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Development and assessment of a landscape-scale ecological threat index for the Lower Colorado River Basin

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ABSTRACT

Anthropogenic disturbances impact freshwater biota but are rarely incorporated into conservation planning due to the difficulties in quantifying threats. There is currently no widely accepted method to quantify disturbances, and determining how to measure threats to upstream catchments using disturbance metrics can be time consuming and subjective. We compared four watershed-scale ecological threat indices for the Lower Colorado River Basin (LCRB) using landscape-level threats of land use (e.g., agricultural and urban lands), waterway development and diversions (e.g., number of canals, dams), and human development (e.g., road and railroads density, pollution sites). The LCRB is an ideal region to assess ecological threat indices because of the increasing need for conservation to ensure the persistence of native fishes in highly altered habitat. Each threat was measured for severity (i.e., level of influence on the upstream watershed) and frequency throughout each watershed; both severity and frequency were measured using two different methods. Severity values were based either on peer-reviewed literature and weighted in accordance to their published ecological impact, or assumed equal severity across stressors. Threat frequency was calculated according to either the presence/absence of each stressor, or on the relative density of each stressor in the watershed. Each measure of severity was combined with a measure of frequency, creating four ecological threat indices, and transformed to a 0–100 scale. Threat indices were highly correlated (slopes of 0.94–1.63; R^2 of 0.82–0.98), and were highest for watersheds close to urban centers, including Phoenix, Tucson, and Flagstaff, Arizona, and Las Vegas, Nevada. Road crossings and density appeared to be the most influential stressors in the index, but the removal of any individual stressor only changed the index by <5.1 units. Our results indicate that a simpler index with less subjectivity (i.e., presence/absence of a stressor in a watershed) provides similar results to the more subjective measure of threats (i.e., peer-reviewed threat severity). Because these threats have been linked to ecological health, the development of the index should be a useful tool to identify regions of greatest potential threat to aquatic biota and can aid in conservation planning for the Lower Colorado River Basin.

1. Introduction

By integrating the upstream activities of humans across the landscape, freshwater ecosystems are subjected to the full panoply of anthropogenic threats. Water needs for consumption and energy production, increasing agricultural and urban land use, climatic warming, over-fishing, and invasive species, collectively endanger freshwater biodiversity at levels greater than that for terrestrial and marine systems (Richter et al., 1997; Allan et al., 2005; Naiman et al., 2008; Palmer et al., 2008). An estimated 68% of all mussel species, 51% of crayfish species, 40% of amphibian species, and 46% of fish species in the fresh waters of the United States are considered vulnerable, imperiled, critically imperiled, or presumed extinct (Master et al., 1998; Jelks et al., 2008).

Intact freshwater systems are becoming increasingly rare and many require protection from a range of threats (Abell et al., 2007). Past efforts to conserve freshwater ecosystems and their species have largely focused on restoring patterns and rates of natural physical, chemical, and biological processes (Roni et al., 2008). More recently, there has been increasing emphasis to integrate

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landscape-level anthropogenic threats (e.g., dams, land use) into freshwater conservation planning because these systems are heavily influenced by development within their watershed. There is a growing consensus among scientists that the restoration of ecological processes and reduction of anthropogenic threats are both vital to successful freshwater conservation (Groves, 2003).

Human disturbances affect freshwater systems through multiple processes across a hierarchy of spatial and temporal scales (Stewart et al., 2001; Wang et al., 2003). However, numerous studies have revealed strong relationships between the physical landscape and key biological responses in lotic ecosystems (Roth et al., 1996; Allan et al., 1997). There are a number of challenges to incorporating watershed disturbance in conservation planning, such as the lack of precise knowledge on how stressors influence freshwater resources and how these interactions vary depending on the particular riverine landscape. With continued interest in watershed planning and increased availability of spatial datasets that has been linked to biotic integrity (e.g., Walters et al., 2009) there is promise for greater efficiency of freshwater conservation planning efforts in the future.

Ecological risk assessment represents an important component of conservation planning because it provides a means to balance and compare ecological risks associated with environmental hazards (i.e., hydrologic alteration, land use change, water quality; Graham et al., 1991). Managers can use threat indices to recommend management options for terrestrial and freshwater protection (Abell, 2002; Abell et al., 2007). Recent studies have employed a number of methods to quantify anthropogenic threats in freshwater ecosystems, such as the ecological risk index (Mattson and Angermeier, 2007), the human-threat index (Sowa et al., 2007), and a disturbance index by weighting stressors (Wang et al., 2008). These methodologies proposed threat indices using somewhat subjective scoring criteria of each stressor based on published studies, expert opinion, and best guesses. For example, previous studies have ranked each threat as high, medium, or low relative to its perceived effect on descriptors of biotic integrity (e.g., Mattson and Angermeier, 2007) and ranked frequency scores for the density of the threats (e.g., 26–50% agriculture was ranked a 2 on scale of 1–4 by Sowa et al., 2007). Unfortunately, subjectivity in risk assessment only introduces greater uncertainty in the process of conservation planning.

In light of limited resources for conservation planners to conduct regional risk assessments there is a need to assess the sensitivity of threat indices to the method used to rank the threats. Efforts to quantify anthropogenic threats and develop threat indices for freshwater management are relatively new and warrant questioning. The primary objectives of our study are (1) to develop a suite of ecological threat indices that quantify landscape-level risk from anthropogenic factors in the Lower Colorado River Basin (LCRB) – considered the lifeline of the American southwest and one of the most critically imperiled global habitats (Olson and Dinerstein, 1998), and (2) conduct a comparative analysis of the threat indices to assess whether, and if so how, the different methods affect perceived levels of ecological risk. We aim to determine if more time and cost effective methods that are simpler to compute produce similar results compared to more time consuming and subjective measurements of risk.

2. Materials and methods

2.1. Study site

The LCRB is an ideal model system for the development of an ecological risk index because it exemplifies the growing conflicts between human and ecosystem needs for fresh water. The LCRB provides water for human consumption, agriculture, hydroelectricity, and recreation to over 30 million people in the southwestern United States and northwestern Mexico (Minckley and Marsh, 2009), while supporting a highly endemic but drastically declining fish fauna (Minckley and Deacon, 1968; Olden and Poff, 2005). The southwestern United States is the fastest growing region in the nation, with a human population growth rate of 20.7% in Arizona between 1990 and 2000 (U.S. Census Bureau, 2001).

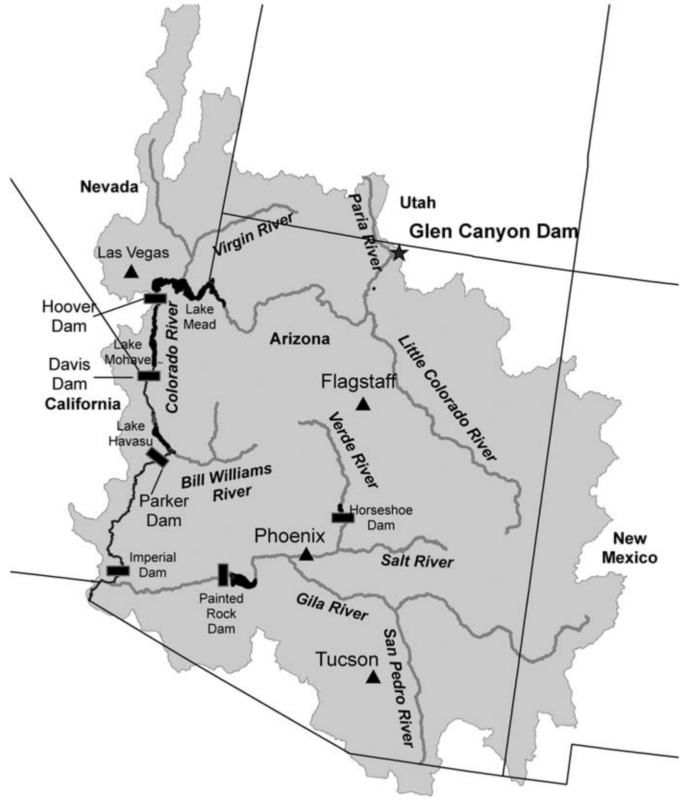

Fig. 1. The Lower Colorado River Basin (LCRB) with major rivers depicted in grey, major cities denoted with black triangles and state boundaries delineated by black lines (the southeast boundaries of Nevada and California follow the Colorado River). Glen Canyon Dam is depicted as a star to highlight our use of this barrier as the upper end of the LCRB. Other major dams are depicted with a cross and the larger reservoirs are displayed in black.

The LCRB drains approximately 350,000 km^2 within the southwestern USA and Sonora, Mexico (Fig. 1). Beginning below the confluence of Paria River in northeastern Arizona, the political boundary of the LCRB includes all tributaries flowing into the Colorado River thereafter, encompassing 26,000 km of stream (Blinn and Poff, 2005). For this study we used a more biologically relevant boundary and included all upstream catchments entering the Colorado River below Glen Canyon Dam (Fig. 1).

2.2. Stressor data

Anthropogenic stressors were selected based on their published impact on freshwater systems and availability of spatial data for the LCRB. These stressors were used to develop a suite of ecological threat indices. Stressors analyzed included metrics of human disturbance related to water diversion and use (canals and dams > 2 m in height), diversions (including rights and claims under USA public water codes), indicators of landscape fragmentation (roads, railroads, stream crossings), land use (urban and agricultural), and possible pollution sources (mines, non-point discharge elimination system permitted sites [NPDES], waste facilities [i.e., Superfund sites (ranked by the US Environmental Protection Agency [EPA] as

Table 1
Anthropogenic stressors, sources, and scale of spatial data used to derive ecological threat indices for the Lower Colorado River Basin. The source year for the dataset is provided in parentheses.

Stressors	Data source	Scale
Canals	US Geological Survey, National Hydrography Dataset (2005)	1:100,000
Dams	US Army Corp of Engineers, National Inventory of Dams (2000)	1:250,000
Diversions	California State Water Resources Control Board (2007), Nevada Division of Water Resources (2006), Utah Division of Water Rights (2004), Arizona Department of Water Resources (2000), New Mexico Office of the State Engineer (2007)	1:100,000
303d-listed	Impaired stream classification developed by Environmental Protection Agency, Water Quality Standards Database (2002)	1:100,000
Urban and Agriculture Landcover	National Landcover Database (2000)	30 m
Mines	US Geological Survey, Mineral Resources Database (2005)	1:1000
Non-point Discharge Elimination System	Environmental Protection Agency, Permit Compliance System (2006)	
Railroads	US Census Bureau, Tiger files (2006)	1:100,000
Roads	US Census Bureau, Tiger files (2006)	1:100,000
Stream crossings	US Census Bureau, Tiger files (2006)	1:100,000
Waste facilities	Environmental Protection Agency, Superfund (2006), Toxic Release Inventory (TRI; 2006) and – Resource Conservation and Recovery Act, hazardous waste sites (RCRA; 2006) databases	1:100,000

the most contaminated sites in the US), toxic release inventory sites and hazardous waste facilities]), and EPA-designated 303d impaired stream classification. Data were collected primarily from state and federal agencies (Table 1) and summarized for the land area upstream of the outflow for all stream segments, referred to as the watershed scale from herein. This resulted in nested watersheds with downstream watersheds accumulating the stressors that occurred in upstream watersheds (i.e., stressor data from all the entire upstream area was included so the watershed that included the terminus of the Colorado River included the entire LCRB). We recognize that freshwater ecosystems are intimately embedded in a matrix of human use, and therefore threats associated with a downstream watershed were not independent of threats associated with upstream watersheds. For each watershed, stressor data was converted to a density value. Point stressor (i.e., mines, NPDES sites, waste facilities, diversions, and stream crossings) density was calculated as number of stressors per square km. For linear stressors (i.e., roads, railroads, canals, and 303d streams), density was calculated as length (m) of the stressor per square km. Land cover (i.e., urban and agriculture) density was calculated as a percentage of total land cover. Dam storage area density was calculated as total storage volume (m^3) per square km. Methods used to quantify threat were based upon published threat indices (Mattson and Angermeier, 2007; Sowa et al., 2007) and were created using a two-tier framework: calculating stressor frequency and stressor severity (see below).

2.2.1. *Stressor frequency*

The first tier of the hierarchical framework included two methods of quantifying stressor frequency. First, we used binary scores representing presence/absence of a stressor to calculate stressor frequency. Second, we assigned scores representing relative den-

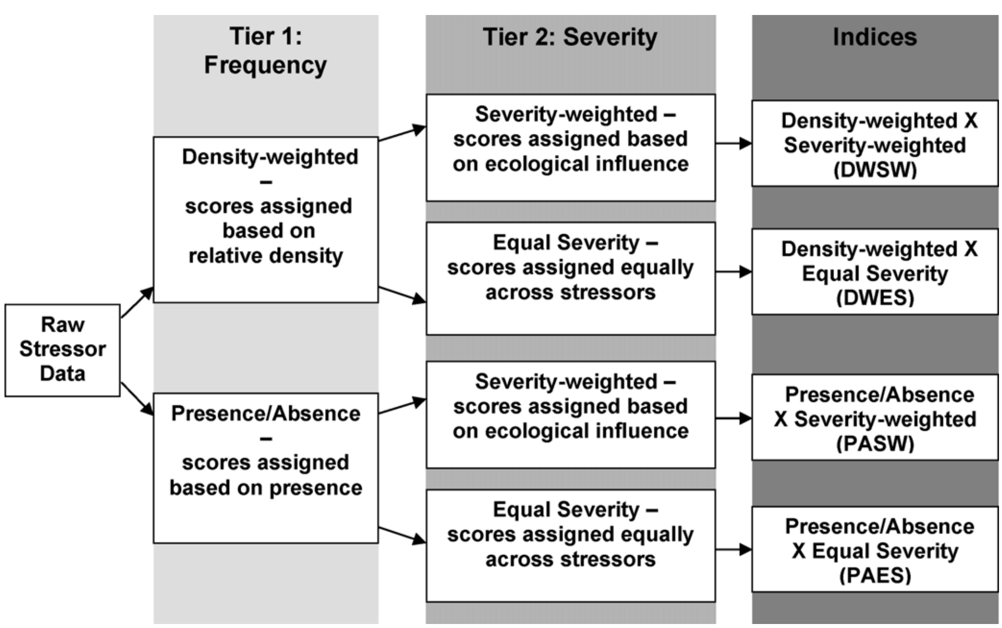

Fig. 2. Schematic illustrating the calculation of the four ecological threat indices created within the Lower Colorado River Basin. Threat values are the product of severity scores and frequency scores. Each index uses different methods for quantifying risk from the same raw spatial data.

Table 2
Summary of the mean and range of stressor densities and rankings of density-weighted frequency scores used to calculate the density-weighted ecological risk indices for watersheds in the Lower Colorado River Basin. The ranking methods were equal quartile except where literature-based ranks were used (agriculture: Wang et al., 1997; Allan, 2004; urban: Wang et al., 2000; Allan, 2004; Wheeler et al., 2005).

Stressors (units of measure)	Mean (range)	Density-weighted frequency scores				
		0	1	2	3	4
Agriculture (%)	0.3 (0–93)	0	>0–10	11–25	26–50	>50
Canals (m/km^2)	3.73 (0–14,326)	0	>0–2	3–4	5–21	>21
Dam storage (m^3/km^2)	12.05 (0–68,717)	0	>0–1	2–5	6–19	>19
Diversions (no./km^2)	0.19 (0–228)	0	>0–0.09	0.10–0.22	0.23–0.47	>0.47
Mines (no./km^2)	0.05 (0–32.9)	0	>0–0.018	0.019–0.05	0.06–0.16	>0.16
NPDES[a] (no./km^2)	0.0006 (0–2.28)	0	>0–0.0005	0.0006–0.0008	0.0009–0.003	>0.003
Railroads (m/km^2)	6.51 (0–5355)	0	>0–10	11–26	27–65	>65
Roads (m/km^2)	659 (0–74,431)	0	>0–370	371–651	652–1020	>1020
Stream crossings (no./km^2)	0.33 (0–427)	0	>0–0.18	0.19–0.29	0.30–0.51	>0.51
Urban (%)	0.9 (0–100)	0	>0–1	2–3	4–8	>8
Waste facilities (no./km^2)	0.0004 (0–1.9)	0	>0–0.0001	0.0002–0.0005	0.0006–0.002	>0.002
303d-listed (m/km^2)	13 (0–30,018)	0	>0–3	4–17	18–70	>70

[a] NPDES: Non-point Discharge Elimination System.

sity of the stressor (Fig. 2). All watersheds having a density of zero for a particular stressor would receive a score of zero for frequency of that stressor. With the exception of land cover, remaining watersheds were ranked based on four equal quartiles of the density of the stressor (following Mattson and Angermeier, 2007). Watersheds with high relative densities receive higher frequency scores whereas watersheds with relatively low densities receive lower scores (Fig. 3). Density-based frequency scores for urban and agricultural land cover (see Table 2) were based on published literature relating stressor density to freshwater ecosystem health (Wang et al., 1997, 2000; Allan, 2004; Wheeler et al., 2005).

2.2.2. *Stressor severity*

The second tier of the hierarchical framework included two methods of quantifying stressor severity (Fig. 2). First, all stressors were weighted according to their potential impact on variables of ecological integrity (i.e., habitat quality, water quality, biotic interactions, energy, and flow regime; Karr, 1991). Stressor severity weights were assigned using the scoring system in Mattson and Angermeier (2007). Each stressor was scored between zero and three for each measure of ecological integrity; a score of zero suggested no influence on ecological integrity and a score of three suggested major influence on ecological integrity (Table 3). The scores were then summed across all ecological integrity variables to produce one cumulative severity score for each stressor. Second, we assigned equivalent weights across stressors suggesting that all environmental factors have equal impact.

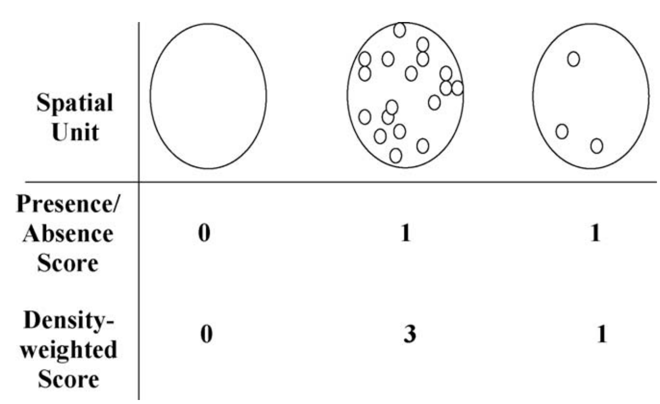

Fig. 3. Two methods for quantifying frequency scores. The large circles represent the spatial unit of analysis (i.e., watershed) while the small circles represent a single point stressor (e.g., mine). Presence/absence scores are based on presence of a stressor while density-weighted frequency scores are based on equal quartiles of the density of the stressor.

These two tiers were used to calculate an index of anthropogenic threats. Frequency scores (tier 1) were multiplied by severity scores (tier 2) to generate a threat value for each stressor and then summed across stressors to calculate an overall threat value for each watershed. We calculated four indices representing different approaches to quantify risk using the same raw stressor data, including: (1) density-weighted scores × severity-weighted scores (DWSW); (2) density-weighted scores × equal severity scores (DWES); (3) presence/absence scores × severity-weighted scores (PASW); and (4) presence/absence scores × equal severity scores (PAES) (Fig. 2).

2.3. *Statistical analysis*

Each threat index was transformed to range from 0 to 100 (scores were divided by the maximum score within the four approaches so the highest score for each approach was 100) to facilitate comparisons among indices. General linear regression was used to analyze the relationships between watershed threat values based on the different indices. All plots indicated a linear relationship between the normal quantiles and the residuals (normal probability plots) and thus no deviation from the assumption of normality. Next, we conducted a sensitivity analysis to determine the influence of each stressor on the overall index values. The difference between the standardized index values with all stressors included and the standardized index values with each individual stressor removed was computed for each of the four methods. This analysis determined how sensitive the overall index was to individual stressors (the change in stressor index score after an individual stressor [e.g., roads, stream crossings, etc.] was removed). Pearson correlation analysis was used to determine if the change in the index after each individual stressor ($N = 12$; one value for each of the 12 stressors) was removed was similar across all four indices.

3. Results

We developed threat scores using the four methods for the connected network of stream segments in the LCRB (73,078 stream segments). Watershed size ranged from headwater segments (<1 km^2) to the entire LCRB (345,278 km^2) with a mean of 2088 km^2. Anthropogenic stressors varied greatly throughout the LCRB (Table 2). Agricultural land use ranged from 0 to 93% and averaged 0.3% while urban land use ranged from 0 to 100% with a mean of 0.9% among watersheds. However, <0.1% of watersheds were >8% urban. Dam storage volume ranged from 0 to 68,717 m^3/km^2 with an average of 12.05 m^3/km^2. Road density ranged from 0 to 74,431 m/km^2 with an average of 659 m/km^2 while impaired

Table 3
Summary of weighted severity scores by the ecological integrity categories (Karr, 1991) based upon Mattson and Angermeier (2007). A score of zero suggests no influence whereas a score of 3 suggests severe influence on the variable of ecological integrity. Values are summed across ecological integrity variables to produce the weighted severity score. Units of measure and abbreviations were provided in Table 2.

Stressors	Severity weightings					
	Water quality	Habitat quality	Biotic interactions	Flow regime	Energy source	Total severity
Agriculture	3	3	1	2	3	12
Canals	2	3	2	3	2	12
Dam storage	3	3	3	3	3	15
Diversions	0	1	0	3	0	4
Mines	3	2	1	1	1	8
NPDES	3	1	1	2	3	10
Railroads	2	2	0	0	1	5
Roads	2	2	0	1	2	7
Stream crossings	2	2	1	1	1	7
Urban	3	3	1	2	3	12
Waste facilities	3	2	1	0	2	8
303d-listed	3	0	1	0	1	5

(303d) stream reach density ranged from 0 to 30,018 m/km^2 with an average of 13 m/km^2.

The mean standardized threat values for all watersheds across the four threat indices ranged from 13 to 26. On average, watershed values of the PAES index were the greatest (mean = 26.1) with a range from 0 to 100, followed by the PASW index (mean = 23.4, range 0–100). Watershed values of DWES ranged from 0 to 85 and had a mean of 15.1, whereas DWSW values ranged from 0 to 88 with a mean of 13.1 (Fig. 4). The threat values for each method were highly correlated (slopes of 0.94–1.63; R^2 of 0.82–0.98; Table 4).

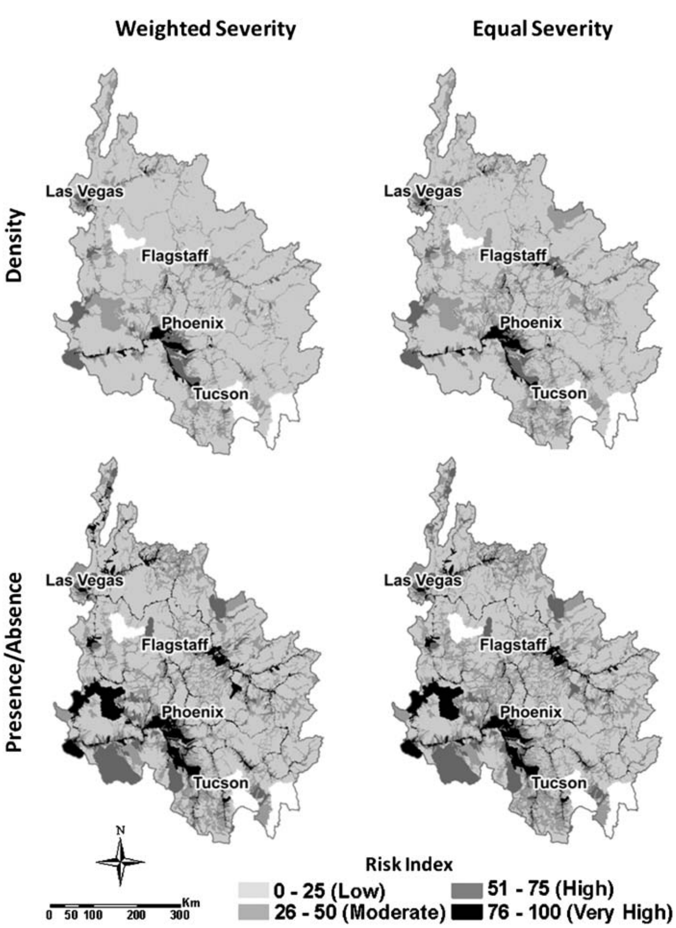

Fig. 4. Ecological threat values at the watershed scale of the Lower Colorado River Basin. Index methods are noted on left-hand side and top of maps. Threat values were calculated based on the upstream watershed but presented at the focal stream segment for clarity.

Threats indices were typically highest near urban centers (Fig. 4). Watersheds near Phoenix, Tucson, and Flagstaff, Arizona USA, and Las Vegas, Nevada USA typically had the highest threat scores. Eighty-one percent of watersheds with scores of 76–100 were within 60 km downstream of the cities of Phoenix, Tucson, and Las Vegas. Watersheds not associated with urban areas but having high threat scores (extreme northcentral and southwestern part of the LCRB; Fig. 4) had high road densities (1157 and 709 m/km^2, respectively) thus increasing their overall threat score. By contrast, much of northern and western portions of the basin, which are typically rural, had low threat scores.

Sensitivity analyses showed that none of the ecological threat indices were dramatically sensitive to any one stressor (Table 5). The mean change in values for all ecological threat indices was greatest when roads were removed from the assessment (mean = 2.8–5.1; SD = 1.9–3.0), followed by stream crossings regardless of index type (mean = 2.4–4.3; SD = 2.4–3.0). Ecological threat indices changed the least when railroads and impaired (303d) stream reaches were removed (mean = 0.8–1.8; SD = 0.8–1.6; mean = 0.6–1.9; SD = 0.8–1.6, respectively). Overall, the change in index values were relatively low; the mean change was <2 for 58% (28 of 48) of the calculations and >5 for only 2% (1 of 48) of the calculations (Table 5). Changes in the ecological threats were correlated with one another, suggesting that the removal of an individual stressor had similar results among methods. The changes in values for the PAES index were strongly related to values for the PASW (r = 0.63, P = 0.03), DWES (r = 0.97, P < 0.001), and DWSW index (r = 0.74, P = 0.006). The changes in the PASW index were also highly correlated with changes in the DWSW (r = 0.95, P < 0.001), but was not strongly related to changes in the DWES index (r = 0.50, P = 0.10). Changes in DWES index were also related to changes in the DWSW index (r = 0.66, P = 0.02).

Table 4
Linear regression statistics for comparing four different methods of threat index development. A slope of 1.0 indicates indices produce similar threat values. Higher R^2 values indicate the reliability of a linear relationship between two indices. Standard error of slope (SE) is a variability measure; the smaller the SE, the less variable the relationship is between the indices. All tests were significant (P < 0.0001). DWSW: density-weighted scores × severity-weighted frequency scores; DWES: density-weighted scores × equal severity scores; PASW: presence/absence scores × severity-weighted scores; PAES: presence/absence scores × equal severity scores.

Index test	Slope (SE)	R^2
PASW = PAES	1.00 (0.0005)	0.98
DWSW = DWES	0.94 (0.0006)	0.97
PAES = DWES	1.53 (0.0027)	0.82
PASW = DWSW	1.63 (0.0027)	0.84

Table 5
Mean change in threat index after stressor was removed for the four methods to calculate a watershed risk index for the Lower Colorado River Basin. Stressors are ordered in mean change in risk across the four indices. DWSW: density-weighted scores × severity-weighted frequency scores; DWES: density-weighted scores × equal severity scores; PASW: presence/absence scores × severity-weighted scores; PAES: presence/absence scores × equal severity scores. Units of measure and abbreviations were provided in Table 2.

Stressor removed	Change in index when stressor is removed							
	PAES		PASW		DWES		DWSW	
	Mean (SD)	Range	Mean (SD)	Range	Mean (SD)	Range	Mean (SD)	Range
Roads	5.1(3.0)	0–8	4.1(2.4)	0–7	3.2(1.9)	0–8	2.8(2.1)	0–11
Stream crossings	4.3(3.0)	0–8	3.4(2.7)	0–6	2.7(2.7)	0–8	2.4(2.4)	0–12
Urban	2.5(2.4)	0–8	3.4(3.2)	0–11	1.3(1.4)	0–8	1.7(2.1)	0–15
Dam storage	1.8(1.4)	0–8	3.2(2.4)	0–14	1.3(1.1)	0–8	2.1(2.1)	0–21
Mines	2.2(2.1)	0–8	2.1(2.1)	0–8	1.7(1.9)	0–8	1.4(1.9)	0–8
Diversions	2.9(2.7)	0–8	1.4(1.1)	0–4	2.0(2.1)	0–8	1.1(1.6)	0–10
Agriculture	2.0(1.9)	0–8	2.7(2.4)	0–11	1.1(0.8)	0–8	1.5(1.9)	0–19
Canals	1.8(1.6)	0–8	2.5(2.1)	0–11	1.3(1.1)	0–8	1.7(1.9)	0–18
NPDES	1.8(1.6)	0–8	2.0(1.6)	0–10	1.3(1.1)	0–8	1.3(1.4)	0–10
Waste facilities	1.9(1.6)	0–8	1.6(1.6)	0–8	1.3(1.1)	0–8	1.0(1.4)	0–8
303d-listed	1.9(1.6)	0–8	1.1(1.1)	0–5	1.4(1.4)	0–8	0.6(0.8)	0–5
Railroads	1.8(1.6)	0–8	1.0(0.8)	0–5	1.4(1.1)	0–8	0.8(1.4)	0–12

4. Discussion

Threat scores were typically highest in watersheds located in close downstream proximity to urban centers. This is in line with the observation that watersheds with even just 10% urban land or impervious surface are associated with reduced ecological health (Wang et al., 2000; Allan, 2004; Utz et al., 2009). Although the mean percent of urban land was <1% in the LCRB, watersheds with at least 8% urban land were primarily located within the four largest cities in the basin, suggesting these urban centers have a greater risk to aquatic biota than the more rural areas, which is similar to other regions (Angradi et al., 2009).

All ecological threat indices were most sensitive to the inclusion of stressors describing density of roads and stream crossings, suggesting that these variables play the largest role in defining watershed risk. Road density is a useful indicator for land use change (Trombulak and Frissell, 2000) and therefore our threat assessment could be used for detecting watershed-scale changes in land use that would impact the aquatic biota. Many of the metrics used in our assessment were linked to urbanization (i.e., road and railroad densities, stream crossing density, canals). Increased urbanization and canalization has been linked to greater spread of invasive fishes (Cowley et al., 2007) and reduced fish diversity, abundance, and biotic integrity in the LCRB (Minckley and Deacon, 1968; Mueller and Marsh, 2002; Rinne and Miller, 2006), suggesting that our risk index may be a useful tool to determine where landscape-level disturbance may most affect aquatic biota. Aquatic biota within watersheds with increased road density and crossings are more susceptible to contamination caused by runoff (Wheeler et al., 2005), fragmentation of streams that may limit fish migrations (Fagan et al., 2005; Bouska and Paukert, 2010) and reduce genetic diversity, and increases the risk of non-native fish introductions (Wheeler et al., 2005). Our results suggest that areas near the urban centers in the LCRB have the greatest ecological risk, which were also areas with low native fish diversity. By contrast, watersheds with low risk scores were typically in areas with greatest native fish diversity in the LCRB (Olden and Poff, 2005; Olden, unpublished data). Because this region has one of the fastest growing human populations in the USA, risk scores of suburban areas are likely to increase with greater urbanization and other forms of land use change. Therefore, prioritizing areas adjacent to urban centers for watershed protection may be needed in the future.

5. Conclusions

Our results suggest that all four methodologies for calculating ecological threat produced similar index values regardless of the approach used to estimate threat frequency or severity. The high correlation between all indices suggests that the decision of how the frequency and severity of different stressors are tabulated may be less important in the conservation planning process. Despite this, it is important to recognize that our investigation may have overshadowed differences that are observed at smaller spatial scales due to the large sample size of our analysis (>70,000 watersheds) and lack of independence among watersheds which was required to properly quantify risk in a riverscape setting. Future research could evaluate the robustness of our findings across regional and spatial scales.

Previous approaches to ecological threat analysis vary in both complexity and subjectivity. Unless known or estimated thresholds of ecological importance have been published, the onus is on the researcher to define threshold values based on best available knowledge or assume equal contributions. For example, Mattson and Angermeier (2007) used equal intervals of stressor densities to define frequency scores whereas Sowa et al. (2007) based their calculations on quartiles of stressor densities. While there are advantages and disadvantages to each method, it is unknown whether one method is more ecologically relevant and thus more appropriate for conservation planning. Other studies have used expert opinion to assess the impacts of anthropogenic threats (Richter et al., 1997; Halpern et al., 2007). These methods of assessment are often time consuming and our results suggest that the simple and less-subjective methods provide similar results.

Our study provides a framework to assess landscape-level threats on aquatic biota and aid future research on identifying the causal relationships and mechanistic responses to biota from these threats. Additional research exploring the relationships between individual stressors, threat indices, and biological responses will help to refine regional-scale conservation planning efforts. Our assessment provides a useful examination of how to calculate a watershed-based index that can be used and modified for different stressors identified in different systems or regions.

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