the literature, included urban and agricultural land uses (Finkenbine and Mavinic 2000; Fitch and Adams 1998; Wang and others 2000), roads (Forman and Alexander 1998), and dams (Ligon and others 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 (e.g., Table 2). We 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 3). 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 is 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 minimum and high impacts from individual threats can be identified. Maps can be generated to illustrate spatial distribution of subunits with low, moderate, and high ERI-T values. This facilitates a comparison of individual threats across a region, thus 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 reflect 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), 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 can 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 multistressor risk assessments, it uses the intrinsic properties 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 can be updated as needed. Parts of the protocol are based on expert opinion and local circumstances, which make it generally applicable. Conservation planners might 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, Tennessee (55 400 km^2) (Figure 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 and others 2000). The UTRB comprises mainly forest (65%) and agricultural lands (25%), with 6% of the basin urbanized (Hampson and others 2000). Although only a small portion of the basin has been developed, human activities have caused a continual decline in ecological integrity (Bolstad and Swank 1997; Diamond and Serveiss 2001, Diamond and others 2002; Neves and Angermeier 1990). Today, the UTRB has the greatest number of imperiled species per unit area in the continental United States (Hampson and others 2000).

Methods

We identified 12 major threats within the UTRB that could be characterized as either point data or land-cover categories: row crops, 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 (Diamond and Serveiss 2001, Diamond and others 2002, Hampson and others 2000; Smith and others 2002; Upper Tennessee River Roundtable 2000) and represent major pollution sources within the UTRB (Carpenter and others 1998). Pasturelands account for the majority of agricultural uses, with row crops occupying <3% of the entire study area (Hampson and others 2000). Impacts on the riparian corridor due to poor pasture management might be long-lasting (Harding and others 1998). Urbanization is an increasing and chronic threat to aquatic ecosystem integrity (Wang and others 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 the 1992 National Land Cover Data set and summarized from 30-m^2 cells. Surface hydrography from the 1999 National Hydrography

Table 2 Frequency scores (0: not present, 1: minimum, 2: moderate, and 3: maximum impact) used to compute risk indexes for 12 major threats within the UTRB

Threat	Frequency rank scores				Classification method or literature used in rankings
	0	1	2	3	
Row crops (%)	<2%	2–9%	10–49%	>50%	Wang and others (2000)
Pasture (%)	<2%	2–9%	10–49%	>50%	Wang and others (2000)
Urbanized areas (%)	<2%	2–9%	10–49%	>50%	Finkenbine and Mavinic (2000), Wang and others (2001)
Industrialized areas (%)	<2%	2–9%	10–49%	>50%	Finkenbine and Mavinic (2000), Wang and others (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

Note: 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

Table 3 Matrix of severity ranks (low = 1, moderate = 2, and high = 3) for major threats within the 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 (30-m buffer)	Low			X	X	X	
	Medium						9
	High	X	X				
Railroad density	Low			X	X	X	
	Medium	X	X				7
	High						

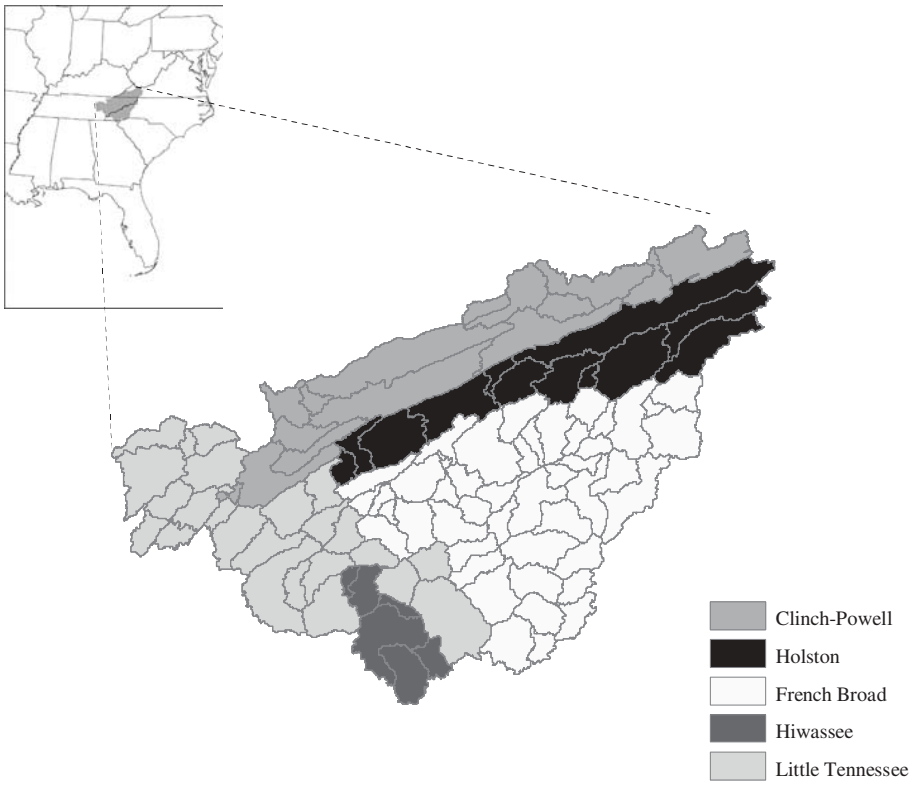


Fig. 3 The upper Tennessee River basin, USA. Delineation is of 107 subunits based on fouth-order streams. Major river drainages are depicted (shaded regions)

Data set was used to identify fourth-order Strahler stream reaches. We 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 fourth-order output. Surface flow within the five major watersheds listed in Figure 3 corresponds to the USGS hydrologic units. Dam location information was extracted from the 2001 National Inventory of Dams database maintained by the US Army Corp of Engineers. The TIGER/Line 2000 database (US 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 100 m) as an indicator of surface erosion.

We used the US 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. The US EPA NPDES-permitted facilities include various types of animal feeding operation, sewer and storm water overflow, and water pretreatment. Effluent data from municipal and manufacturing-related sites provided complementary, unduplicated information on point-source threats to ecological integrity.

We 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, might provide a more accurate assessment of local impacts, including such variables was not within the scope of our study. We also did not address land-use changes, as the ERI is not temporally or spatially explicit at this time. The ERI was constructed so that an alternative suite of threats could be used and/or severity scores could be updated as more knowledge becomes available.

Results and Discussion

Threats with direct or continual influences on streams generally exhibited higher severity scores than threats located farther from streams or with intermittent ef-

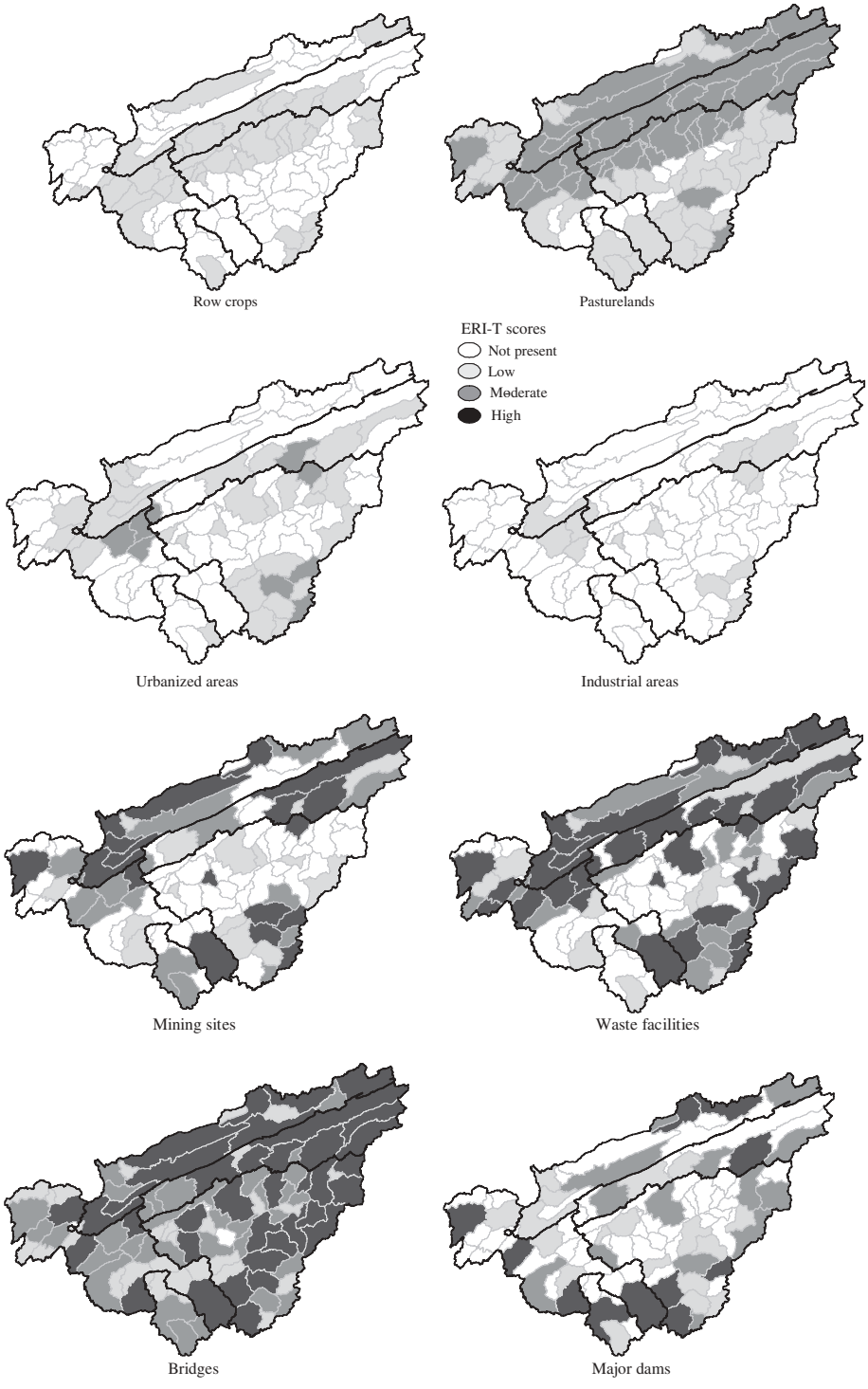
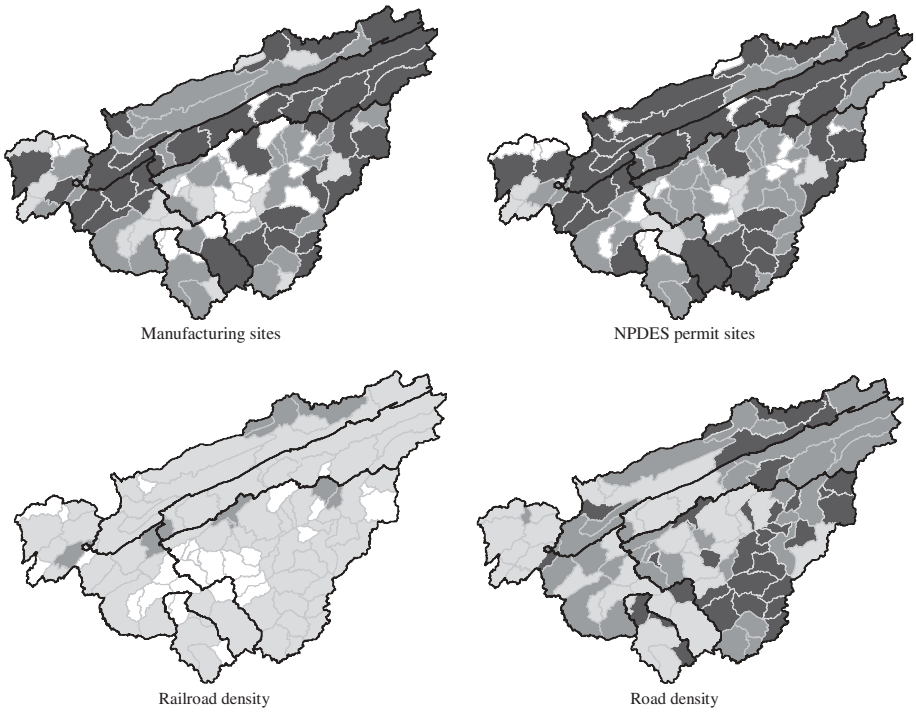


Fig. 4 Maps of ERI-Ts for 12 anthropogenic threats in 107 subunits within the UTRB. Low-, moderate-, or high-frequency scores were assigned by subunit

fects (Table 2). This pattern reflects land uses without adequate riparian buffers as well as threats occurring within stream channels, such as manufacturing sites located next to a stream or the direct impact of dams (Ligon and others 1995). The resultant ecological changes, such as water temperature changes, increased sediment, habitat alteration, and vegetation changes, have local and regional impacts on ecosystem functions



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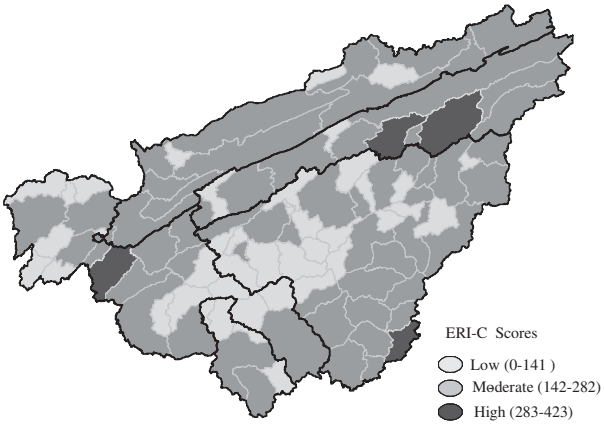

Fig. 5 Maps of subunits within the UTRB with low, moderate, and high values of an ERI-C to aquatic ecosystem integrity

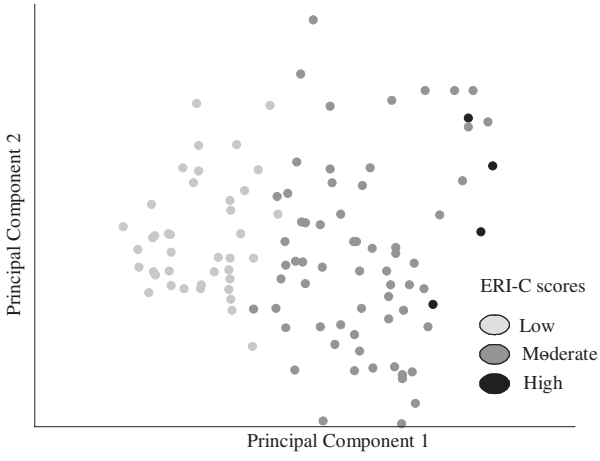

Fig. 6 Ordination of 107 subunits in principal components space defined by ERI-Ts. Subunits are labeled by their ERI-C. Some points overlap

(Hughes and Hunsaker 2002). All threats were weighted equally, and any differences in upstream versus downstream impacts were not considered in our analysis.

Maps of ERI-T scores reflect subunit-specific risk patterns for individual threats (Figure 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 data, such as manufacturing, waste disposal, and NPDES permit sites, suggest that industry-related land uses are much more prevalent than their areal extent indicates. This might 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 4).

Table 4 Loadings of major threats on the first two principal components (PCI and PCII) of 107 subunits within the UTRB

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

Note: Variance in threat composition among subunits explained by PCI and PCII is also shown

The 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 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 can be identified by high frequencies of risk or by low frequencies of severe threat. The 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 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 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 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 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 4), suggesting that dams impact UTRB streams independently of other threats.

Differences in risk patterns among drainages reflect predominant land uses (Table 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 were least common in the Clinch-Powell drainage and were also ranked as a higher risk in this drainage. The importance of waste facilities, bridges, pastures, row crops, and manufacturing sites varied significantly among the drainages (Table 5). These results are consistent with other studies that have found different causes of impairment as the spatial extent of analysis is varied (Moss 2000; Rabeni 2000).

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

Table 5 Mean frequencies and variances of 12 threats in watersheds of major drainages of the UTRB

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	2.0 (5.55)	1.0 (1.58)	1.0 (1.04)	1.0 (2.57)	1.0 (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)

Note: The number of subunits (n) in each drainage is also shown

impacts downstream. We 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.

Our case study demonstrates that the ERI is a useful tool for evaluating risk in local and regional conservation planning. Risk indexes combined the potential impact of human activities on system drivers with frequency of occurrence measures to summarize potential harm to system resources. Furthermore, large national databases seem adequate for use in prioritizing conservation areas when anthropogenic effects are explicitly addressed. We also found that the relative importance of threats is sensitive to the spatial grain of analysis.

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 might 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 it directs regional planning of conservation needs. Other freshwater-based classification approaches implicitly include risk in their respective viability assessments (Abell and others 2000; Groves and others 2002). Our approach complements these classification frameworks by addressing risks explicitly so that risk at various spatial extents can be integrated and compared.

We envision the ERI protocol as a tool for quantifying human disturbance patterns using readily available land use data. It is meant to provide an objective overview of impacts of human activities with data that are easily obtained. It also has the potential to be included in other classification frameworks (e.g., Higgins and others 2005). Although we rely on expert opinion to rank regional-level impacts to ecological integrity, analogous techniques have been shown to be useful for large-scale studies (Barve and others 2005; Bryce and others 1999; Walker and others 2001; Wiegers and others 1998). We believe that our approach of linking anthropogenic stressors with ecosystem drivers will prove useful in identifying areas that should be considered for conservation actions.

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, we believe that borrowing a framework and tools from an established field is advantageous to developing a new approach (Stem and others 2005). 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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