ASSESSING THREATS TO NATIVE FISHES OF THE LOWER COLORADO RIVER BASIN

by

KRISTEN LEAH PITTS

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Dr. Craig P. Paukert

ABSTRACT

We investigated the influence of anthropogenic threats and hydrologic alteration on fish assemblages within the Lower Colorado River Basin (LCRB). Life history traits of fish assemblages for individual stream segments were summarized by presence/absence data of current (1980-2006) records. To assess anthropogenic threats, we developed a series of ecological risk indices at various scales (e.g., catchment, watershed, aquatic ecological system and upstream of aquatic ecological system) and related each index to fish life-history traits to determine the method and scale that best relates to biotic metrics. Hydrologic alteration was quantified using the Indicators of Hydrologic Alteration (IHA) software to calculate hydrologic alteration values using the range of variability approach (RVA). Ecological risk indices within all scales were strongly correlated (r^2>0.54, p<0.0001) to one another. Relationships between fish life history traits and ecological risk indices occurred only at the catchment and watershed scales. Strongest relationships were at the watershed scale where increased levels of anthropogenic risk were related to reduced occurrences of native, fluvial dependent species (r^2=0.12, p<0.0001) and increased occurrences of nonnative generalist species (r^2=0.22, p<0.0001). The percent agriculture was positively related to indices of alteration of low flows (r=0.401, p=0.006) while forested land cover was negatively related to alteration of low flow events (r=-0.384, p=0.008). Relationships between indices of hydrologic alteration and fish traits indicate the occurrence of piscivorous, nonnative fishes increased with alteration of low flow events whereas occurrence of fluvial dependent fishes that preferred rubble substrate decreased with alteration of low flow events (r=0.64, p=0.001). Our analysis suggests that ecological risk indices and hydrologic alteration in the LCRB are related to composition of biotic

communities. Incorporating cost-effective risk indices into conservation planning will likely increase the effectiveness of conservation efforts while understanding biotic responses to modified flow regimes are a necessity in sustainable development of water resources as human populations grow and water resources decrease in the LCRB.

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Chapter 1 - Assessing threats to native fishes of the Lower Colorado River Basin: development and assessment of ecological risk indices

Abstract

Anthropogenic disturbances often influence biotic communities but are rarely incorporated into conservation planning due to the difficulty in quantifying associated risk to biota. We developed a series of ecological risk indices at various scales (e.g., catchment, watershed, aquatic ecological system and upstream of aquatic ecological system) for the Lower Colorado River Basin and related each index to fish life-history traits to determine the method and scale that best relates to biotic metrics. Four different ecological risk indices were developed using severity and density weightings of individual stressors (e.g., agricultural land use, canals, dams, etc.). Ecological risk indices within all scales were strongly correlated ($r^2>0.54$, $p<0.0001$) to one another. Relationships between fish life history traits and ecological risk indices occurred only at the catchment and watershed scales. Strongest relationships were at the watershed scale where increased levels of anthropogenic risk were related to reduced occurrence of native, fluvial dependent species ($r^2=0.12$, $p<0.0001$) and increased occurrence of nonnative generalist species ($r^2=0.22$, $p<0.0001$). Our analysis suggests that ecological risk indices are related to biotic communities (fish traits) and therefore are applicable for prioritizing areas for native fish conservation, but the spatial scale of risk index development is important. Incorporating cost-effective risk indices into conservation planning will likely increase the effectiveness of conservation efforts.

Introduction

It is estimated that 68% of all freshwater mussel species, 51% of crayfish species, 40% of amphibian species, and 39% of freshwater fish species in the United States are considered vulnerable, imperiled, critically imperiled, or presumed extinct (Master et al. 1998). Efforts to manage and conserve these species often focus on maintaining the natural physical, chemical, and biological processes within ecosystems (Abell et al. 2000). Although well-intentioned, conservation planning efforts rarely integrate landscape-level anthropogenic threats that may significantly alter the ecosystem (Mattson and Angermeier 2007). While restoration of ecological processes is vital in successful freshwater conservation, conservation priorities will likely prove to be more effective by recognizing and assessing the role of anthropogenic stressors in an ecosystem (Cowx 2002; Groves 2003).

Rivers naturally collect surface water from surrounding land and therefore incorporate landscape influences at multiple spatial scales (Allan 2004). Numerous landscape influences have been implicated as sources of stress to biotic assemblages throughout the U.S. Among these, agricultural land use, municipal land use, exotic species, impoundments, land use change, channelization and hydropower generation have greatly contributed to the imperilment of aquatic biota (Richter et al 1997; Wilcove et al. 1998; Cowx and Collares-Pereira 2002).

There have been increased efforts to quantify risk associated with stressors because of the increased availability of large-scale datasets and the ability to delineate stream reaches and associated catchments. The ecological risk index (ERI; Mattson and Angermeier 2007) combines risk-based components (i.e. frequency and severity of stress sources) with biotic drivers to produce relative risks for watersheds. The ERI weights individual stressors by their severity, or impact on ecological integrity (i.e., flow regime, physical habitat, water quality, energy sources, and biotic interactions; Karr 1991) and frequency. The human-threat index (HTI), created by Sowa et al. (2007), generated relative rankings for 11 uncorrelated measures of human disturbance to produce a composite three-digit number reflecting individual and cumulative disturbances. Wang and others (2008) developed a disturbance index by first assessing influence of disturbances on fish assemblages and then using that influence as weighting factors to produce overall disturbance values. These efforts suggest that threat assessments may be useful tools for aquatic conservation.

These indices have been created as relatively cost-effective tools for conservation management, but have not been tested or validated to determine the fish community response to stressors (but see Wang et al. 2008). Because one measure of an index's utility is its response to various metrics, there is a need to evaluate the risk indices to determine if there are relationships with measures of ecological health. The Lower Colorado River Basin (LCRB) is an ideal region to test risk indices because of the large database of fish records and the increasing need for conservation measures to ensure the persistence of native fishes in the midst of highly altered fish communities (Rahel 2000; Mueller and Marsh 2002). The large area of the LCRB and the extensive fish records for the region should allow an evaluation of whether risk indices are an effective tool in targeting high and/or low risk locations for native fish species. If effective, risk indices would allow locations within the LCRB to be assessed for conservation practicality, based on current risk. Application of an effective risk index would also allow conservation managers, planners and policy makers to assess the current set of threats and native fish presence while considering future threat projections.

The first objective of this study is to develop a suite of ecological risk indices which quantify risk from anthropogenic threats at various spatial scales. This objective involves developing methods of transferring raw spatial data into risk values. The second objective is to compare results of these various risk indices to one another. This objective seeks to qualitatively disclose strengths and weaknesses of each method and scale. The final objective is to assess the usefulness of risk indices by relating risk to current fish distributions.

Methods

Site description

The LCRB drains 362,750 km² within the state boundaries the Arizona, California, New Mexico, Utah and Nevada, USA (Figure 1; Blinn and Poff 2005). Beginning below the confluence of Paria River in northeastern Arizona, the LCRB includes all tributaries flowing into the Colorado River thereafter, encompassing 26,000 km of stream (Blinn and Poff 2005; Olden and Poff 2005). Major tributaries to the Lower Colorado River include the Gila, Virgin, Bill Williams and Little Colorado rivers (Figure 1.1).

Stressor data

Anthropogenic stressors were selected based on their known influence on aquatic species assemblages and availability of spatial data. Stressors included canals, dams (>2 m high) , roads, railroads, stream crossings, diversions (including rights and claims under public water codes), urban and agricultural land use, mines, non-point discharge elimination system permitted sites (NPDES), waste facilities (i.e., Superfund sites, toxic release inventory sites and hazardous waste facilities), and EPA-sanctioned 303d impaired stream classifications. Spatial stressor data were collected primarily from state and federal agencies (Table 1.1).

Stressor data were summarized at four spatial scales: 1) catchment; 2) watershed; 3) aquatic ecological system (AES); and 4) upstream of the AES (AES_{UP}). Catchment scale included all land draining into an individual stream segment. Watershed scale included all land draining into upstream reaches of an individual stream segment (Figure 1.2). Aquatic ecological systems are broad-scale regions reflecting distinct biological communities and are delineated using abiotic factors within a zoogeographic context (Figure 1.3; Higgins et al. 2005; Sowa et al. 2007; Whittier et al. 2008). The upstream of AES scale includes all land draining into upstream reaches of AES boundaries.

Ecological Risk Index Methodology

Methods used to quantify risk indices were based upon published risk indices (Mattson and Angermeier 2007; Sowa et al. 2007) and were created using a two tier hierarchical framework (see below). First, spatial stressor data were converted to density at each scale. Point stressor (i.e., mines, NPDES sites, waste facilities, diversions, and stream crossings) density was calculated as number of stressors per square km of the spatial unit (i.e., catchment). For linear stressors (i.e., roads, railroads, canals, and 303d streams), density was calculated as length (m) of the stressor per square km of the spatial unit. Land cover (i.e., urban and agriculture) density was calculated as square km of the land cover category per square km of the spatial unit. Dam density was calculated as total storage area (square m) per square km of the spatial unit.

The first tier of the hierarchical framework included two classes of severity; 1) weighted scores based on ecological impact of stressors and 2) all stressors having equal severity (Figure 1.4). For the ecological impact method of calculating severity, all stressors were weighted, using peer-reviewed literature, on their potential impact to the various aspects of ecological integrity (i.e., habitat quality, water quality, biotic interactions, energy, and flow regime; Karr 1991; Table 1.2). Each measure of ecological integrity was scored between 0-3; with a score of zero suggesting little influence and a score of three suggesting major influence (Mattson and

Angermeier 2007). The scores were then summed across all ecological integrity variables to produce one severity score for each stressor. The equal severity method gave each stressor equal weighting which would suggest all stressors have equal impact (Table 1.2).

The second tier of the hierarchical framework included two classes of stressor frequency; 1) binary scores representing presence/absence of a stressor and 2) scores representing relative density of the stressor (Figure 1.3). For the presence/absence method, each stressor was determined to be present or absent in the spatial unit of interest (e.g., catchment, AES, etc.). For example, if the spatial unit examined had no mines within its boundaries, it would have a score of zero for frequency of mines (Figure 1.5). However, if the spatial unit had mines within its boundaries, regardless of density, it would have a score of one for frequency of mines. The second method of quantifying frequency accounts for relative density of each stressor. Under this method, all spatial units having a density of zero for a particular stressor have a score of zero frequency of that stressor. With the exception of land cover, remaining spatial units are ranked based on four equal quartiles of the density of the stressor (not including densities of zero). Spatial units with high relative densities receive a higher score for the frequency of the stressor whereas spatial units with relatively low densities receive lower scores (Figure 1.5). Density-based frequency scores for urban and agricultural land cover were based on published literature relating stressor density to aquatic ecosystem health (Table 1.3; Wang et al. 1997; Wang et al. 2000; Allan 2004; Wheeler et al. 2005).

These two tiers were used to calculate an index of anthropogenic threats. Severity scores (tier 1) were multiplied by frequency scores (tier 2) to generate a risk value for each stressor and then summed across stressors to calculate an overall risk value for each spatial unit examined. The four different indices calculated at each scale are: 1) Severity-weighted scores x density-weighted frequency scores; 2) Severity-weighted scores x presence/absence scores; 3) Equal severity scores x density-weighted frequency scores and 4) Equal severity scores x presence/absence scores (Figure 1.4) and represent four different approaches to quantify risk using the same raw stressor data. These four indices were calculated for each spatial scale.

Fish data

We used fish species presence/absence to determine if any of the 16 calculated stressor indices were related to fish assemblage structure. Recent (1980 – 2006) fish collections were obtained from the LCRB Aquatic Gap Analysis Project (GAP) database of over 80,000 fish sampling locations and 1.5 million individual fish records within the LCRB from various sources (e.g., Arizona Game and Fish Department, Arizona State University, US Geological Survey, Utah Heritage Database, US Forest Service, etc.). Data collected included geo-referenced point locations (verified by agency personnel), species name, site description, and date collected. Fish were sampled with various gears (e.g., hoop nets, dip nets, gill nets, minnow traps, trammel nets, seines and electrofishing) but electrofishing and seining accounted for 82% of all stream segments with recorded gear type. Although samples were not collected for the objectives of this study, other studies have indicated how large, historical databases can be effectively utilized for similar purposes as our study (e.g., Fagan et al. 2002; Fagan et al. 2005; Olden and Poff 2005; Olden et al. 2006a).

At the catchment and watershed scales, presence/absence was determined by summarizing the 1980-2006 records for each individual stream segment and AES. Sampling events from hatcheries and ponds were removed from analysis as were fish recorded as re-introduced but not necessarily established spawning populations. Because LCRB encompasses a large area in which species assemblages may differ due to biogeographic constraints, life history traits were used to allow functional comparison of sites throughout the basin (Poff and Allan 1995; Scott and Helfman 2001). By using life history traits instead of species, we can focus more on the broad natural history of the fish assemblages instead of species composition. Life history characteristic data primarily came from Olden et al. (2006a) with additional species data collected from peer-reviewed literature and online databases (e.g., FishBase). There were a total of 70 life history variables per spatial unit (i.e., catchment). All life history variables were categorical (i.e., native species) therefore numerical variables (i.e., spawning temperature) were divided into equal thirds resulting in three categories per variable (i.e., low, medium, high).

Data analysis

Each of the 16 risk indices was standardized to range from 0 (low risk) to 100 (high risk) to compare indices. Frequency histograms of risk values for each index at all scales were compared to assess distribution of risk. Linear regression was used to assess the relationship of risk values between indices. In this analysis, a slope of 1.0 would indicate the two risk indices were very similar.

Principal components analyses (PCA) were used to reduce the dimensionality of the fish life history characteristic data at the various scales. These analyses used each site (e.g., catchment, AES) as an observation. The PCA axes were interpreted based on the component loadings of each life history trait. Variable loadings with absolute values greater than 0.20 on each axis were interpreted. Analysis of covariance was used to test the relationship between risk indices and principal component (PC) scores with stream order (for catchment and watershed scales only) and river basin (refer to Figure 1.1) as covariates. Bonferroni corrections were used to adjust the alpha level for multiple tests within each scale (alpha=0.05/12 or 0.004).

Results

A total of 73,078 stream segments were used in our analysis at the catchment scale (mean area=5.1 km^2), whereas only 36,379 stream segments were analyzed at the watershed scale (mean area=4,180 km^2; i.e., there were fewer stream segments analyzed because headwater streams have no upstream catchments). A total of 386 AES' (mean area=966 km^2) were used at the AES scale, and 197 AES' had associated upstream reaches for the AES_{UP} scale (mean area=38,943 km^2; Table 1.4). Most stressors had similar mean densities among scales (Table 1.4). For example, mean mine density ranged from 0.4 – 0.5 per km^2 and mean diversions ranged from 0.2 – 0.3 per km^2. Stressors with mean densities not consistent across scales included dams and 303d impaired segments. Mean dam storage size ranged from 7.4 – 201.3 m^2/km^2 while mean density of 303d impaired segments ranged from 11.6 – 42.7 km/ km^2.

Catchment scale

Risk values at the catchment scale were typically low throughout the basin with mean values ranging between 7 and 15 (Table 1.5). Presence/absence indices tended to have greater risk values (range: 0-100) than density-weighted indices (range: 0-65). Risk values were similar for the severity-weighted x density-weighted index and the equal severity x density-weighted index. Linear regressions of the risk indices indicated no difference between indices (Table 1.6). However indices with the same frequency scoring method were more strongly correlated. For example, values for the equal severity x presence/absence index were strongly related to values for the severity-weighted x presence/absence index (r^2=0.95, slope=0.92, $p<0.0001$), and values for the equal severity x density-weighted index were strongly related to values for the severity-weighted x density-weighted index (r^2=0.95, slope=1.06, $p<0.0001$). Values for the equal severity x presence/absence index were less related to values for the equal severity x density-weighted index (r^2=0.78, slope=0.61, p<0.0001) as were risk values for the severity-weighted x presence/absence index when related to risk values for the severity-weighted x density-weighted index (r^2=0.80, slope=1.34, p<0.0001).

The greatest risk was spatially concentrated near urban centers (Figure 1.6). The two density-weighted indices showed few high risk areas, primarily near Phoenix, Arizona (Salt and Lower Gila river basins) and Las Vegas, Nevada (mainstem Colorado River Basin). The two presence/absence indices indicated high risk areas near St. George, Utah (Virgin River Basin), Las Vegas, Nevada (mainstem Colorado River Basin), Phoenix, Arizona (Salt and Lower Gila river basins), downstream of Tucson, Arizona (Santa Cruz River Basin) and large sections of the southern half of the LCRB.

There were 1,718 stream segments with fish data used to assess if risk scores were related to fish traits. The first three axes of the fish trait PCA explained 46.6% of the variation among catchments. The first PC axis explained 23.2% and had high loadings of fish considered not fluvial dependent, having small length of larvae and preferring low velocities and silt/mud substrate (Figure 1.7). Fish species that scored high on axis one included fathead minnow (*Pimephales promelas*) and common carp (*Cyprinus carpio*). The second axis explained 12.9% of total variation and had high loadings of fish considered to be native, benthic, herbivorous/detrivorous, having low maximum size, short lifespan, narrow diet breadth, low fecundity, high shape factor and preferring moderate velocities. Fish species that scored high on axis two included woundfin (*Plagopterus argentissimus*), desert sucker (*Catostomus clarkii*) and Sonora sucker (*Catostomus insignis*). The third axis explained 10.5% of variation and had high loadings of fish preferring cold temperatures, rubble substrate, moderate to fast current, having high maximum size, low maturity, low spawning temperature, and being external bearing, pelagic spawners. Fish species that scored high on axis three included rainbow trout (*Oncorhynchus mykiss*), brown trout (*Salmo trutta*), and brook trout (*Salvelinus fontinalis*). Fish species that scored low on axis three included western mosquitofish (*Gambusia affinis*) and topminnow (*Poeciliopsis occidentalis*). Results of this PCA were used for both catchment and watershed scales.

Numerous significant relationships were found between PC scores and risk indices at the catchment scale, however all relationships had very low predictive power based on regression coefficients. Severity-weighted x density-weighted risk values were related with fish trait PCI (r^2=0.0005, p=0.0003; Table 1.5) as was severity-weighted x presence/absence risk values (r^2=0.0013, p=0.001). Severity-weighted x density-weighted risk values and equal severity x density-weighted risk values were related with fish trait PCII (r^2=0.0006, p=0.0002; r^2=0.001, p=0.006; respectively) and PCIII (r^2=0.07, $p<0.0001$; r^2=0.07, $p<0.0001$; respectively; Figure 1.8). Both severity-weighted x presence/absence risk values and equal severity x presence/absence risk values were also correlated with fish trait PCII (r^2=0.003, $p<0.0001$; $r^2<0.001$, $p<0.0001$; respectively) and PCIII (r^2=0.073, $p<0.0001$; r^2=0.078, $p<0.0001$, respectively). Although there were statistically significant relationships with fish PC scores and risk values at the catchment scale, regression coefficients were always <0.08.

Watershed scale

The majority of stream segments had low risk values at the watershed scale with mean risk values ranging from 17 to 36 (Table 1.5). Density-weighted indices typically had lower risk values and narrower value ranges than presence/absence indices. Linear regression indicated similar results as with catchment scale in that there was no difference between indices; however indices with the same frequency method were more strongly related to one another. For example, values for the equal severity x presence/absence index were strongly related to values for the severity-weighted x presence/absence index (r^2=0.98, slope=1.03, p<0.0001; Table 1.6), and risk values for the equal severity x density-weighted index were strongly related to risk values for the severity-weighted x density-weighted index (r^2=0.98, slope=1.0, p<0.0001). Values for the equal severity x presence/absence index were less strongly related to values for the equal severity x density-weighted index (r^2=0.82, slope=0.55, p<0.0001) as were values for the severity-weighted x presence/absence index when related to values for the severity-weighted x density-weighted index (r^2=0.82, slope=1.73, p<0.0001).

All indices indicated greater risk in downstream stream segments (Figure 1.9). The density-weighted indices show risk highest in the Virgin, Lower Gila, Santa Cruz river basins with portions of the Gila, mainstem Colorado and Salt river basins also having high risk watersheds. The presence/absence indices show risk to be widespread throughout larger rivers (Figure 9).

There were 1,519 watersheds with fish data used to assess if risk values were related to fish traits. All four risk indices were significantly related to all three fish trait PC axes and typically had higher predictive power than at the catchment scale. The strongest relationships were with fish trait PCI for the severity-weighted x density-weighted index (r^2=0.13, p<0.0001; Figure 1.10), severity-weighted x presence/absence index (r^2=0.22, p<0.0001), equal severity x density-weighted index (r^2=0.15, p<0.0001) and equal severity x presence/absence index (r^2=0.21, p<0.0001). These relationships indicate that sites with increased risk were more associated with generalist species such as common carp and fathead minnow. Relationships between risk indices and fish trait PCII were statistically significant for severity-weighted x density-weighted values (r^2=0.11, p<0.0001), severity-weighted x presence/absence values

(r^2=0.12, $p<0.0001$), equal severity x density-weighted values (r^2=0.09, $p<0.0001$) and equal severity x presence/absence values (r^2=0.12, $p<0.0001$), but had lower predictive power. These relationships indicate sites with low risk being associated with species such as woundfin, desert sucker and Sonora sucker. Risk indices were also significantly related to fish trait PCIII but had relatively low predictive power ($r^2<0.009$; Table 1.6).

Aquatic Ecological System (AES) scale

The mean risk value of the different risk indices at the AES scale ranged from 31 to 61 (Table 1.5). Similar to catchment and watershed scales, density-weighted indices tended to have lower risk values than presence/absence indices. Also, linear regression indicated no difference between indices, with stronger associations between indices with the same frequency method (presence/absence: r^2=0.96, slope=1.08, $p<0.0001$; density-weighted: r^2=0.97, slope=0.96, $p<0.0001$) whereas indices with different frequency treatments were less related (equal severity: r^2=0.74, slope=0.80, $p<0.0001$; severity r^2=0.75, slope=1.00, $p<0.0001$; Table 1.6).

Density-weighted indices indicated the greatest risk to be in central Arizona near Phoenix (Salt River Basin) and Tucson (Santa Cruz River Basin) and Las Vegas, Nevada (northwestern region of the mainstem Colorado River Basin; Figure 1.11). The presence/absence indices indicated widespread high risk throughout the LCRB, concentrating in the same urban areas in the Santa Cruz, Salt and Lower Gila river basins as well as portions of the Virgin, mainstem Colorado, and Verde river basins.

There were 173 AES' and 103 AES_{UP} with fish data used to relate risk values to fish traits. The first three axes of the PCA for the AES and upstream of AES scales explained 47.4% of the variation among sites. The first axis explained 25.0% of total variation and had high loadings of fish which spawn on various substrates (generalist spawners) and had high length at maturity. Fish species that scored high on this axis included channel catfish (*Ictalurus punctatus*) and flathead catfish (*Pylodictis olivaris*). The second axis explained 12.9% of total variation and had high loadings of fish considered to be benthic, not fluvial dependent and herbivorous/detrivorous, preferring cold temperatures, rubble or silt/mud substrate, moderate to fast currents, and having low spawning temperature. Fish species that scored high on this axis included desert sucker and Sonora sucker while species scoring low on this axis included topminnow and western mosquitofish. The third axis explained 9.5% of total variation and had high loadings of fish that preferred sand substrate and moderate current and were external bearing and pelagic spawning while having low maximum size, narrow diet breadth and low fecundity. Fish species associated with this axis included woundfin and longfin dace (*Agosia chrysogaster*). This PCA was used for both AES and AES_{UP} scales.

At the AES scale there were no significant relationships between fish trait PC axes and risk indices after Bonferroni correction (Table 1.5). The equal severity x density-weighted risk values tended to be related to PCII ($p=0.05$) and PCIII ($p=0.033$), and severity-weighted x presence/absence values tended to be related to PCI ($p=0.04$) and PCII ($p=0.042$). However, all relationships had p-values greater than the Bonferroni-adjusted alpha level of 0.0004 and were considered non-significant.

Upstream of Aquatic Ecological System (AES_{UP}) scale

Mean risk values varied from 46 to 86 among indices (Table 1.5). Similar to all other scales, density-weighted indices tended to have lower risk values than presence/absence indices. Also similar to other scales, values for the equal severity x presence/absence index were strongly related to values for the severity-weighted x presence/absence index ($r^2=1.00$, slope=0.98, $p<0.0001$; Table 1.6), and risk values for the equal severity x density-weighted index were strongly related to risk values for the severity-weighted x density-weighted index (r^2=0.97, slope=0.98, p<0.0001). Values for the equal severity x presence/absence index were less strongly related to values for the equal severity x density-weighted index (r^2=0.54, slope=0.76, p<0.0001) as were values for the severity-weighted x presence/absence index when related to values for the severity-weighted x density-weighted index (r^2=0.54, slope=0.77, p<0.0001).

Density-weighted frequency scores at the upstream of the AES scale indicated greatest risk indices in the Lower Gila and Santa Cruz basins (Figure 1.12). Presence/absence indices at this scale show high risk to be widespread throughout the entire LCRB. All indices consistently indicate the mainstem Colorado River to have low risk between the confluences of the Little Colorado and Virgin rivers.

No significant relationships were found between risk indices and fish trait PC axes (Table 1.5). The severity-weighted x density-weighted and equal severity x density-weighted indices were both related to PCII (p=0.047; p=0.013; respectively). However, all relationships had p-values greater than the Bonferroni-adjusted alpha level of 0.0004 and were considered non-significant.

Discussion

Our results suggest that the development of an ERI may be a useful tool for native fish conservation, but the scale and methods used to create the index are important. Most stressors were consistent in density across scales implying these stressors are homogenous throughout the landscape. Inconsistent densities across scales would suggest stressors are not uniform throughout the landscape. In the case of dams, since the catchment scale had the highest mean density it is suggestive that although dams can impact various scales (Poff and Hart 2002) they often have more influence at the localized catchment scale. Mean risk values of indices increased as spatial scale increased. The lowest mean risk values were at the catchment scale and increased at watershed, AES and AES_{UP} scales. However, the catchment and watershed scales were the only two scales to have significant relationships with fish trait PC axes, and relationship between risk values and fish traits was strongest at the watershed scale.

While anthropogenic disturbances affect aquatic systems through multiple processes within different scales (Stewart et al. 2001; Wang et al. 2003), our results suggest the most meaningful scale to utilize risk indices for conservation efforts was the watershed scale. These results are similar to numerous studies which have assessed the influence of the landscape at the watershed scale to different aspects of stream ecology (Schlosser 1991; Roth et al. 1996; Allan et al. 1997). Wang et al. (2003) suggested with natural landscapes, local environmental factors are more related to fish assemblages, whereas fish assemblages in increasingly modified landscapes have stronger relationships with watershed factors. Although large protected areas have been established to protect freshwater systems (Saunders et al. 2002), these preserves are rare and primarily focus on protecting terrestrial ecosystems. Citizen-based watershed groups are increasingly common (Griffin 1999) and regularly focus aquatic conservation efforts at the watershed scale, but can be restricted by political boundaries within watersheds (Allan et al.1997). These boundaries can challenge the watershed approach to river management; however restoration efforts are often localized undertakings which are part of a larger watershed management plan (Bernhardt et al. 2007). This suggests that integrative watershed management can be effective, and an ERI could be easily incorporated into watershed management plans.

While indices using the presence/absence frequency method typically resulted in greater risk values than density-weighted indices, there was a high degree of correlation between all indices. The presence/absence indices are predisposed to have greater risk because, prior to standardization, there were narrower ranges of risk values as compared to the density-weighted indices and therefore resulted in greater risk values. Density-weighted indices account for varying densities of stressors therefore greatly inflate the range of risk values prior to standardization. Since density-weighted scores are based on equal quartiles, it is highly unlikely that many spatial units will have high risk scores for these indices.

Although density-weighted indices were hypothesized to be more strongly correlated with fish traits, results suggest that presence/absence indices are slightly more robust than density-weighted indices. At the catchment scale, all indices were significantly related to at least one of the fish trait PC axes; however, these relationships are likely driven by large sample size of sites and had low predictive power. At the watershed scale, the strongest relationships with stressor indices were with non-fluvial dependent fishes (e.g., common carp) preferring low velocities and silt/mud substrates. While all four indices at the watershed scale were related to these life history traits, the presence/absence indices had higher regression coefficients.

Even the most robust risk indices had fairly low predictive power (r^2=0.20) at the watershed scale, which is likely related to the coarse scale of the fish traits (presence/absence) and landscape-level stressors used. Nonetheless, our analysis suggests that the ERI is related to biotic communities (fish traits) and therefore is applicable for prioritizing areas for native fish conservation. Incorporating cost-effective risk indices into conservation planning may greatly increase the effectiveness of conservation efforts. As risk indices are utilized and as links between anthropogenic activities and ecosystem drivers are better established, indices are likely to be improved, increasing predictive ability. Ecological risk indices can also identify patterns in the landscape which can be useful not only for conservation and management purposes but also land-use planning (Mattson and Angermeier 2007). However, limitations in the development of risk indices include 1) incorporating stressors in which no spatial data is available; 2) incorporating a ‘distance-weighting’ in which risk is weighted by distance of a stressor from a stream reach; and 3) incorporating landscape-level factors which lower or lessen risk to aquatic fauna such as mitigation efforts, riparian buffers and other landscape-level patterns (Wang et al. 2008; Mattson and Angermeier 2007).

While this analysis was primarily conducted to better understand the effectiveness of the risk indices using different scales and methods, use of the LCRB allows us to make insights on the influence of risk indices on fish assemblages within the basin. As hypothesized, risk was consistently highest near the urban centers of Phoenix, Las Vegas and Tucson metropolitan areas for all indices, and near St. George, Prescott and Flagstaff in most indices. This is likely because the presence and densities of urban land use, roads, railroads, canals, stream crossings, NPDES, waste facilities tend to be highest near population centers. Changes brought about with increased urban land area include increases of pollutants in runoff, altered hydrology due to increased impervious surface area, increased water temperatures, bank destabilization, channelization, and limited interactions between the river and its floodplain (Paul and Meyer 2001; Allan 2004).

Stream segments in the LCRB with high risk commonly have fishes such as fathead minnow and common carp present whereas stream segments with low watershed risk did not have those species. Common carp are ubiquitous throughout the LCRB (Minckley et al. 2003) and are considered highly adaptable, generalist species with high fecundity, rapid growth, longevity, tolerance of a broad range of water quality, and a wide diet breadth (Harris 1996). Common carp have been found to be associated with highly disturbed streams (Schade and Bonar 2005). Fathead minnows are widespread throughout the LCRB, tolerant to pollution and often reach high abundance where few other species would be able survive (Minckley 1973; Becker 1983). Stream segments that had fewer fish which are native, benthic, herbivorous/detrivorous, have low maximum size, short lifespan, narrow diet breadth, low fecundity, high shape factor and prefer moderate velocities were related to areas of high risk at the watershed scale. Therefore, sites with less risk have native species such as desert sucker, Sonora sucker, and woundfin present whereas sites with greater risk do not have these species present. Woundfin are classified as an endangered species (U.S. Fish and Wildlife Service 1994) while desert sucker and Sonora sucker are considered sensitive species (U.S. Department of the Interior 2005). Conservation efforts for these and other native desert fishes need to include protection of suitable habitats within watersheds, of which an ERI can help prioritize.

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). Currently at just over 6 million people in Arizona alone, population trends are projected to continue with population approaching 11 million by 2030 (U.S. Census Bureau 2004). These population increases not only exert increased demands on natural resources, but also suggest increased levels of anthropogenic activities. Human population has been proven a predictor of biotic homogenization of fishes (Olden et al. 2006b) while ecological risk indices show increased levels of anthropogenic risk to be related to declines in native, fluvial dependent species and increases in nonnative generalist species, suggesting anthropogenic activities increase nonnative introductions, native extirpations and habitat alteration driving biotic homogenization. The LCRB has a rapidly changing landscape which will likely continue at the expense of losing aquatic habitat and diversity, unless planning and management can work together to minimize risk to prioritized watersheds. Ecological risk indices can be a useful tool in prioritizing watersheds for conservation needs to achieve conservation goals.

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Table 1.1 Sources of spatial data for anthropogenic stressors used to derive ecological risk indices for the Lower Colorado River Basin.

Stressor	Data Source*
Canals (km/km^2)	US Geological Survey, National Hydrography Dataset (2005)
Dams (m^2/ km^2)	US Army Corp of Engineers, National Inventory of Dams (2000)
Diversion ($no./km^2$)	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)
303d Impaired stream classification ($no./km^2$)	Environmental Protection Agency, Water Quality Standards Database (2002)
Urban and Agricultural Landcover (km^2/ km^2)	National Landcover Database (2000)
Mines ($no./km^2$)	US Geological Survey, Mineral Resources Database (2005)
Non-point Discharge Elimination System ($no./ km^2$)	Environmental Protection Agency, Permit Compliance System (2006)
Railroads (km/km^2)	US Census Bureau, Tiger files (2006)
Roads (m/km^2)	US Census Bureau, Tiger files (2006)
Stream Crossings ($no./km^2$)	US Census Bureau, Tiger files (2006)
Superfund/Toxic Release/Hazardous Waste Facilities ($no./ km^2$)	Environmental Protection Agency, Superfund (2006), Toxic Release Inventory (TRI; 2006) and Resource Conservation and Recovery Act (RCRA; 2006) databases

*Year signifies the most current year of data included in the dataset.

Table 1.2. Summary of severity scores by the ecological integrity categories (Karr 1991). Weighted severity scores are based upon peer-reviewed literature whereas equal severity scores were assigned on the assumption that all stressors have equal influence. 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.

	Severity Weightings					Severity Score Methods	
Stressors	Water Quality	Habitat Quality	Biotic Interactions	Flow Regime	Energy Source	Weighted Severity	Equal Severity
Agriculture	3	3	1	2	3	12	1
Canals	2	3	2	3	2	12	1
Dams	3	3	3	3	3	15	1
Diversions	0	1	0	3	0	4	1
Mines	3	2	1	1	1	8	1
NPDES	3	1	1	2	3	10	1
Railroads	2	2	0	0	1	5	1
Roads	2	2	0	1	2	7	1
Stream Crossings	2	2	1	1	1	7	1
Urban	3	3	1	2	3	12	1
Waste Facilities	3	2	1	0	2	8	1
303d Streams	3	0	1	0	1	5	1

Table 1.3. Breakdown of density-weighted frequency scores at the catchment scale.

Stressor	Density-weighted frequency scores 0	1	2	3	4	Ranking method
Agricultural (km^2/km^2)	0	1-10%	11-25%	26-50%	>50%	Wang et al. 1997; Allan 2004
Canals (m/km^2)	0	1-79	80-348	349-927	>927	Equal Quartile
Dams (m^2/km^2)	0	1-18	19-73	74-787	>788	Equal Quartile
Diversions ($no./km^2$)	0	0.1-0.2	0.3-0.5	0.6-1.0	>1.0	Equal Quartile
Mines ($no./km^2$)	0	0.1-0.2	0.3-0.4	0.5-0.7	>0.7	Equal Quartile
NPDES ($no./km^2$)	0	0.01-0.07	0.008-0.15	0.16-0.3	>0.3	Equal Quartile
Railroads (m/km^2)	0	1-117	118-307	308-790	>790	Equal Quartile
Roads (m/km^2)	0	1-416	417-838	839-1474	>1474	Equal Quartile
Stream Crossings ($no./km^2$)	0	0.1-0.2	0.3-0.4	0.5-1.0	>1.0	Equal Quartile
Urban ($no./km^2$)	0	0.1-1%	2-3%	4-8%	>8%	Wang et al. 2000; Wheeler et al. 2005; Allan 2004
Waste Facilities ($no./km^2$)	0	0.01-0.04	0.05-0.10	0.11-0.30	>0.30	Equal Quartile
303d (m/km^2)	0	1-256	257-834	835-1538	>1539	Equal Quartile

Table 1.4. Summary of the mean and range of stressor densities at the four scales used to calculate the density-weighted ecological risk indices in the Lower Colorado River basin.

Stressor	Catchment	Watershed	AES*	Upstream of AES
	Mean (Range)	Mean (Range)	Mean (Range)	Mean (Range)
Area (km^2)	5.1 (0.0003 - 7,683)	4180 (1 - 343,896)	966 (54 - 7,683)	38,953 (550 - 343,971)
Agriculture (km^2/ km^2)	0.8 (0 - 100)	0.2 (0 - 78)	1.6 (0 - 44)	0.7 (0 - 9.1)
Canals (m/km^2)	12.2 (0 - 20,053)	2.8 (0 - 1,117)	17.7 (0 - 565)	7.8 (0 - 147.6)
Dams (m^2/km^2)	201.3 (0 - 7,660,974)	7.4 (0 - 66,584)	172.5 (0 - 55,967)	11.0 (0 - 212.6)
Diversions (no./km^2)	0.3 (0 - 587)	0.2 (0 - 23)	0.2 (0 - 6.4)	0.2 (0 - 0.9)
Mines (no./km^2)	0.05 (0 - 79)	0.05 (0 - 7.2)	0.05 (0 - 0.4)	0.04 (0 - 0.3)
NPDES (no./km^2)	0.001 (0 - 5)	0.0006 (0 - 0.6)	0.001 (0 - 0.03)	0.001 (0 - 0.01)
Railroads (m/km^2)	22.6 (0 - 9,437)	6.1 (0 - 823)	14 (0 - 253)	11.6 (0 - 82.9)
Roads (m/km^2)	789 (0 – 28,927)	679 (0 - 11,730)	822 (13 - 5,779)	941 (33 - 4911)
Stream Crossings (no./km^2)	0.6 (0 - 673)	0.3 (0 - 20)	0.3 (0 - 1.1)	0.3 (0.008 - 0.7)
Urban (km^2/ km^2)	1.3 (0 - 100)	0.8 (0 - 100)	2 (0 - 50.5)	2 (0 - 39)
Waste Facilities (no./km^2)	0.0006 (0 - 3)	0.0003 (0 - 0.3)	0.002 (0 - 0.2)	0.0014 (0 - 0.04)
303d (m/km^2)	42.7 (0 - 61,067)	11.6 (0 - 2,671)	19.9 (0 - 1,531)	20 (0 - 758)

*AES= aquatic ecological system

Table 1.5. Analysis of covariance results of relationships between risk indices and fish trait PC axes with stream order (at catchment and watershed scales only) and basin as covariates. Mean and range of standardized risk values for each index are listed as we ll. Significance levels were adjusted with a Bonferroni correction at each scale (α=0.05/12 or 0.004), with bolded P-values as significant after correction.

Scale	Index*	Mean	Range	PCI	PCII	PCIII
Catchment	SW x PA	12.7	0-100	**p=0.001**	**p<0.0001**	**p<0.0001**
	ES x PA	14.7	0-100	p=0.078	**p<0.0001**	**p<0.0001**
	SW x DW	7.6	0-65	**p=0.0003**	**p=0.0002**	**p<.0001**
	ES x DW	8.9	0-60	p=0.208	p=0.0062	**p<0.0001**
Watershed	SW x PA	25	0-100	**p<0.0001**	**p<0.0001**	**p<0.0001**
	ES x PA	35.5	0-100	**p<0.0001**	**p<0.0001**	**p<0.0001**
	SW x DW	17.1	0-80	**p<0.0001**	**p<0.0001**	**p<0.0001**
	ES x DW	20.3	0-85	**p<0.0001**	**p<0.0001**	**p<0.0001**
AES	SW x PA	58.6	13-100	p=0.040	p=0.042	p=0.385
	ES x PA	60.3	17-100	p=0.570	p=0.208	p=0.325
	SW x DW	31.1	3-90	p=0.391	p=0.657	p=0.786
	ES x DW	33.6	4-88	p=0.889	p=0.050	p=0.033
Upstream	SW x PA	85.3	36-100	p=0.051	p=0.055	p=0.084
AES	ES x PA	85.1	33-100	p=0.102	p=0.139	p=0.880
	SW x DW	46	10-89	p=0.096	p=0.047	p=0.402
	ES x DW	48.2	10-90	p=0.208	p=0.013	p=0.092

*Index method codes: SW=severity-weighted, ES=equal severity, DW=density-weighted, PA=presence/absence

Table 1.6. Linear regression statistics of regressing each risk assessment method with each other at four different spatial scales. A slope of 1.0 indicates indices produce similar risk values. High r2 values would indicate that the two indices were highly related, even if the slope may not be close to 1.0. Samples sizes are 73,078 for the catchment scale, 36,379 for the watershed scale, 386 for the AES scale, and 197 for the upstream AES scale. All tests were significant (p<0.0001).

Scale	Index Test*	Slope	Standard Error	r^2
Catchment	SW PA = ES x PA	0.92	0.0008	0.95
	SW x DW = ES x DW	1.06	0.0009	0.95
	ES x PA = ES x DW	0.61	0.0012	0.78
	SW x PA = SW x DW	1.34	0.0025	0.80
Watershed	SW PA = ES x PA	1.03	0.0007	0.98
	SW x DW = ES x DW	1.09	0.0011	0.96
	ES x PA = ES x DW	0.55	0.0013	0.82
	SW x PA = SW x DW	1.73	0.0043	0.82
AES	SW PA = ES x PA	1.08	0.0108	0.96
	SW x DW = ES x DW	0.96	0.0089	0.97
	ES x PA = ES x DW	0.80	0.0239	0.74
	SW x PA = SW x DW	1.00	0.0297	0.75
AES_{UP}	SW PA = ES x PA	1.00	0.0101	0.98
	SW x DW = ES x DW	0.97	0.0098	0.98
	ES x PA = ES x DW	0.76	0.0506	0.54
	SW x PA = SW x DW	0.77	0.5044	0.54

*Index method codes: SW=severity-weighted, ES=equal severity, DW=density-weighted, PA=presence/absence

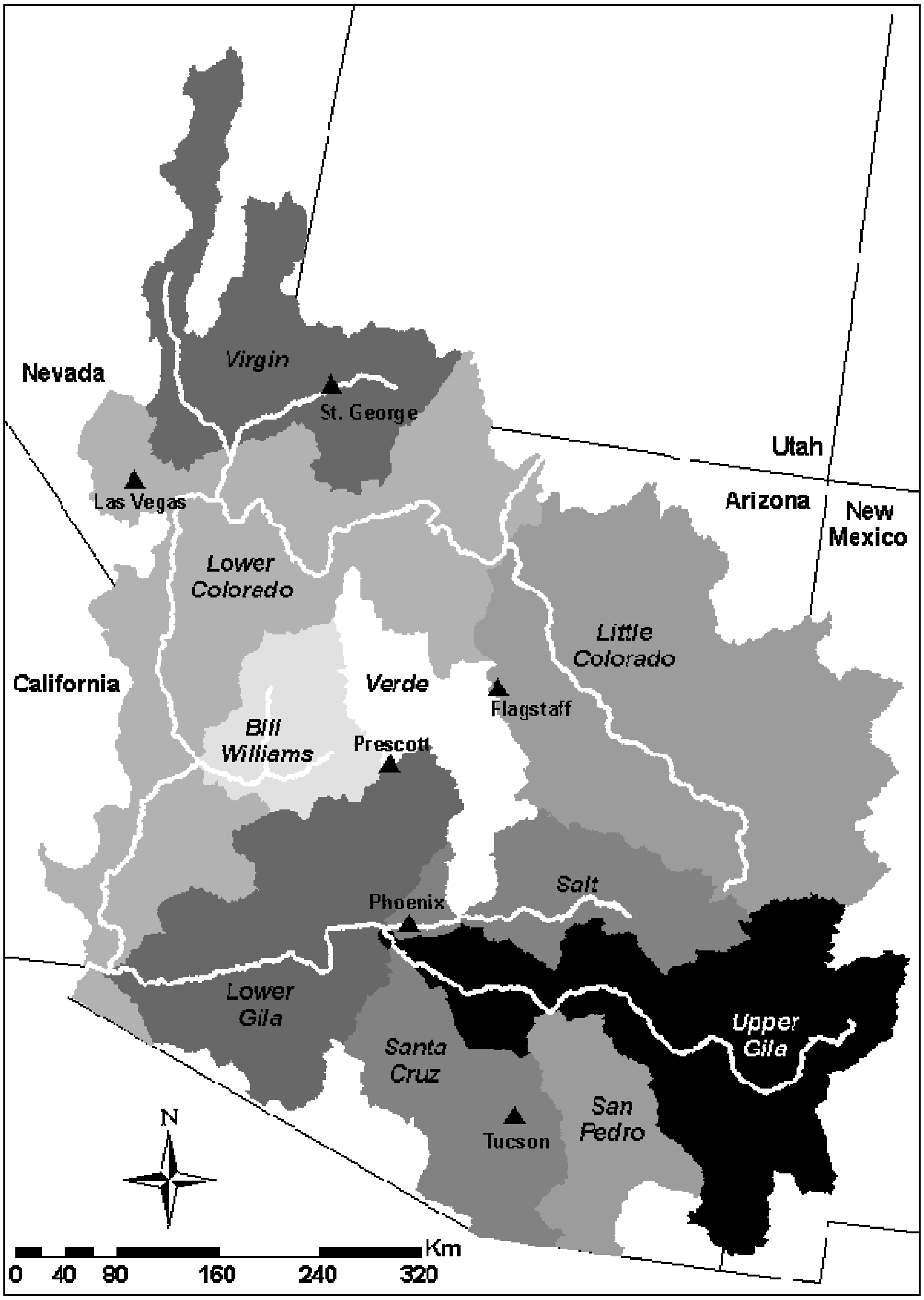

Figure 1.1. The geographic location of the Lower Colorado River Basin with major river basins shaded and labeled. Major rivers are highlighted in white, major cities are denoted with black triangles and states by background lines.

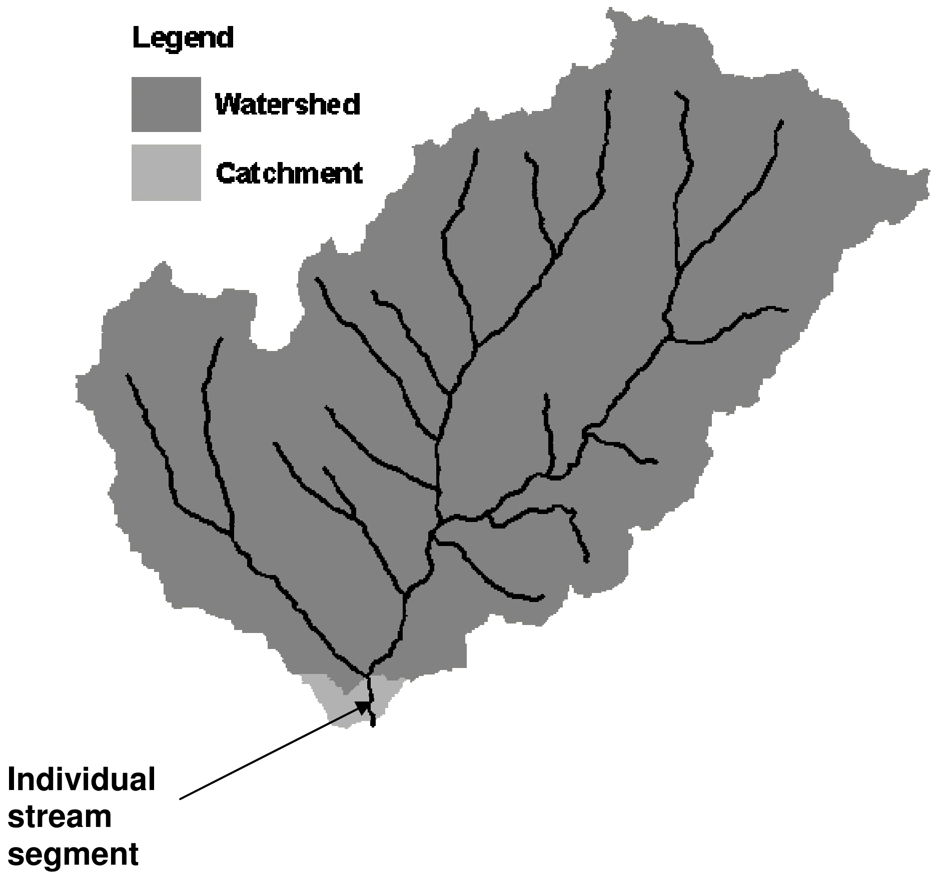

Figure 1.2. Catchment and watershed scale for an individual stream segment of interest.

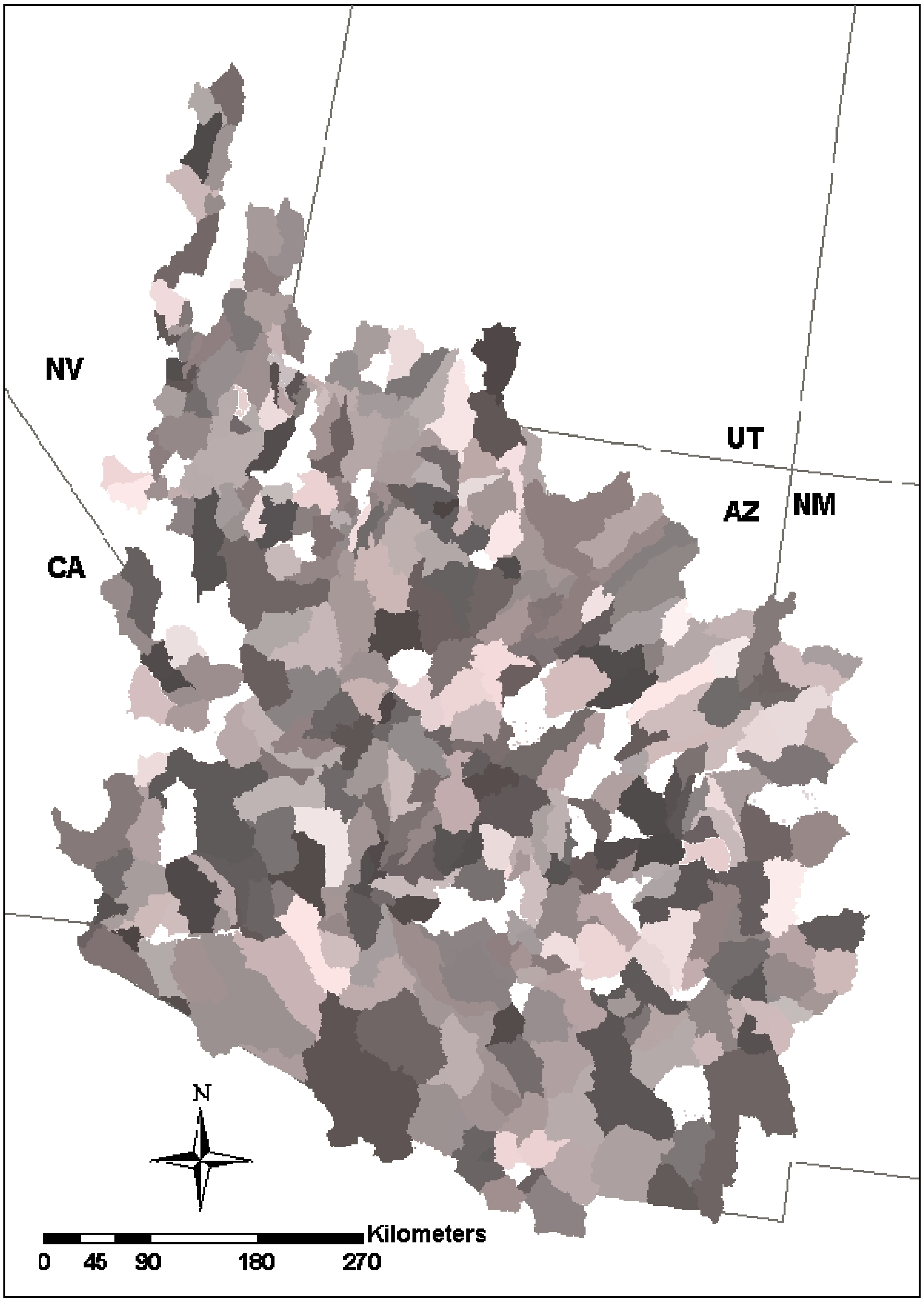

Figure 1.3. Lower Colorado River Basin delineated by the 386 aquatic ecological systems (AES) boundaries.

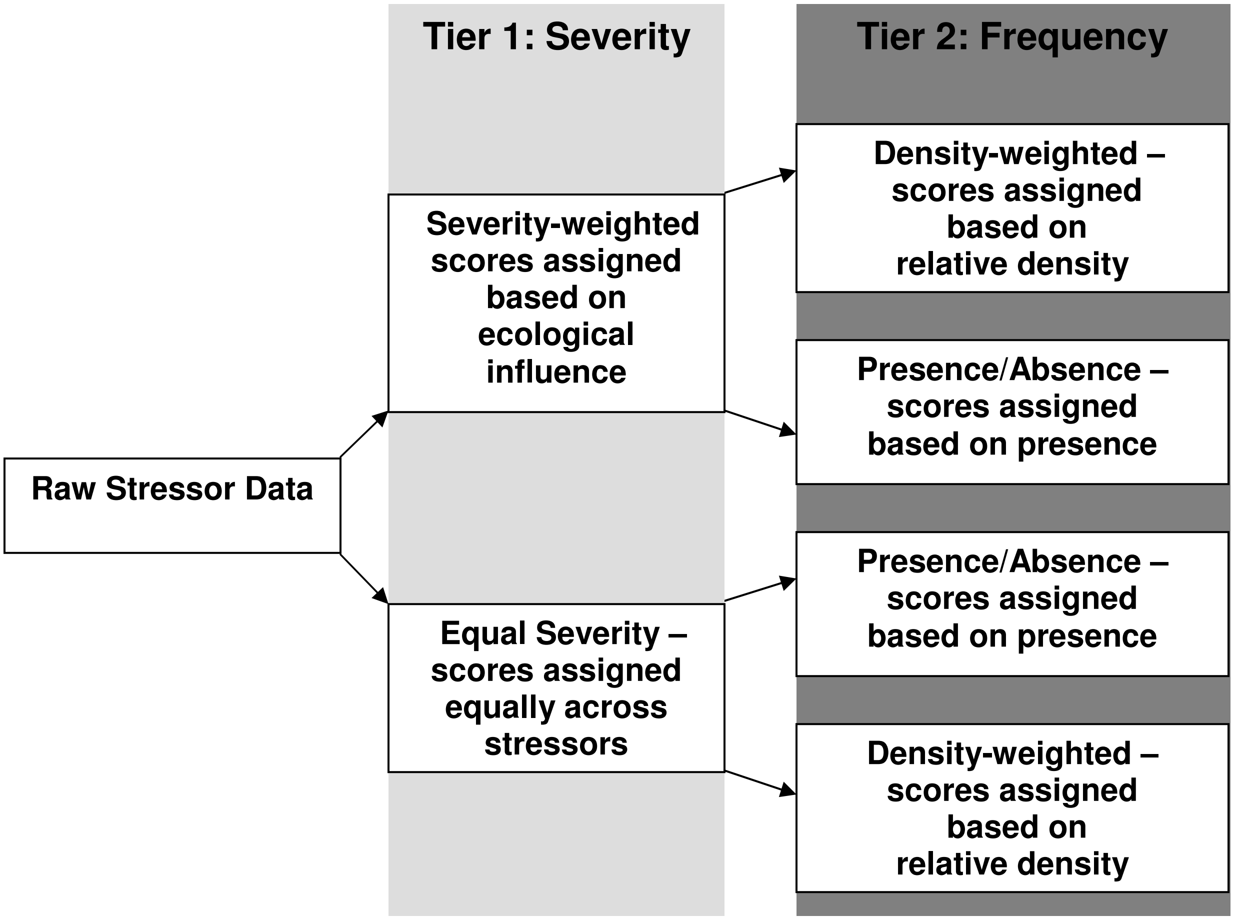

Figure 1.4. Schematic for developing the four ecological risk indices created for each of four scales within the Lower Colorado River Basin. Severity scores are multiplied by frequency scores to generate risk values. Each index uses different methods for quantifying risk from the same raw spatial data.

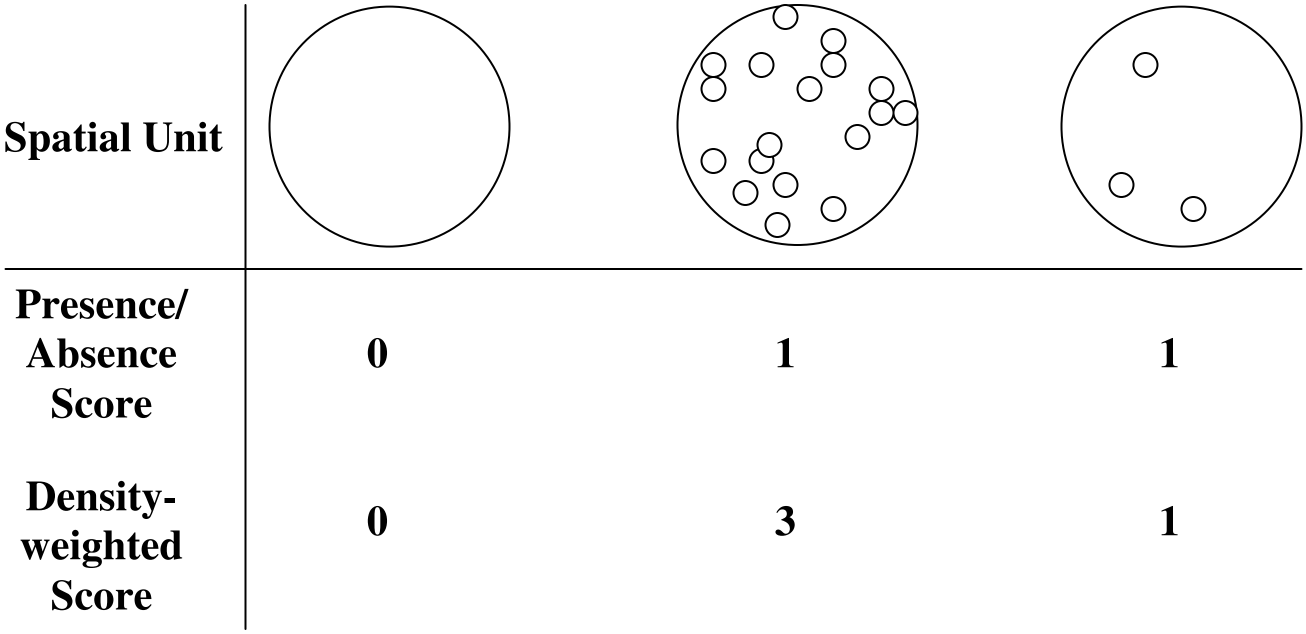

Figure 1.5. Two methods for quantifying frequency scores. The large circles represent the spatial unit of analysis (e.g., catchment) while the small circles represent a single point stressor (e.g., mine). Presence/absence scores are based on presence of a stressor while density-weighted frequency scores are based on equal quartiles of the density of the stressor.

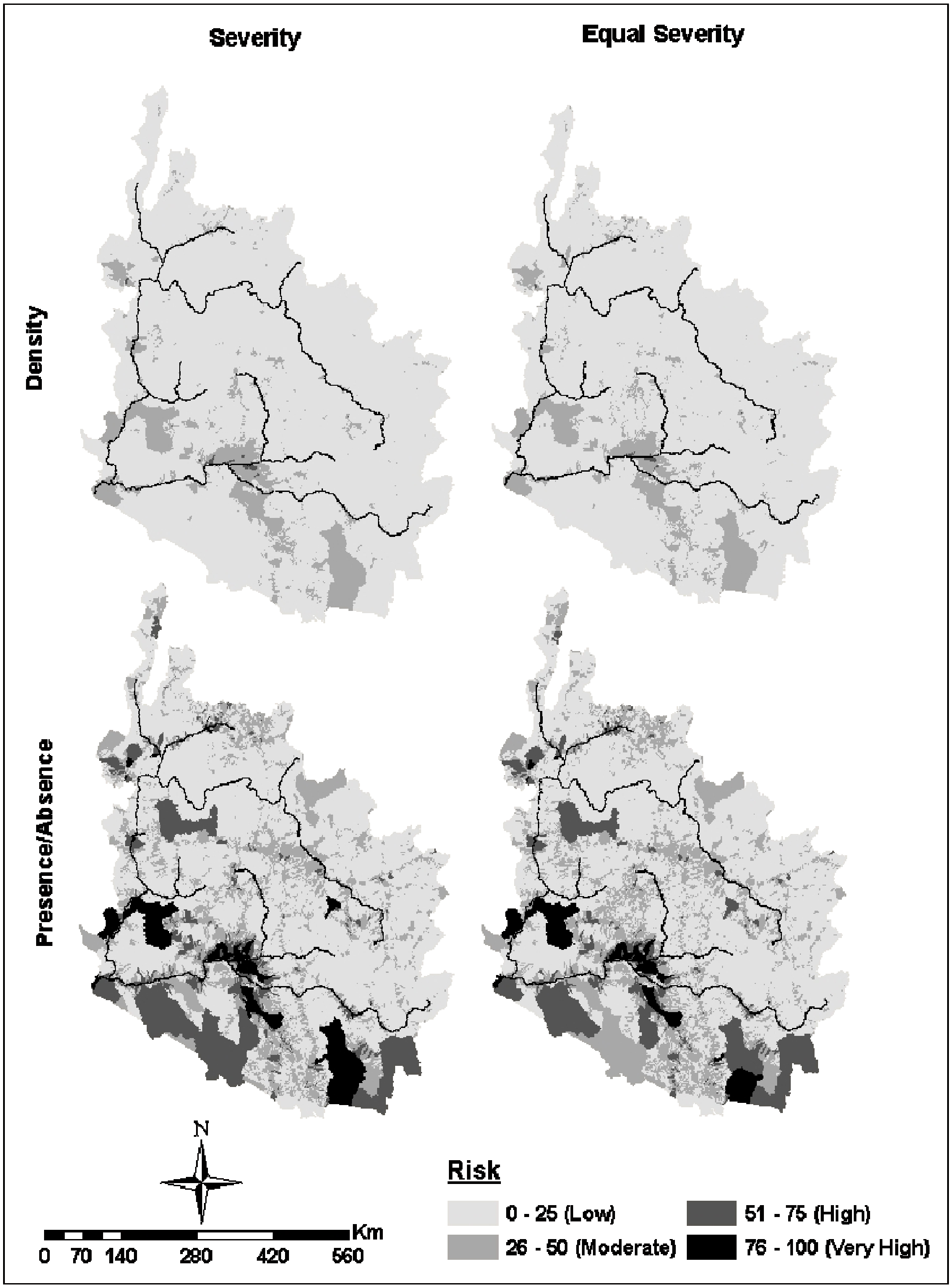

Figure 1.6. Ecological risk values at the catchment scale for the Lower Colorado River Basin. Major rivers are outlined in black. Index methods are noted on the left-hand side and top of maps.

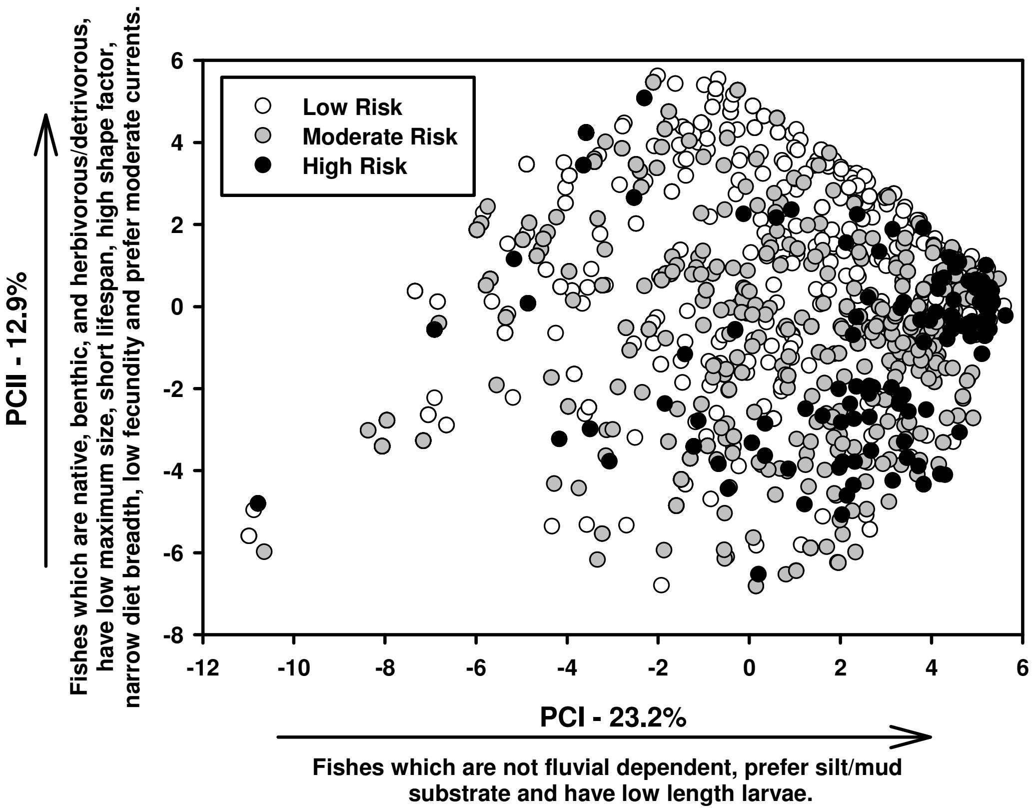

Figure 1.7. Fish trait principal components analysis results showing the first two principal components used for the catchment and watershed scales. Fishes that scored high on PC I include fathead minnow and common carp whereas fishes that scored high on PC II include desert sucker and Sonora sucker. Stream segments are color-coded by risk according to the severity-weighted x density-weighted index at the watershed scale.

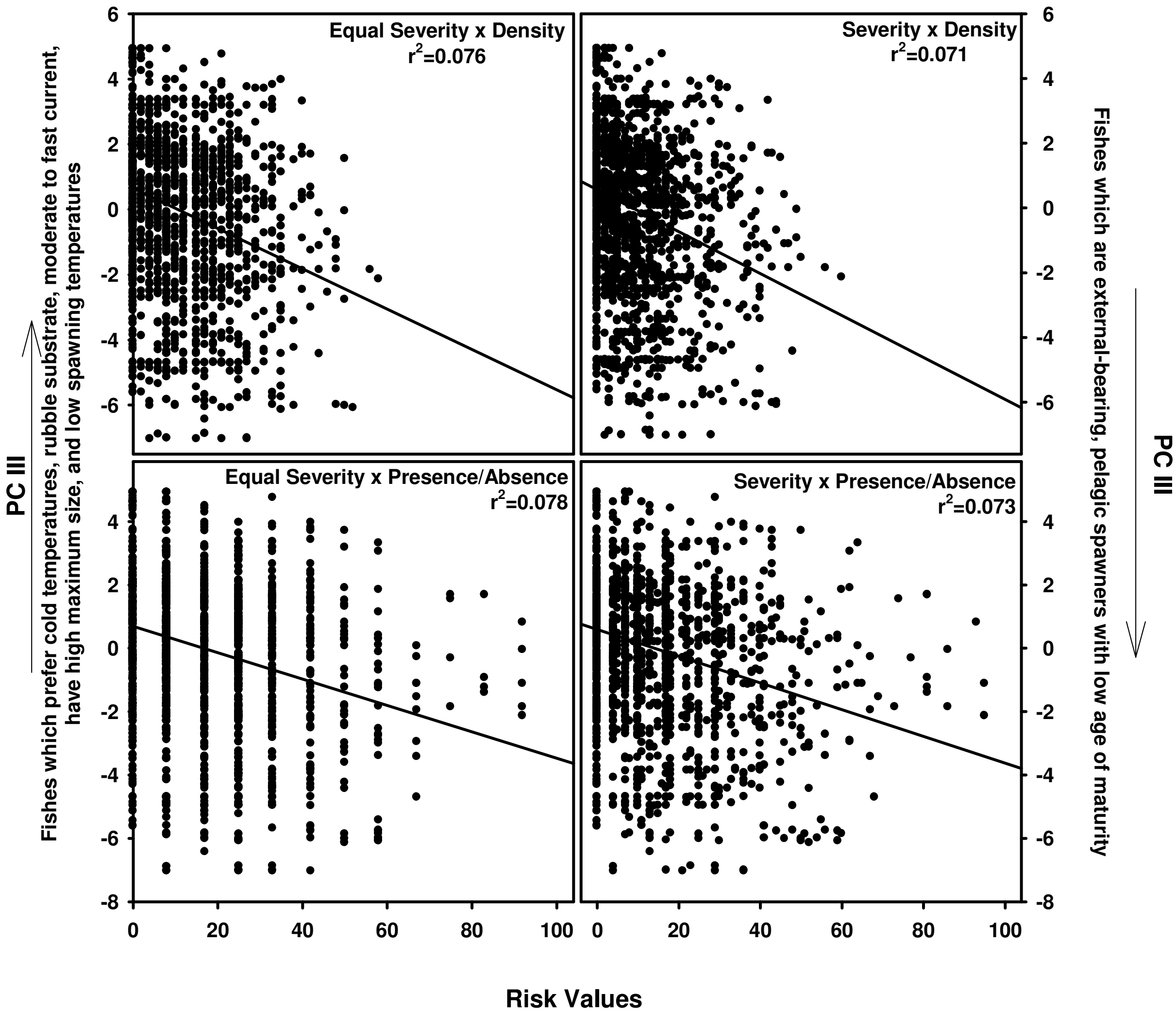

Figure 1.8. Linear regressions between risk indices at the catchment scale and fish trait PC III. Fish species scoring high on PC III included rainbow and brown trout and fishes scoring low on PC III included western mosquitofish. Sample size for all analyses was 1,718.

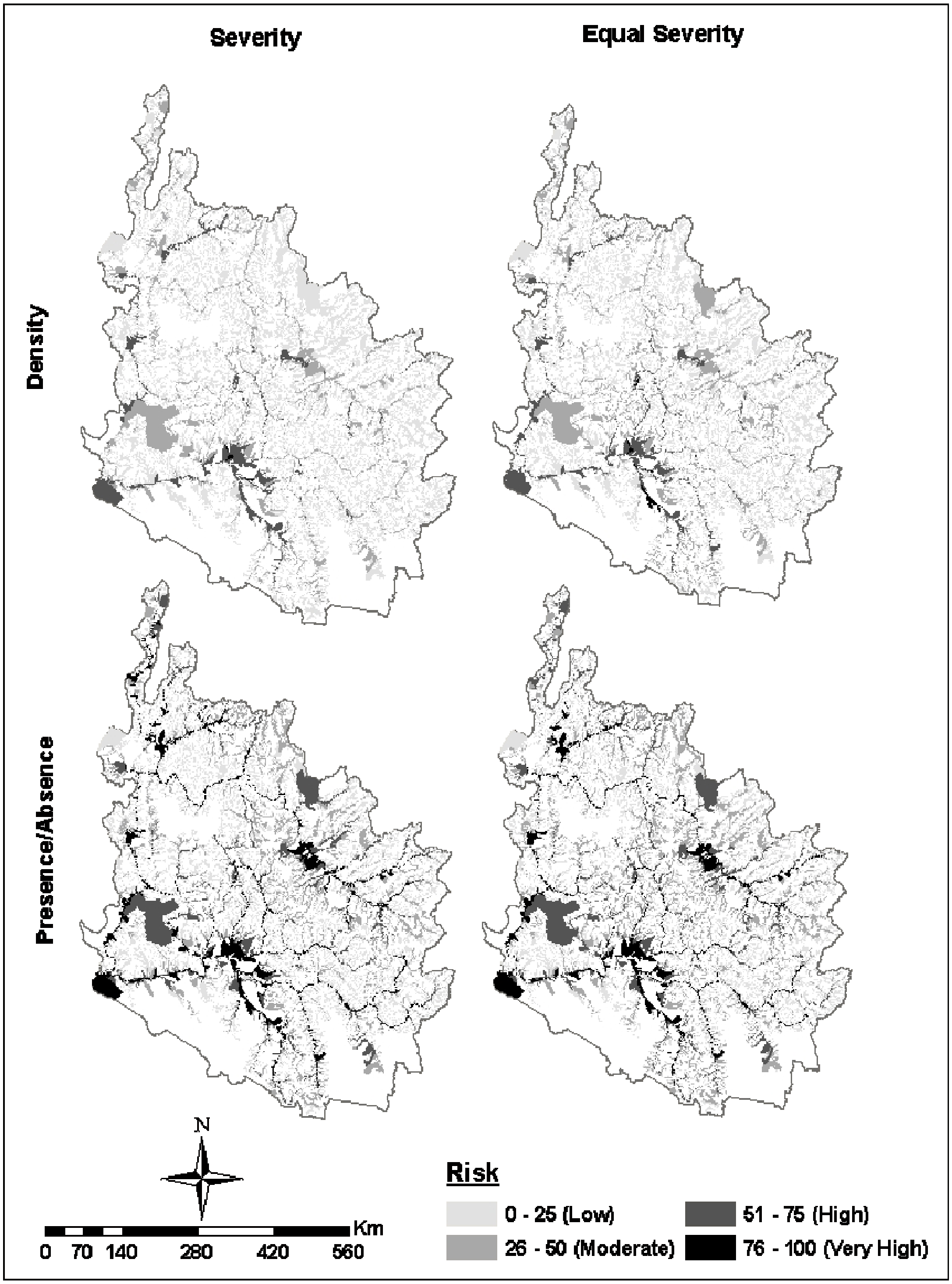

Figure 1.9. Ecological risk values at the watershed scale of the Lower Colorado River Basin. Index methods are noted on left-hand side and top of maps.

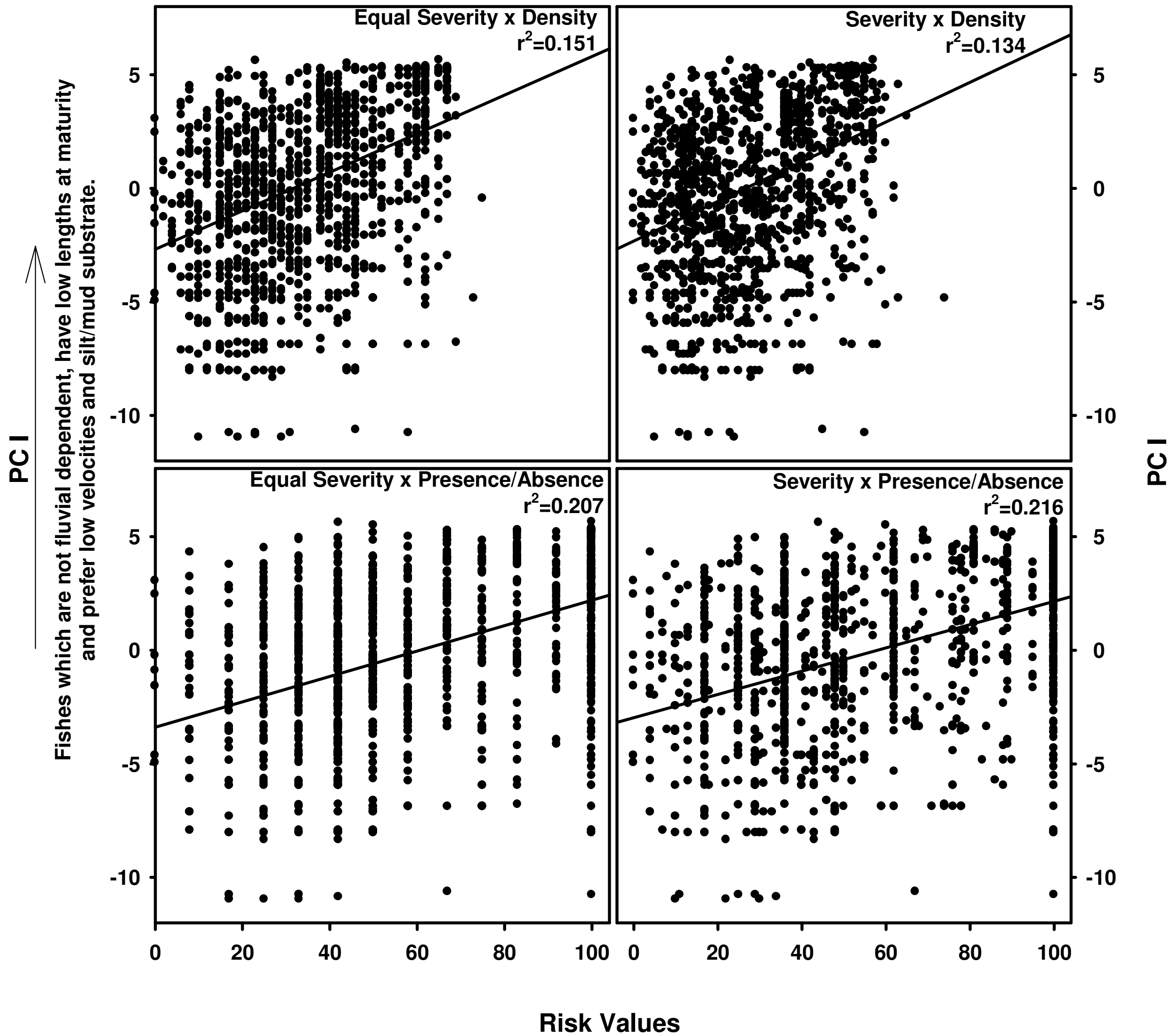

Figure 10. Linear regressions between risk indices at the watershed scale and fish trait PC I. Fish species scoring high on PC I include fathead minnow and common carp.

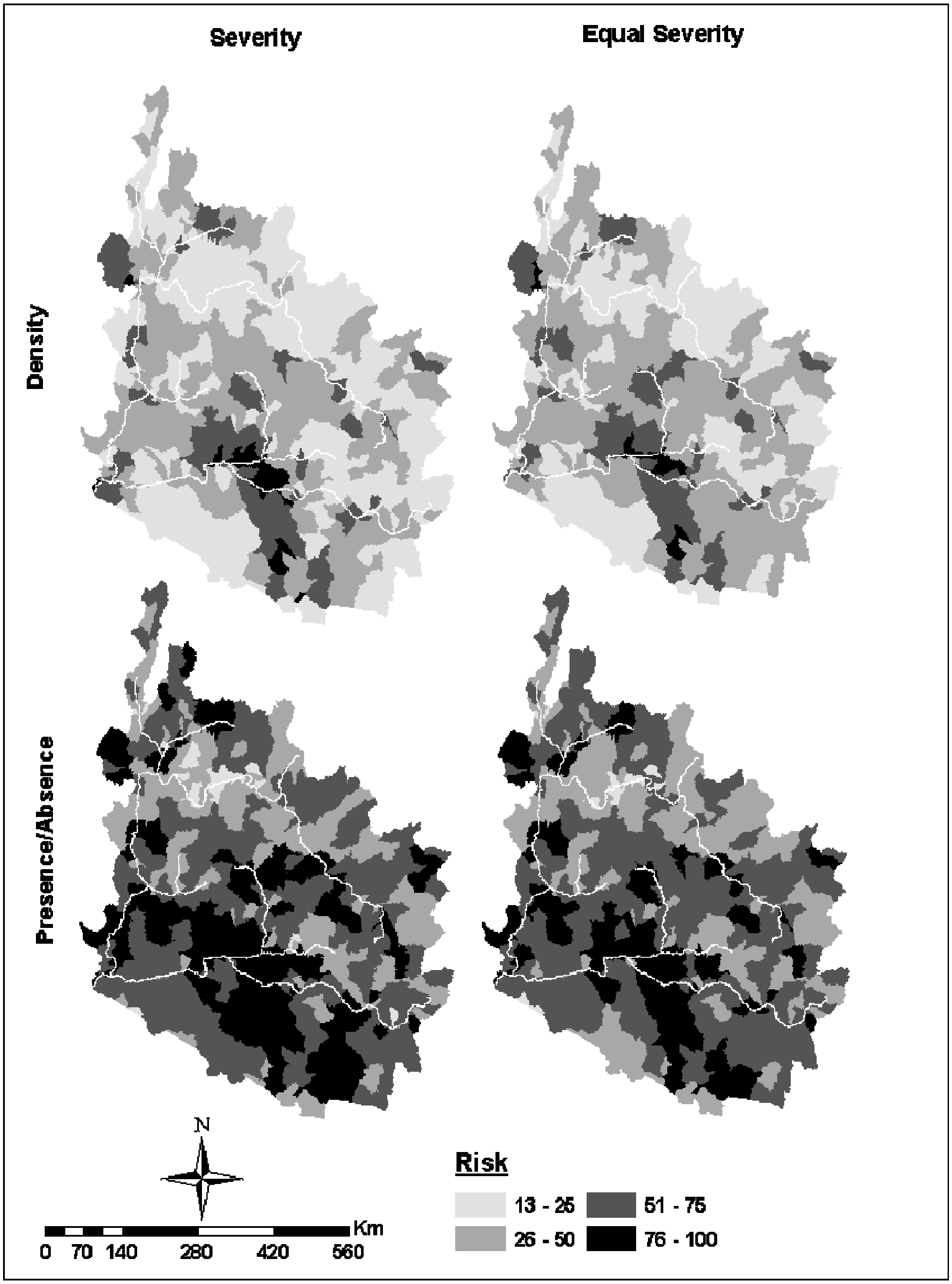

Figure 1.11. Ecological risk values at the AES scale for the Lower Colorado River Basin. Major rivers are outlined in white. Index methods are noted on the left-hand side and top of maps.

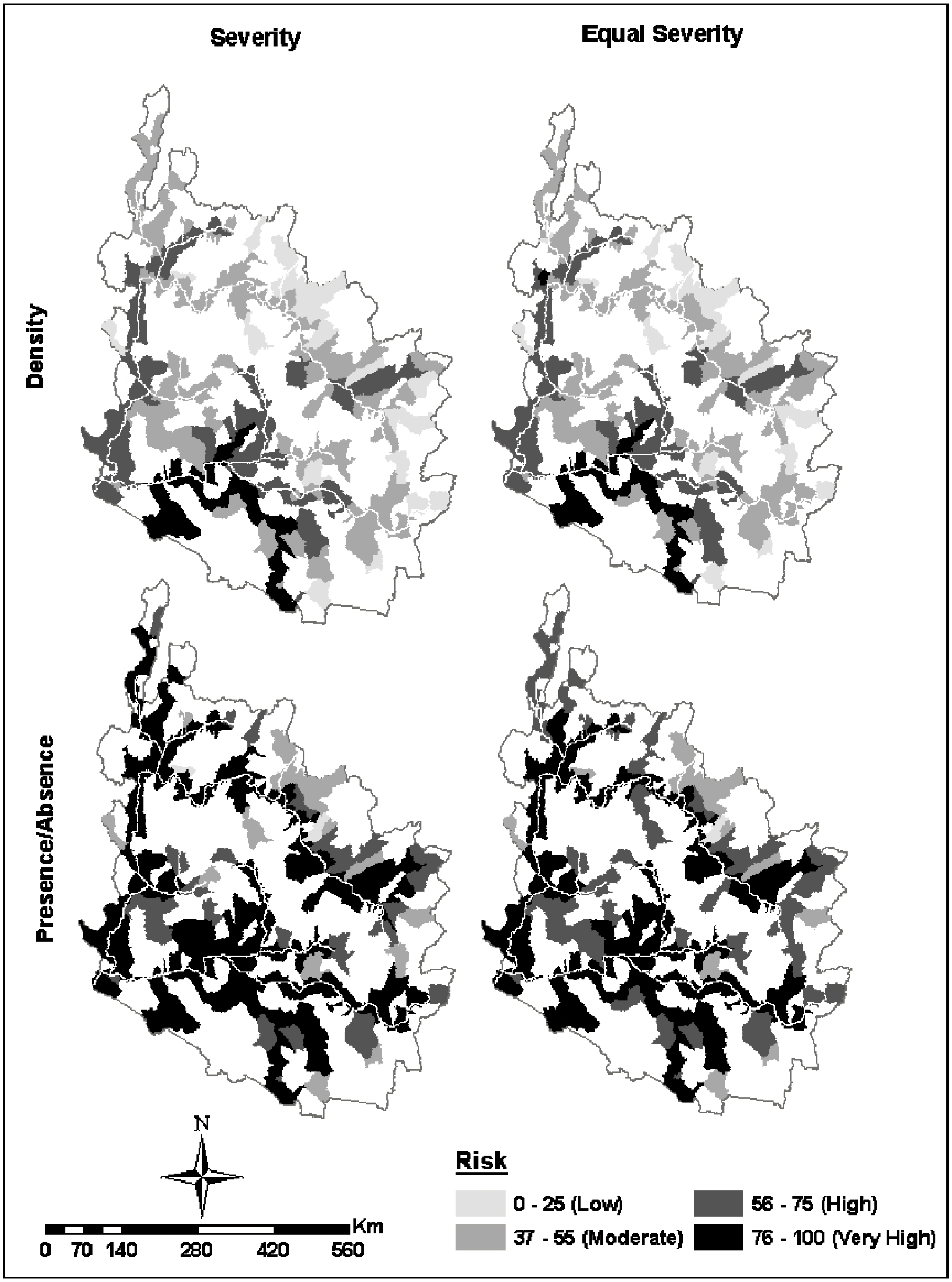

Figure 1.12. Ecological risk values at the upstream of AES scale for the Lower Colorado River Basin. Major rivers are outlined in white. Index methods are noted on the left-hand side and top of maps.

Chapter 2 - Alteration of flow regime and its effects of fish assemblages in the Lower Colorado River Basin

Abstract

We determined hydrologic alteration of 33 different hydrologic variables using current (1986-2006) and historical US Geological Survey discharge data for 48 gage stations within the Lower Colorado River Basin (LCRB) to understand associations with landscape-level influences and fish assemblages. The Indicators of Hydrologic Alteration (IHA) software was used to calculate hydrologic alteration values using the range of variability approach (RVA). Landscape-level sources of hydrologic alteration analyzed include percent agriculture, urban, forest and scrub as well as densities of roads, canals and dams. Life history traits of fish assemblages near each gage station were summarized by presence/absence of current (1980-2006) species records. Hydrologic alteration did not differ among basins due to high variability among and within basins ($p>0.14$). Percent agriculture was positively related with indices of alteration of low flows ($r=0.402$, $p=0.006$) while forested land covers was negatively related with alteration of low flow events ($r=-0.384$, $p=0.008$). Occurrence of piscivorous, nonnative fishes increased with alteration of low flow events whereas occurrence of fluvial dependent fishes preferring rubble substrate decreased with alteration of low flow events ($r=0.64$, $p=0.001$). Our results suggest that land use was associated with hydrologic alteration in the LCRB and these alterations do affect fish assemblage structure. Understanding landscape influences of river flows and biotic responses to modified flow regimes are a necessity in sustainable development of water resources as human populations grow and water resources decrease in the LCRB.

Introduction

Hydrologic variability is a well established habitat characteristic of lotic systems (Poff 1996). Stream flow largely determines the physical instream habitat (i.e. riffle, runs and pools) which strongly affects the aquatic (Gorman and Karr 1978; Bunn and Arthington 2002) and riparian communities (i.e., vegetation, birds, insects, small mammals; Busch and Smith 1995; Richter and Richter 2000). Native flora and fauna have evolved to natural flow regimes via adaptations in life history, morphology and behavioral characteristics (Lytle and Poff 2004).

Flow regimes are naturally shaped by geographical constraints such as climate, elevation and geology (Poff et al. 1997; Sankarasubramanian et al. 2001). However, human alterations, such as, land use, roads and dam construction, often change the natural flow regime (Schlosser 1991; Changnon and Demissie 1996). Road corridors increase runoff, which can subsequently alter channel morphology and increase stream discharge (Forman and Alexander 1998). Dams change upstream waters from predominantly lotic to lentic, reducing flows while increasing depth and width, and impacting downstream waters by altering discharge, often resulting in further alterations in channel morphology, sediment loads, water chemical properties, and thermal conditions (Baxter 1977). Structural changes such as dams and channelization often restrict lateral and longitudinal flow connectivity of river systems as well (Bunn and Arthington 2002). These and other anthropogenic activities can strongly influence fish communities as a result of modified flows.

In the arid southwestern United States, tributaries are primarily fed by summer monsoons and rainstorms, historically creating highly variable hydrologic conditions. Therefore, in the Lower Colorado River Basin (LCRB), the natural flow regime is often characterized by high annual variability, large spring and late summer monsoon-driven floods, and extended periods of low to no flow. However, the LCRB now has numerous dams and diversions, and a rapidly increasing human population, highly regulating the once turbid and flashy system (Mueller and Marsh 2002). These changes in hydrology consequently alter resource timing and availability, and therefore favor species with different life-history traits than historic flow patterns (Marchetti and Moyle 2001; Bernardo et al. 2003).

Historically few species could adapt and survive in the Lower Colorado River Basin and only 31 species are listed as native to the basin. However, in part due to drastic human alterations to the flow regime, approximately half of over 90 introduced non-native species now occur throughout the LCRB (Rinne and Janisch 1995). Within the LCRB, Meffe (1984) found the native Sonoran topminnow (*Poeciliopsis occidentalis occidentalis)* was more behaviorally adapted than the non-native western mosquitofish (*Gambusia affinis*) to persisting through flash flooding in central Arizona. In the San Juan River, native species increased when the river discharge emulated a natural flow regime (Propst and Gido 2004). Since many species synchronize life history traits with specific flow events, altered flow regimes may put these species at risk (Bunn and Arthington 2002; Humphries et al. 2002). The successful establishment of exotic and introduced species is regularly linked to hydrologic alteration (Moyle and Light 1996; Bunn and Arthington 2002), often at the expense of native species. Therefore, native species have to adapt not only to modified flows, but also to competition and predation from non-native species. The native fish of the LCRB are quickly declining, with 25% of fish species listed as threatened or endangered under the Federal Endangered Species Act (Whittier et al. 2006), while non-natives are spreading (Mueller and Marsh 2002; Olden and Poff 2005).

Therefore, the objectives of this study are to 1) quantify hydrologic alteration throughout the LCRB to determine 2) relationships between hydrologic alteration and possible sources of alteration (e.g., land use, dams), and 3) the influence of hydrologic alteration on fish life-history characteristics. As human population grows, climate changes and water becomes more valuable, it is essential to understand how alterations in the natural flow regime impact natural systems and imperiled species.

Methods

Site Description

The LCRB drains 362,750 km² from Arizona, California, New Mexico, Utah and Nevada (Figure 2.1; Blinn and Poff 2005). The LCRB begins below the confluence of Paria River in northwestern Arizona and includes all tributaries flowing into the Colorado River thereafter, encompassing 26,000 km of stream (Blinn and Poff 2005; Olden and Poff 2005). Major tributaries include the Gila, Virgin, Bill Williams and Little Colorado rivers. Major watersheds within the basin include Salt, Verde, Gila, San Pedro, Santa Cruz, Lower Colorado, Virgin, Little Colorado, and Bill Williams (Figure 2.1).

The LCRB is located within the Southwestern United States, which has the fastest human population growth in the nation with a growth rate of 20.7% between 1990 and 2000 (U.S. Census Bureau 2001). Currently Nevada and Arizona are the two fastest growing states (Bernstein 2007b). The human population in Arizona has grown 20.2% from 5,130,632 in 2000 to 6,166,318 in 2006 (http://quickfacts.census.gov/qfd/states/04000.html). Population trends are projected to continue with Arizona's population approaching 11 million by 2030 (U.S. Census Bureau 2004). These population increases exert demands on natural resources in the LCRB, which in turn influence hydrology and aquatic biota.

Data collection

All US Geological Survey (USGS) gaging stations within the LCRB with sufficient current and historic daily discharge records (i.e. 20 years continuous data; Olden and Poff 2003) were used in the analysis. Gage stations within the LCRB were screened for long-term discharge data; current discharge was defined as between the years 1986 and 2006 whereas historic discharge was defined as the earliest twenty years of continuous discharge data, which ranged from 1909-1929 to 1965-1985. Gage stations on canals or those that did not have two periods of 20 years of continuous data were removed from further analysis

Possible sources of hydrologic alteration examined included urban land use (Paul and Meyer 2001; Roy et al. 2005), agricultural land use, dams (Rosenberg et al. 2000; Poff et al. 2007), roads, canals and diversions. Forest and scrub (shrubs <5 m tall with shrub canopy typically greater than 20% of total vegetation) land covers were also included in analysis as these two land cover classes make up the majority of land cover in this region (20% and 70% of the basin's land cover, respectively). Land use data came from the Multi-Resolution Land Characteristic Consortiums 2001 National Land Cover Data (http://www.epa.gov/mrlc/nlcd-2001.html). Dam data was retrieved through the US Army Corps of Engineers National Inventory of Dams database (http://crunch.tec.army.mil/nidpublic/webpages/nid.cfm). Road data came from US Census Bureaus 2006 TIGER files (http://www.census.gov/geo/www/tiger/). Surface water diversions (including rights and claims under public water codes) came from state water agencies. Densities of each of these possible sources of hydrologic alteration were calculated at the watershed scale, representing the entire upstream drainage area. Urban, agriculture, forested and scrub land covers were calculated as percent area of the respective land cover within the watershed. Densities of roads and canals were calculated as total road or canal length (km) divided by land area (km^2) while density of diversions was calculated as number of diversions divided by land area (km^2). To incorporate size of dam, dam density was calculated as storage area (km^2) divided by land area (km^2).

All fish locations were obtained as part of an ongoing study of the LCRB Aquatic Gap Analysis Project (GAP) database which has over 80,000 fish sampling locations and 1.5 million individual fish records within the LCRB from various sources (e.g. Arizona Game and Fish Department, US Geological Survey, Arizona State University, US Forest Service, etc.). Data collected includes geo-referenced point locations verified by agency personnel, species name, site description, and date collected. Fish were sampled with various gears (e.g., hoop nets, dip nets, gill nets, minnow traps, trammel nets, seines and electrofishing) however electrofishing and seining accounted for 88% of all samples with a recorded gear type. Although samples were not collected for the objectives of this study, various studies exhibit how large, historical databases can be effectively utilized (e.g., Fagan et al. 2002; Fagan et al. 2005; Olden and Poff 2005; Olden et al. 2006).

All fish collected from 1980 to 2006 within the catchment (i.e., land draining into a single stream segment) of where a gage station was located were considered representative of the current fish assemblage near that gage station. Sampling events from hatcheries and ponds were removed from analysis as were fish recorded as re-introduced and not established. Numerous sampling events occurred at each gage station throughout 1980 to 2006; therefore presence/absence records for all sampling events within a catchment were compiled. Because of the possible influence of collecting an individual species at just one sampling event (out of numerous events at that catchment), the proportion of sampling events in which individual species were found was calculated by dividing the number of sampling events in which a species is caught by the total number of sampling events occurring within that stream segment. The proportion of sampling events by species was then translated to proportion of sampling events by life history characteristics for each catchment. Because LCRB encompasses a large area in which species assemblages may differ due to biogeographic constraints, life history traits were used to allow functional comparison of sites throughout the basin (Poff and Allan 1995; Scott and Helfman 2001). Life history characteristics thought to be influenced by hydrologic alteration (Poff and Allan 1995; Roy et al. 2005) served as biotic variables used in analysis, including trophic guild (e.g., herbivore/detritivore, invertivore, omnivore and piscivore), fluvial dependence, substrate preference (e.g., rubble, sand, silt/mud and general), velocity preference (e.g., slow, slow to moderate, and moderate to fast), swim factor (ratio of minimum depth of caudal peduncle to the maximum depth of caudal fin) and shape factor (ratio of total body length to maximum body depth). Life history characteristic data primarily came from Olden et al. (2006) with additional species data collected from various sources including peer-reviewed literature and online databases (e.g., FishBase). All life history variables were categorical (e.g., trophic guild) therefore numerical variables (e.g., shape factor) were divided into equal thirds based upon frequency resulting in three categories per variable (e.g. low, medium and high).

Range of Variability Approach

The Indicators of Alteration (IHA) software (Richter et al. 1996; The Nature Conservancy 2006) was used to produce flow variables of magnitude, timing, frequency, duration and rate of change of flow events (Table 2.1). The IHA methodology has been used extensively in rivers including the Colorado River (Richter et al. 1998), Missouri River (Galat and Lipkin 2000; Pegg and Pierce 2002), Henry’s Fork of the Snake River (Benjamin and Van Kirk 1999), Tallapoosa River (Irwin and Freeman 2002), and Illinois River (Koel and Sparks

2002). Hydrologic indices of each gage station were calculated for both historic and current (1986-2006) time periods so an assessment of alteration between time periods can be used.

The range of variability approach (RVA) was used to assess alteration from historic discharge (Richter et al. 1997; Richter et al. 1998). The RVA values range from -1 to 1 in which decreases in variability (from historic discharge) are represented by positive values, increases in variability by negative values and no change in variability is represented by zero. An example of how RVA values are calculated is found in Figure 2.2. The frequency of years of current discharge (1986-2006) which fall between the 25^{th} and 75^{th} percentile of historic discharge (1922-1942) is subtracted from the historic frequency and then divided by the historic frequency. This example uses June discharge at a gage station on the Colorado River within the Grand Canyon, AZ. After Glen Canyon Dam was constructed in 1963, substantial changes occurred in discharge. In this case, the frequency of years in the current time period (1986-2006) which fall between the 25^{th} and 75^{th} percentile of historic equals zero. Therefore the RVA value is calculated as 1, suggesting a high degree of alteration. Absolute values of RVA values were used in analysis so values closer to zero suggest little alteration whereas values closer to one suggest greater alteration. An RVA value is calculated for each of the 33 IHA variables for each gage station.

Data Analysis

A principal components analysis (PCA) was used to reduce the 33 hydrologic alteration (RVA) scores into fewer metrics that described variation in hydrologic alteration across sites (Johnson 1998). Analysis of covariance was used to determine if mean PC scores varied by river basin using year of historic period and stream order as covariates. Spearman rank correlations were used to determine relationships between land cover/land use variables and hydrologic alteration PC scores with significance levels adjusted with a Bonferroni correction for the eight comparisons (α=0.1/8 or α=0.013). Another PCA was used to reduce the dimensionality of the 16 fish life history characteristics into a few metrics that best described life history characteristics among sites. Spearman rank correlations were used to determine associations between hydrologic alteration PC scores and fish life history traits PC scores.

Results

A total of 48 of 294 USGS gage stations within the LCRB had at least 20 years of historical and current gage data and were used in this study (Figure 3). These gage stations were distributed throughout the LCRB, with relatively high numbers (>6) within the Upper Gila, Little Colorado, Verde and Virgin basins (Table 2.2). Several gage stations had numerous temporal breaks in historic records thus making it difficult to assess a static twenty-year historical period for each gage. Therefore, while all current hydrologic records include data from 1986 to 2006, historic records include a continuous set of 20 years of data beginning between 1909 and 1966.

The PCA reduced the set of 33 individual hydrologic alteration variables into a smaller, uncorrelated subset of variables. The first three PCs cumulatively explained 43.8% of the variance among sites (Table 2.3). Component loadings greater than $|0.25|$ were used to interpret PC axes (Fischer and Paukert In press). The first PC axis (PC_1; 24.8%) was primarily related to alteration of low flows and baseflow (annual thirty-day minimum flow, average June flows, and baseflow). The second PC axis (PC_2; 10.5%) was an index of alteration of maximum flows with high loadings of annual maximum one, three, seven, thirty and ninety-day flows. The third PC axis (PC_3; 8.5%) was an index of alteration of rate of change and low flows (number of zero-flow days, Julian date of minimum flow, fall rate, rise rate, reversals and annual one-day minimum flows).

Mean PC scores of hydrologic alteration did not differ by basin due to high variability within basins (Figure 2.4). In general, the mainstem Colorado, Bill Williams, Santa Cruz and Virgin river basins had positive values on PC_1, an index of alteration of low flows and baseflow, while the Little Colorado, Salt Gila, San Pedro and Verde river basins had negative values but means did not differ ($F=1.19$, $df=8, 37$, $p=0.329$). PC_2, an indicator of alteration of the magnitude of maximum flows, had positive values in the Colorado and Little Colorado river basins and negative values in the Bill Williams and Salt river basins and near zero in other basins, but means did not differ ($F=1.55$, $df=8, 37$, $p=0.174$). A second axis signifying alteration of low flows and rate of change, PC_3, had positive PC scores in the Santa Cruz, Little Colorado and Bill Williams river basins and negative PC scores in the Virgin River Basin and near zero scores in other basins, but means did not differ ($F=1.66$, $df=8, 37$, $p=0.141$). Based on the mean value of the first three hydrologic alteration principal components, the five most altered gage stations include the Santa Cruz River near Laveen, AZ, the Muddy River near Moapa, NV, the Colorado River near the Grand Canyon, AZ, the Gila River at Kelvin, AZ and the San Pedro River at Charleston, AZ (Figure 2.3). The five least altered gage stations include the New River near Rock Springs, AZ, Cherry Creek near Globe, AZ, San Francisco River near Reserve, NM, Aravaipa Creek near Mammoth, AZ, and the Verde River near Clarkdale, AZ (Figure 2.3).

Sources of hydrologic alteration had few significant correlations with PC scores at the watershed scale. The percent agriculture in the watershed increased with PC_3 ($r=0.402$, $p=0.006$), while percent forest decreased with PC_3 ($r=-0.384$, $p=0.008$; Figure 2.5). Therefore agriculture was positively and forested land cover negatively related to alteration of low flows at the watershed scale. No other land cover or stressor variables had a significant relationship ($p>0.10$) with any of the hydrologic alteration principal components.

Of the 48 gage stations with at least 20 years of current and historical flow data, 23 had fish records within the same catchment (Figure 2.3). The gage stations with fish were distributed throughout the basin with at least three in the Salt, Verde and Upper Gila River Basins (Table 2.2). Of the 36 total species collected from 1980-2006 throughout all basins, 18 were native and 18 were nonnative (Table 2.4). The number of sampling events summarized for each gage station ranged between 1 and 175 with a mean of 17.

The PCA of life history variables reduced the 16 life history characteristics into three axes that cumulatively explained 79.6% of variation among sites (Table 2.5). Component loadings greater than $|0.25|$ were used to interpret PC axes. The first fish life history trait PC (PC_{LH1}; 43.7%) had high loadings of fishes that had both high and low swim factors, high shape factors, were considered herbivorous/detrivorous, omnivorous, or invertivorous and preferred slow currents. Fish species scoring high on this principal component included fathead minnow (*Pimephales promelas*) and common carp (*Cyprinus carpio*). Components which scored high on the second PC (PC_{LH2}; 23.9%) included piscivory, low shape factors and preference of silt or mud substrate. Components scoring low on PC_{LH2} included fluvial dependence, preference of rubble substrate and moderate to fast currents. Fish species associated with high scores on PC_{LH2} included flathead catfish (*Pylodictis olivaris*), channel catfish (*Ictalurus punctatus*) and largemouth bass (*Micropterus salmoides*); fish species associated with low scores on PC_{LH2} included rainbow trout (*Oncorhynchus mykiss*), brown trout (*Salmo trutta*), speckled dace (*Rhinichthys osculus*) and loach minnow (*Rhinichthys cobitis*). Component loadings which scored high on the third PC (PC_{LH3}; 12.0%) included preference of sand or general substrates and slow to moderate currents. Fishes that scored high on PC_{LH3} included red shiner (*Cyprinella lutrensis*) and longfin dace (*Agosia chrysogaster*).

Fish assemblages based on life history traits were related to indices of hydrologic alteration related to low flows. The first life history PC was not related to any of the hydrologic alteration principal components ($p>0.16$). The second life history PC was related with the third hydrologic alteration principal component ($r=0.64$, $p=0.001$; Figure 2.6). This relationship suggests alteration of zero flow days, Julian date of minimum flows and fall rate was positively associated with piscivorous fishes with low shape factors which prefer silt or mud substrates such as largemouth bass and flathead catfish. Fishes which are fluvial dependent and prefer moderate to fast currents and rubble substrate, such as rainbow trout, brown trout, speckled dace and loach minnow, were negatively correlated with hydrologic alteration PC_3. No correlations were found between the third life history trait and any hydrologic alteration principal components ($p>0.11$).

Discussion

Alteration by basin

Hydrologic alteration within the LCRB was highly variable within and among basins. The mainstem Colorado, Santa Cruz, Bill Williams and Little Colorado had positive mean scores for at least two of the three principal components analyzed, suggesting these basins tended to have higher alteration, but even within a basin there was high variability of alteration. The mainstem Colorado River is widely known for the alteration of its flow regime due to major dam construction (Patten et al. 2001). The Santa Cruz River Basin is highly influenced from diversions and effluent discharge (Stromberg et al. 2007). The Bill Williams River was a perennial stream for part of its reach before Alamo Dam was constructed in 1968 (Wolcott et al. 1956), but is now ephemeral and flows only in response to precipitation. The Upper Gila River and Salt River basins have negative mean PC scores for all four hydrologic alteration PC axes, suggesting they tended to be less altered systems. The Upper Gila has numerous diversions and groundwater pumping primarily for irrigation, however nine of the ten gage stations within this basin are located upstream of the first major dam (Coolidge). Although the Salt River flow regime is highly regulated due to the Salt River Project (SRP) dams, which supply water to the Phoenix metropolitan area, all gage stations analyzed were located at least 30 km upstream of the first SRP dam and therefore were in the relatively unregulated and unaltered portion of the river (Collier et al. 2000).

Landscape sources of hydrologic alteration

Our results suggest that flow alteration throughout the LCRB was not strongly influenced by a single land use or landscape-level metric analyzed. While stream flow at individual gage stations may be influenced by a single or multiple stressors, we were interested whether any individual stressor had a strong influence on hydrologic alteration throughout the LCRB. Agriculture increased with increased alteration of low flows and rates of change at the watershed scale. Alteration from agriculture is most likely due to water extraction for irrigation as 85% of water in the LCRB is allocated to agricultural and irrigation (Blinn and Poff 2005). Irrigation practices lead to decreased ground water storage (Kjelstrom 1995) and stream flow (Ramireddygari et al. 2000; Haddeland et al. 2006) and our results indicate irrigation practices had impacts on stream flows within the LCRB. Decreased ground water storage and stream flow have substantial influence on baseflow, especially during dry times of the year (June) in which often occurs the annual 30-day minimum flow, and the highest number of days in which flow equals zero. In some cases, diversions and groundwater extractions which exceed recharge rates have caused some perennial rivers to become ephemeral (Wolcott 1956; Stromberg et al. 1996), proving these practices can have lasting effects on hydrology and biota (Harding et al. 1998).

Decreased forest land cover at the watershed scale was associated with increased alteration of low flow and rates of change. Decreases in low flow events may be driving the decline of forests by decreasing the water table from groundwater extraction (Busch and Smith 1995) or by reservoirs which trap sediments and dissolved nutrients that naturally flow downstream resulting in bare saline plains replacing riparian forests and scrubland (Abell et al. 2000). Increases in low flow events may be a result of clearing forests which would have otherwise used water for photosynthesis (Bari et al. 1996).

Dams profoundly alter the habitat of a river and are notorious for their impacts on stream flow, however our analysis showed no relationship between dam density and indices of hydrologic alteration. Richter et al. (1997) indicated impoundments to be one of three significant threats associated with freshwater aquatic ecosystems, primarily due to altered flows, habitat degradation and fragmentation, and blockages for fish migration. Lack of association with dams is likely due to the complex manner in which dams are managed. There are a number of different variables that could be used to assess the size of dams, including dam height and hydraulic residence time (defined as the ratio of dam storage volume [m^3] to its flow-through rate [m^3 per year]; Poff and Hart 2002). However, each dam is managed differently according to its purpose, making it unlikely that a single variable would be able to capture dam complexity.

As the Southwestern U.S. grows, its rapidly increasing population has and will likely continue to lead to increases in urbanized land cover while displacing desert and agricultural landscapes, especially in central Arizona (Jenerette and Wu 2001). Gage stations analyzed had a mean density of 1.3% urban land use within associated watersheds, suggesting relatively little urban land use at these sites. However, St. George (UT), Las Vegas (NV), Phoenix (AZ) and Prescott (AZ) were respectively listed as the 1st, 5th, 10th and 11th fastest growing metropolitan areas within the U.S. in 2007 (Bernstein 2007a). Due to increases in impervious surfaces in urban land covers, runoff increases, lag times are shortened and peak discharges are increased in magnitude, but shortened in duration over naturally pervious land covers (Paul and Meyer 2001). As an area becomes urbanized, water courses get paved over, channelized, rerouted, transformed into storm sewers or completely eliminated (Cairns and Palmer 1995). Impervious surfaces, including roofs, parking lots, roads, shopping malls, and industrial buildings, alter the flow of natural systems considerably. Instead of percolating through the soil to groundwater aquifers or being transpired by vegetation, urban runoff is forced across various surfaces, picking up suspended solids, pesticides, nutrients, oil, and human and animal waste, before entering the waterways (Cairns and Palmer 1995). Reduced pervious surfaces can reduce baseflow discharge as a result (Paul and Meyer 2001). Therefore, as land cover continues to change, these changes will have direct results on flow regimes.

Influence of alteration on fish assemblages

Natural flow regimes are more favorable towards native species, which in response have evolved life history strategies enabling these species to persist (Meffe 1984; Poff et al. 1997; Brouder 2001; Marchetti and Moyle 2001; Bunn and Arthington 2002; Propst and Gido 2004; Propst et al. In press). Natural flow regimes, however, have become highly altered and regulated (Benke 1990; Lytle and Poff 2004). Alteration of natural flow regimes can facilitate the establishment of nonnative fishes which have not evolved under the same natural flow conditions (Bunn and Arthington 2002). Relationships between life history characteristics and hydrologic alteration support evidence that hydrologic regimes are important drivers of biotic communities (Poff et al. 1997; Bunn and Arthington 2002). Our results support the notion that declines in native species as well as increases in nonnative species are, in part, a result of hydrologic alteration. While responsibility of the decline in native fishes in the LCRB has largely been placed on water development and the resulting modifications to the flow regime (Mueller and Marsh 2002), variability among dams and other environmental disturbances (e.g., nonnative fishes) make it difficult to distinguish individual biotic and abiotic effects (Rosenberg et al. 2000).

Our results suggest alteration of rate of change and low flow events influences biotic assemblages. Occurrence of piscivorous fishes preferring silt or mud substrate, slow velocities and having low shape factors (e.g., channel catfish, flathead catfish and largemouth bass) increased when alteration of zero flow days, Julian date of minimum flows and fall rate were greatest. Channel catfish and largemouth bass are ubiquitous nonnative fish species throughout the western U.S. (Schade and Bonar 2005), introduced for the purpose of sport fishing. These fishes were introduced through stocking, but are likely persisting due to altered hydrology and other environmental factors. Such piscivorous fishes, including channel and flathead catfish, have been documented to consume native razorback sucker (*Xyrauchen texanus*) ova and juveniles (Minckley 1983) while largemouth bass has been documented to consume native Colorado pikeminnow (*Ptychocheilus lucius*) juveniles (Marsh and Brooks 1989). These nonnative species can cause drastic declines in the recruitment of native, endemic species and can therefore further detrimentally manipulate biotic communities.

Alteration of rate of change and low flow events was also negatively related to fishes which are fluvial dependent and prefer rubble substrate and moderate to fast velocities. These life history characteristics describe both native and nonnative species found in the LCRB. Speckled dace are widespread throughout the basin (Minckley 1973) while loach minnow are listed as threatened by the Endangered Species Act (U.S. Fish and Wildlife Service 1990).

Rainbow trout were first introduced in the early 1900's and are stocked to maintain a blue-ribbon trophy fishery on the mainstem Colorado River between Glen Canyon Dam and Lees Ferry (Schmidt et al. 1998). Therefore, altered hydrology does not universally benefit nonnatives and disadvantage natives, particularly in tailwater environments such as Lee's Ferry where nonnative salmonids thrive.

The mechanism behind the influence of low flow events on fish assemblages may be related to habitat changes under low flow conditions. With an increased frequency or duration of low flow events, driven by natural climate conditions or diversions of water, riffle habitats disappear first, leaving pool habitats (Propst and Bestgen 1991; Aadlund 1993). Channel catfish, among other nonnative fishes, have been found to be associated with decreased flows and increased pools and less abundant with increased flows (Aadlund 1993; Marchetti and Moyle 2001; Propst et al. in press). Loach minnows are often restricted to riffle habitat (Minckley 1973; Rinne 1989; Propst and Bestgen 1991) where dewatering and impoundments threaten local populations (U.S. Fish and Wildlife Service 1990). During drought conditions, declines in loach minnow abundance associated with increased pool habitat have been documented (Propst et al. in press). Speckled dace typically are found below riffles and often have low abundance or are completely gone during years of low discharge (Minckley 1973; Propst and Gido 2004). Both rainbow and brown trout are dependent upon cold, running waters to complete their life histories (Raleigh 1986; Fausch et al. 2001; Marchetti and Moyle 2001). Abnormally low flows, to the point of no flow, can be destructive to developing embryos in riffle habitat (Raleigh et al. 2001). The timing of low flows has been hypothesized to influence rainbow trout invasion success where low flows occurring after rainbow trout fry emergence are considered beneficial (Fausch et al. 2001). Therefore, the interaction between low flow and creations or destruction of habitats, may be the reason why alteration in low flow events was the most significant relationship with fish life history traits.

Native fishes in the LCRB are highly imperiled as a result of numerous anthropogenic factors, including altered flow regimes. Our results suggest alteration of flow regimes throughout the LCRB is not strongly linked to any single threat; however significant relationships do exist with agriculture and forested land cover at the watershed scale. These alterations, primarily related to low flow events, led to shifts in fish assemblages. Human population in the southwestern U.S. is growing rapidly, leading to increased water consumption and extraction and expansions of urban land cover. These factors as well as global climate change projections of a more arid climate in this region (Seager et al. 2007) will likely lead to increases in the alteration of low flow events. Our results suggest increases in the alteration of low flow events lead to increases in predatory species and a continued decline of native fishes.

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Table 2.1. Hydrologic indices calculated using Indicators of Hydrologic Alteration software representing all five flow components (in bold), calculated for historic and current (1986-2006) periods of time for 48 gage stations in the Lower Colorado River Basin.

Flow Component	Hydrologic Indices
Magnitude of flow events	
Mean flow conditions	Monthly means: January, February, March, April, May, June, July, August, September, October, November, December
	Base flow
Frequency of flow events	
Low flow conditions	Low flow pulse count
High flow conditions	High flow pulse count
Duration of flow events	
Low flow conditions	Number of zero-flow days, annual minimum of one, three, seven, thirty and ninety day means, Low flow pulse duration
High flow conditions	High flow pulse duration, annual maximum of one, three, seven, thirty and ninety day means
Timing of flow events	
Low flow conditions	Julian date of minimum flow
High flow conditions	Julian date of maximum flow
Rate of change in flow events	Number of reversals, rise rate, fall rate

Table 2.2. Number of total gage stations and gage stations with fish data by sub-basin for data in the Lower Colorado River Basin. Fish data include samples from 1986-2006.

Basin	Gage Stations	Gage Stations with fish data
Bill Williams	2	1
Little Colorado	6	2
Lower Colorado	2	1
Lower Gila	1	1
Salt	4	4
San Pedro	2	2
Santa Cruz	3	1
Verde	7	5
Virgin	9	2
Upper Gila	11	4

Table 2.3. Principal component (PC) loadings from principal component analysis of 33 hydrologic alteration variables from 48 gage stations throughout the Lower Colorado River Basin. Variable loadings with absolute values >|0.25| in bold.

		Hydrologic Alteration		
Flow Component	IHA variable	PC1	PC2	PC3
Magnitude	January	0.22	0.08	0.09
	February	0.17	0.09	0.07
	March	0.14	-0.02	0.06
	April	0.10	-0.07	0.18
	May	0.23	-0.17	0.08
	June	**0.26**	-0.09	-0.13
	July	0.16	-0.18	-0.03
	August	0.19	-0.03	0.11
	September	0.18	-0.13	0.15
	October	0.16	-0.03	-0.01
	November	0.17	0.03	0.17
	December	0.20	0.06	0.03
	Baseflow	**0.26**	0.01	-0.07
Frequency	Low flow pulse count	0.16	-0.09	-0.18
	High flow pulse count	0.13	0.11	-0.07
Duration	Zero-flow days	-0.03	0.17	**0.43**
	1 day minimum	0.22	-0.01	**-0.29**
	3 day minimum	0.24	-0.06	-0.22
	7 day minimum	0.24	-0.06	-0.20
	30 day minimum	**0.26**	-0.13	0.02
	90 day minimum	0.18	-0.21	-0.02
	Low flow pulse duration	0.23	-0.14	-0.05
	High flow pulse duration	0.07	-0.08	0.11
	1 day maximum	0.07	**0.34**	-0.20
	3 day maximum	0.11	**0.42**	-0.13
	7 day maximum	0.08	**0.41**	-0.13
	30 day maximum	0.14	**0.38**	0.04
	90 day maximum	0.16	**0.33**	0.02
Timing	Julian date of minimum flow	0.20	-0.03	**0.35**
	Julian date of maximum flow	0.08	-0.01	-0.01
Rate of Change	Number of reversals	0.08	0.03	**0.31**
	Fall rate	0.03	0.10	**0.33**
	Rise rate	0.08	0.03	**0.31**
	Eigenvalue	8.2	3.5	2.8
	Percent variance explained	24.8	10.5	8.5
	Cumulative variance explained	24.8	35.3	43.8

Table 2.4. Native (N) and nonnative (I) fish species in the Lower Colorado River Basin used in analysis.

Common Name	Scientific Name	Nativity
Longfin dace	*Agosia chrysogaster*	N
Black bullhead	*Ameiurus melas*	I
Yellow bullhead	*Ameiurus natalis*	I
Desert sucker	*Catostomus clarkia*	N
Bluehead sucker	*Catostomus discobolus*	N
Sonora sucker	*Catostomus insignis*	N
Flannelmouth sucker	*Catostomus latipinnis*	N
Little Colorado sucker	*Catostomus sp.*	N
Common carp	*Cyprinus carpio*	I
Red shiner	*Cyprinella lutrensis*	I
Northern pike	*Esox lucius*	I
Western mosquitofish	*Gambusia affinis*	I
Humpback chub	*Gila cypha*	N
Gila chub	*Gila intermedia*	N
Roundtail chub	*Gila robusta*	N
Virgin River chub	*Gila seminude*	N
Channel catfish	*Ictalurus punctatus*	I
Green sunfish	*Lepomis cyanellus*	I
Bluegill	*Lepomis macrochirus*	I
Little Colorado spinedace	*Lepidomeda vittata*	N
Spikedace	*Meda fulgida*	N
Smallmouth bass	*Micropterus dolomieu*	I
Largemouth bass	*Micropterus salmoides*	I
Golden shiner	*Notemigonus crysoleucas*	I
Rainbow trout	*Oncorhynchus mykiss*	I
Fathead minnow	*Pimephales promelas*	I
Woundfin	*Plagopterus argentissimus*	N
Black crappie	*Pomoxis nigromaculatus*	I
Gila topminnow	*Poeciliopsis occidentalis occidentalis*	N
Colorado pikeminnow	*Ptychocheilus lucius*	N
Flathead catfish	*Pylodictis olivaris*	I
Loach minnow	*Rhinichthys cobitis*	N
Speckled dace	*Rhinichthys osculus*	N
Brown trout	*Salmo trutta*	I
Walleye	*Sander vitreus*	I
Razorback sucker	*Xyrauchen texanus*	N

Table 2.5. Principal component (PC) loadings from principal component analysis of 16 life history variables from 24 gage stations throughout the Lower Colorado River Basin. Variable loadings with absolute values > |0.25| in bold.

	Life History Characteristic	PC1	PC2	PC3
Substrate Preference	Percent Sand	0.23	-0.18	**0.31**
	Percent Rubble	0.20	**-0.27**	**-0.32**
	Percent Silt/mud	0.24	**0.29**	-0.24
	Percent General	0.19	0.23	**0.29**
Fluvial Dependence	Percent Fluvial Dependent	0.17	**-0.38**	-0.09
Trophic Guild	Percent Herbivore/Detritivore	**0.28**	-0.21	-0.04
	Percent Invertivore	**0.26**	0.13	**0.33**
	Percent Omnivore	**0.27**	-0.13	**0.28**
	Percent Piscivore	0.05	**0.39**	0.11
Body Morphology	Percent Low Swim Factor	**0.29**	-0.09	0.16
	Percent High Swim Factor	**0.30**	0.07	-0.21
	Percent Low Shape Factor	0.19	**0.39**	**-0.32**
	Percent High Shape Factor	**0.29**	**0.25**	-0.10
Velocity Preference	Percent Slow	**0.26**	**0.26**	-0.23
	Percent Slow to Moderate	0.21	0.18	**0.42**
	Percent Moderate to Fast	0.09	**-0.25**	-0.03
	Eigenvalue	7.9	4.3	2.2
	Percent variance explained	43.7	23.9	12.0
	Cumulative variance explained	43.7	67.6	79.6

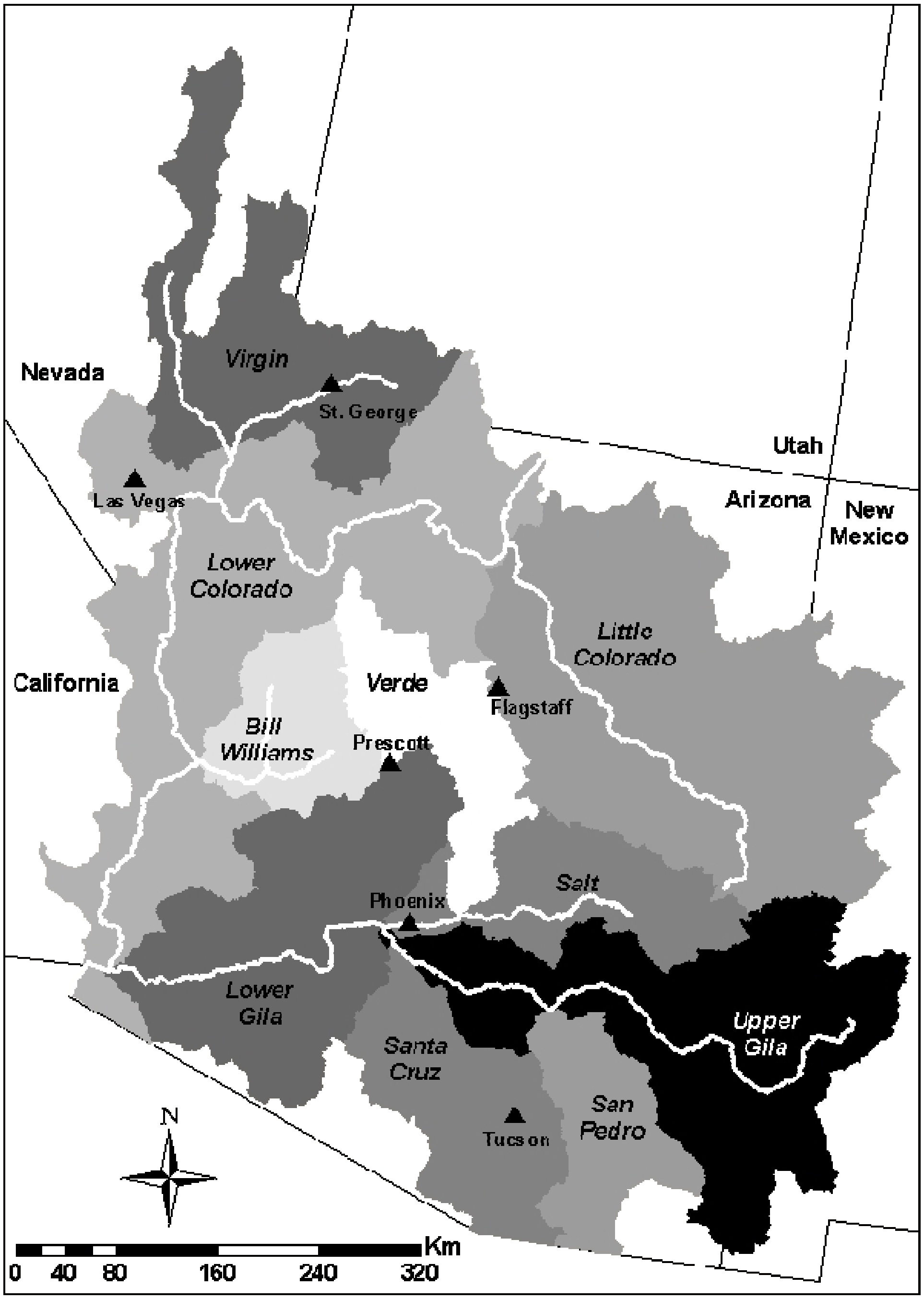


Figure 2.1. The geographic location of the Lower Colorado River Basin. Major watersheds within the basin are denoted by shading and labels, cities by black triangles and major rivers in white.

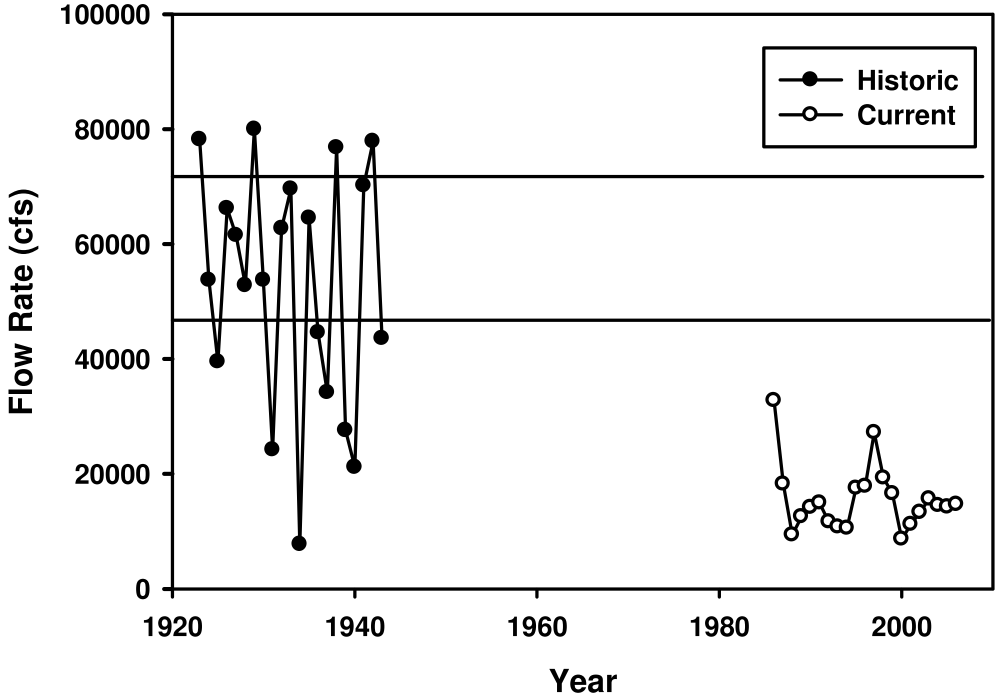


Figure 2.2. An example of the range of variability approach for average June discharge over time at a gage station on the Lower Colorado River within the Grand Canyon. The high bar represents the 75th percentile for the historic period whereas low bar represents the 25th percentile for the historic period. This example shows extreme hydrologic alteration as no current records occurred within the 25th and 75th percentiles of the historic period.

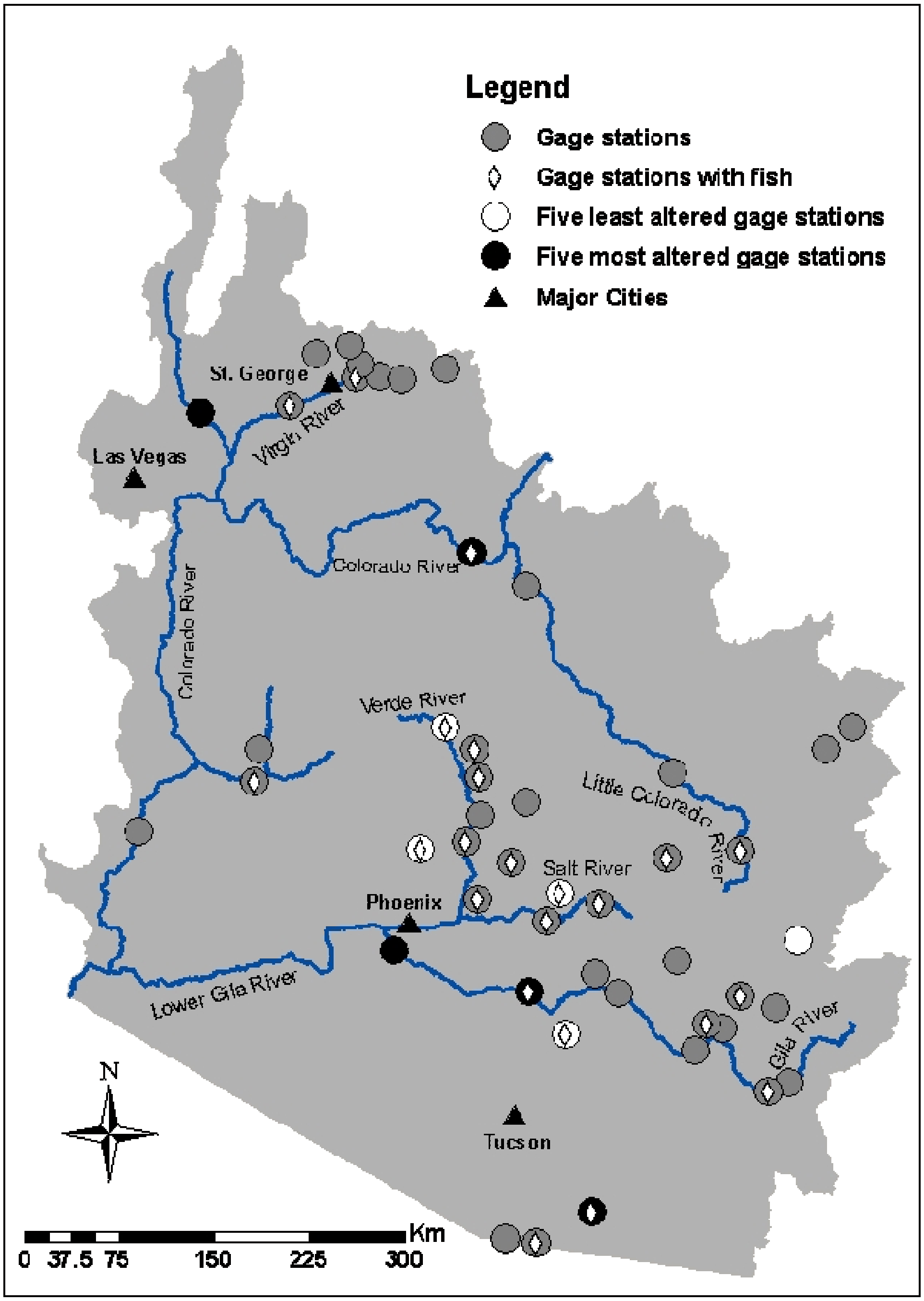


Figure 2.3. The 48 gage stations within the Lower Colorado River Basin with highly greatest altered sites in black and least altered sites in white. Rankings (i.e., low, high) were based on hydrologic alteration principal component scores. The 24 gage stations with fish records are symbolized with diamonds.

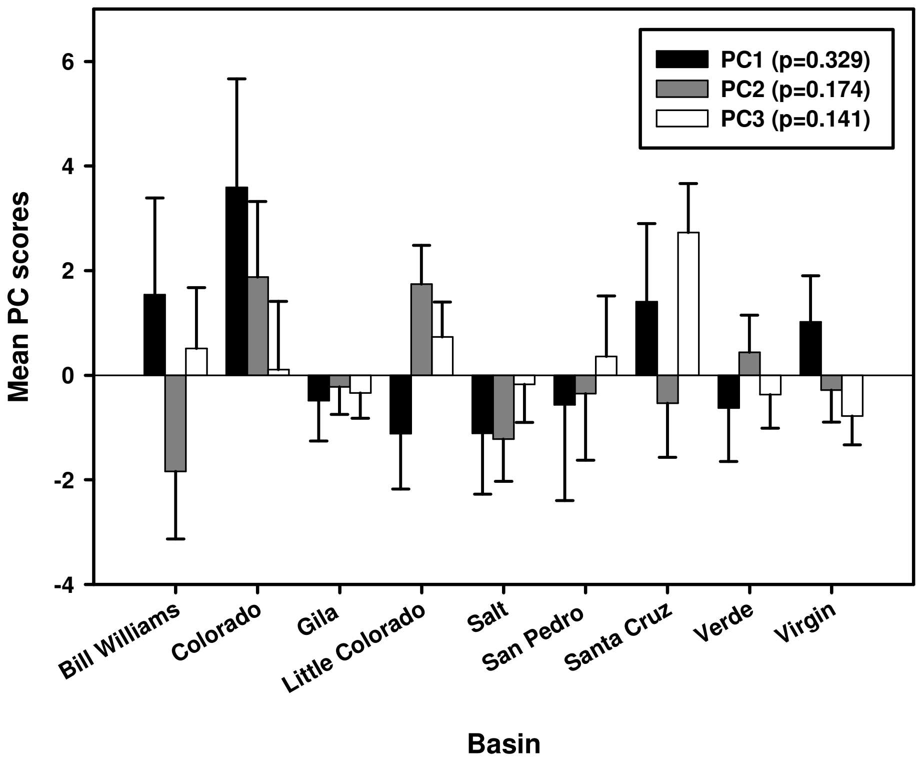
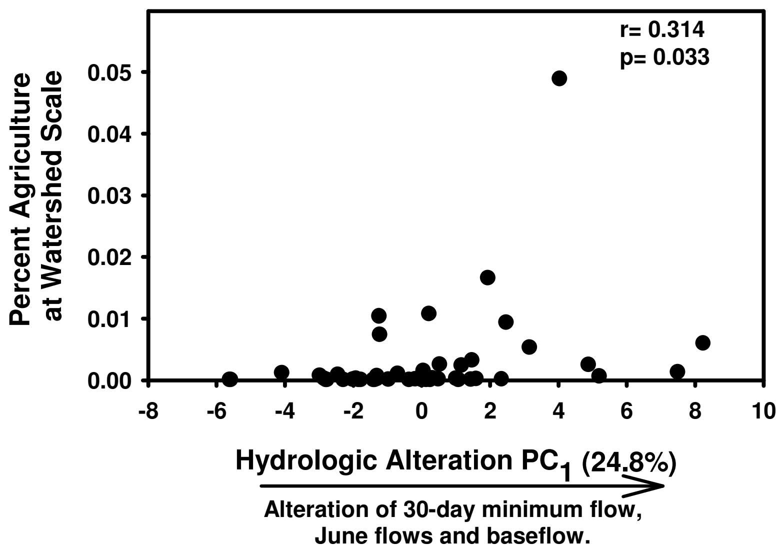
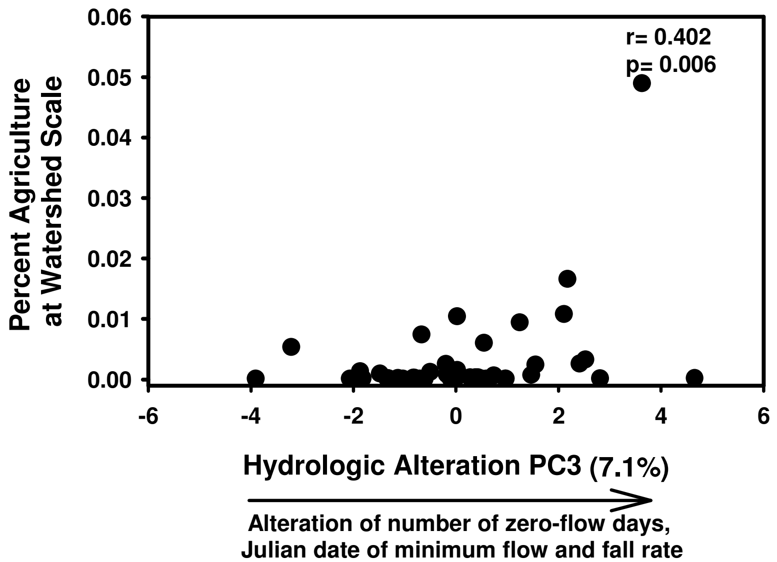


Figure 2.4. Mean hydrologic alteration principal component (PC) scores by sub-basin of the Lower Colorado River Basin. Higher PC scores suggest more alteration of the variables represented by that PC axis. Error bars represent 1 standard error.


r= 0.314
p= 0.033
Percent Agriculture at Watershed Scale
Hydrologic Alteration PC1 (24.8%)
Alteration of 30-day minimum flow, June flows and baseflow.


r= 0.402
p= 0.006
Percent Agriculture at Watershed Scale
Hydrologic Alteration PC3 (7.1%)
Alteration of number of zero-flow days, Julian date of minimum flow and fall rate

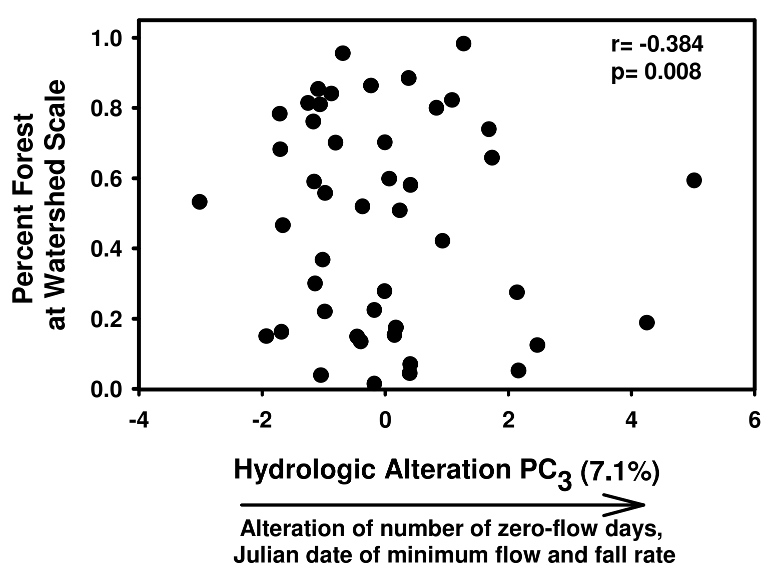

r= -0.384
p= 0.008
Percent Forest at Watershed Scale
Hydrologic Alteration PC3 (7.1%)
Alteration of number of zero-flow days, Julian date of minimum flow and fall rate

Figure 2.5. Spearman rank correlations between hydrologic alteration principal component (PC) axes and landscape-level sources of hydrologic alteration at the watershed scale. Hydrologic alteration PC_1 represents an index of alteration of low flows while PC_3 represents an index of alteration of low flows and rates of change.

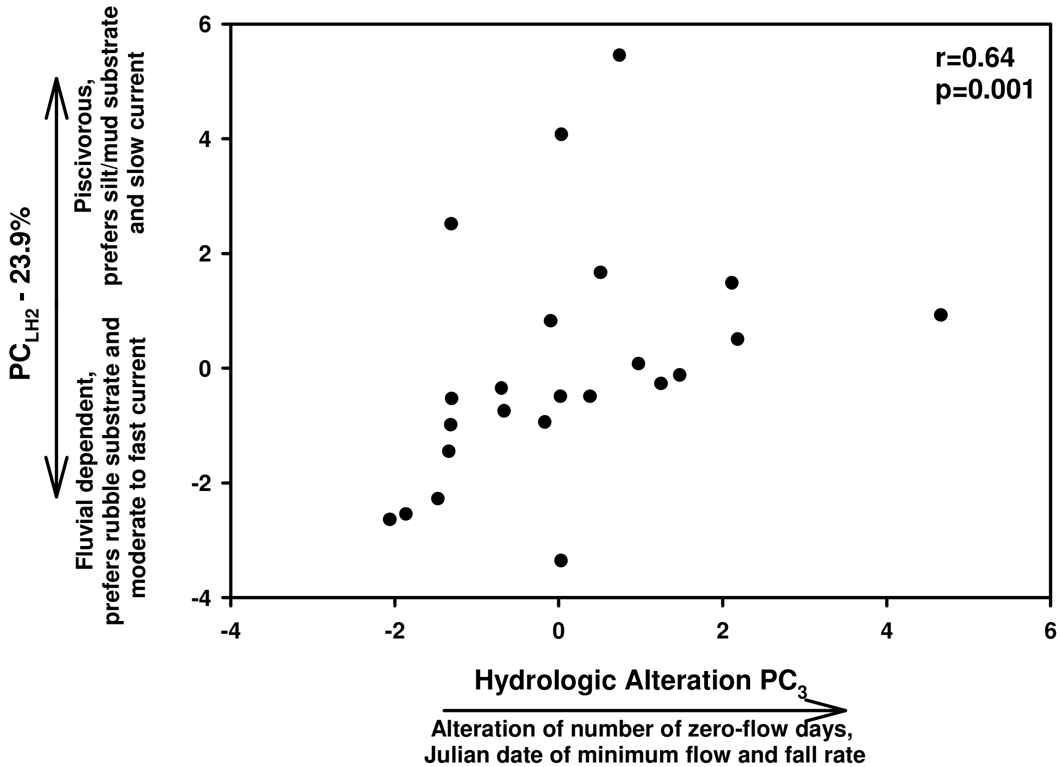


Figure 2.6. Regression results between hydrologic alteration principal components (PC) scores and life history PC scores. Fish species which scored high on PCLH2 included flathead catfish, northern pike and largemouth bass whereas fish species which scored low on the axis included rainbow trout, brown trout, speckled dace and loach minnow.

Appendix A. Description of Indicators of Hydrologic Alteration variables used in this analysis.

Flow component	IHA variable	Description*
Magnitude	January	Mean January flow (cfs)
	February	Mean February flow (cfs)
	March	Mean March flow (cfs)
	April	Mean April flow (cfs)
	May	Mean May flow (cfs)
	June	Mean June flow (cfs)
	July	Mean July flow (cfs)
	August	Mean August flow (cfs)
	September	Mean September flow (cfs)
	October	Mean October flow (cfs)
	November	Mean November flow (cfs)
	December	Mean December flow (cfs)
	Baseflow	Annual 7-day minimum flow/mean annual flow (cfs)
Frequency	Low flow pulse count	Number of annual occurrences during which the magnitude of flow remains below the 25th percentile (low pulse) of all daily values
	High flow pulse count	Number of annual occurrences during which the magnitude of flow remains above the 75th percentile (high pulse) of all daily values
Duration	Zero-flow days	Mean annual number of days having zero daily flow
	1 day minimum	Magnitude of minimum annual flow of 1 day (cfs)
	2 day minimum	Magnitude of minimum annual flow of 3 day (cfs)
	7 day minimum	Magnitude of minimum annual flow of 7 day (cfs)
	30 day minimum	Magnitude of minimum annual flow of 30 day (cfs)
	90 day minimum	Magnitude of minimum annual flow of 90 day (cfs)
	Low flow pulse duration	Duration (days) of annual occurrences during which the magnitude of flow remains below the 25th percentile (low pulse) of all daily values
	High flow pulse duration	Duration (days) of annual occurrences during which the magnitude of flow remains above the 75th percentile (high pulse) of all daily values
	1 day maximum	Magnitude of maximum annual flow of 1 day (cfs)
	2 day	Magnitude of maximum annual flow of 3 day (cfs)

	maximum	
	7 day maximum	Magnitude of maximum annual flow of 7 day (cfs)
	30 day maximum	Magnitude of maximum annual flow of 30 day (cfs)
	90 day maximum	Magnitude of maximum annual flow of 90 day (cfs)
Timing	Julian date of minimum flow	Mean Julian date of the 1-day annual minimum flow
	Julian date of maximum flow	Mean Julian date of the 1-day annual maximum flow
Rate of change	Number of reversals	Number of negative and positive changes in water conditions from one day to the next
	Fall rate	Mean rate of negative changes in flow from one day to the next (cfs/day)
	Rise rate	Mean rate of positive changes in flow from one day to the next (cfs/day)

* cfs= cubic feet per second