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Status of Biotic Integrity, Water Quality, and Physical Habitat in Wadeable East Texas Streams

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LIST OF ACRONYMS

7Q2- lowest seven-day average flow in a two-year period
ALU- aquatic life use
BENT- number of benthic invertivore fish species
CHIR- percent Chironomidae
COLG- percent collector-gatherer functional feeding group
CYPR- number of native cyprinid species
DFFG- percent dominant functional feeding group
DIS- percent of individuals with disease or other anomalies
DO- dissolved oxygen
DOQQ- digital orthophoto quarter-quads
DTAX- percent dominant invertebrate taxon
ECTP- East Central Texas Plains ecoregion
ELM- percent Elmidae
EPT- number of Ephemeroptera, Plecoptera and Trichoptera taxa
ETOL- percent of individuals as tolerant fish species excluding western mosquitofish
GPS- global positioning system
HBI- Hilsenhoff biotic index
HQI- habitat quality index
HYDR-percent Hydropsychidae of total Trichoptera
IBI- index of biotic integrity
INTO- ratio of intolerant to tolerant invertebrate taxa
INTSP- number of intolerant fish species
INVERT- percent of individuals in fish assemblage sample as invertivores
MIN- number of individuals captured per minute of electrofishing
N- density
NOIN- number of non-insect taxa
NONST- percent of individuals in a fish assemblage sample non-native to state
OMNI- percent of individuals in a fish assemblage sample as omnivores
PISC- percent of individuals in a fish assemblage sample as piscivores
PRED- percent predators in an invertebrate sample
SCP- South Central Plains ecoregion
SEIN- number of individuals captured per seine haul
SUN- number of sunfish species
TAXA- taxa richness in an invertebrate sample
TBP- Texas Blackland Prairies ecoregion
TCEQ- Texas Commission on Environmental Quality
TDH- Texas Department of Health
TDS- total dissolved solids
TIND- total number of individuals in a fish assemblage sample
TNOS- total number of species in a fish assemblage sample
TNRCC- Texas Natural Resource Conservation Commission (predecessor agency to TCEQ)
TOC- total organic carbon
TPWD- Texas Parks and Wildlife Department
TSS- total suspended solids
USEPA- United States Environmental Protection Agency
USGS- United States Geological Survey
VSS- volatile suspended solids

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Abstract.— In an effort to evaluate the overall condition of water quality, physical habitat, and fish and invertebrate communities throughout east Texas and determine if differences in those factors could be observed among ecoregions and stream orders and between urban and nonurban sites, we sampled 91 wadeable streams between 1998 and 2000 in three east Texas ecoregions—Texas Blackland Prairies (TBP), East Central Texas Plains (ECTP), and South Central Plains (SCP). Streams were second through fourth order (1:100,000 map scale) and were selected using a randomized systematic design with probabilities set to ensure roughly equal numbers representing each ecoregion and independently, each stream order, and finally to ensure at least 30 reaches in urban or developed areas. Samples were also collected to evaluate the status of mercury contamination in fish tissue across the region. The sites represented a broad range of condition and based upon fish assemblages from all study sites, >83% of stream kilometers were estimated to have an exceptional or high aquatic life use (ALU) compared to >73% for invertebrate kicknet samples. Biotic indicators exhibited a lower proportion of high or exceptional ratings in urban streams compared to nonurban ones. Dissolved oxygen (DO) concentrations were reflective of high or exceptional ALU's for an estimated 71.4% of total stream kilometers. No significant differences in DO were found among ecoregions, stream orders, or between urban and nonurban sites. Nutrient concentrations were less than state screening levels at the majority of sites, and demonstrated no significant differences among ecoregions or between urban and nonurban sites, although phosphorus values were higher in higher order streams. Some physical habitat variables differed significantly between urban and nonurban streams, with the former exhibiting less sinuosity, canopy cover, large woody debris, and natural cover than nonurban ones and more nonagricultural riparian disturbance. Fewer significant differences in physical habitat were observed among ecoregions and most differences in stream order were tied to variables relating to channel morphology and stream size. Urban sites had fish assemblages with fewer total, benthic invertivore, sunfish and intolerant species. Invertebrate metrics were highly variable with respect to significance among ecoregions and stream orders, and between urban and non-urban sites. Hilsenhoff Biotic Index (HBI), percent Chironomidae, and percent dominant taxon were all significantly greater at urban sites. In contrast, ratio of intolerant to tolerant taxa, number of Ephemeroptera, Plecoptera, and Trichoptera (EPT) taxa, and percent Elmidae were all significantly greater at nonurban sites. Benthic subsampling analysis showed that 100 and 200 specimen subsamples produced results that largely were not significantly different from completely picked benthic samples thus indicating that such subsampling protocols may be economically justifiable when used in conjunction with other metrics such as water quality and fish IBIs. Mercury concentrations in fish tissue demonstrated an increasing trend from west to east. In the ecoregion farthest east (SCP), mercury concentrations in whole fish exceeded a predator protection screening level of 0.1 mg/kg in an estimated 87.2% of stream kilometers. Based upon our analysis, we recommend additional refinement and recalibration of assessment tools for biotic integrity and physical habitat.

Biological communities in streams and rivers are associated to varying degrees with distinct habitat types that are ever changing but at equilibrium in natural systems (Karr et al. 1983). Land-use modifications can seriously disrupt that equilibrium and cause marked changes in stream habitats and the corresponding biota (Karr et al. 1983). A result of flood plain development is to reduce the habitat complexity in streams, creating greater homogeneity throughout a system. Loss of stream habitat often occurs through destruction of riparian vegetation with concomitant increases in siltation through bank instability (Karr and Schlosser 1977) and depletion of woody debris within the stream channel. It has long been recognized that stream habitat modifications can result in the reduction or elimination of certain species of fish (e.g., Trautman 1939), aquatic invertebrates (see Gaufin 1973), or other components of the lotic community. Many stream species have narrow habitat requirements and as the system becomes less complex, the fish community may follow a similar trend (Gorman and Karr 1978). While there are always multiple environmental factors that can influence the observed biotic assemblages found in streams, habitat structure and features and physicochemical tolerances have been determined more important than stream order or size in explaining differences (Gorman and Karr 1978; Meffe and Sheldon 1988; Stewart et al. 1992).

Streams in the eastern half of Texas are potentially susceptible to changes in habitat and channel complexity given the fine substrates and dependence upon riparian corridors for bank stability. They are also under pressure for urban and agricultural use given the abundance of water. The majority of the perennial stream kilometers in the state (Gray 1919) are located in three easternmost Texas ecoregions described by Omernik (1987) and Omernik and Gallant (1987; Figure 1)—Texas Blackland Prairies (TBP; ecoregion 32), East Central Texas Plains (ECTP; ecoregion 33), and South Central Plains (SCP; ecoregion 35). The preponderance of perennial streams in these ecoregions, which roughly approximate the subtropical humid climatic region, is not surprising given average annual precipitation totals ranging from 81 to 142 cm per year (Larkin and Bomar 1983). The number of perennial streams and the presence of a large number of population centers has created a particular concern among resource agencies about the potential effects of urban and suburban development on aquatic resources. Urban areas feature a wide variety of disruptions that may be present along a gradient from inner city to suburban and outlying areas (McDonnell and Pickett 1990). About 42% of the state population resides in the three east Texas ecoregions, even though their land area encompasses only 25% of Texas (Kingston 1993). The densest population in the region is found in the TBP with 83.8 persons per km^2 compared to 66 statewide (Kingston 1993).

Texas has 307,686 km of streams and rivers, 64,672 km of which are perennial [Texas Commission on Environmental Quality (TCEQ) 2002]. These aquatic systems are a key element in maintaining the natural heritage of Texas. Cumulatively, streams and rivers assimilate large volumes of wastewater and nonpoint source runoff, and are subjected to varying levels of habitat degradation from urban and agricultural development. Wetlands and riparian corridors associated with the state's streams and rivers are a necessary component in maintaining the integrity of aquatic systems. A 1990 report entitled Region 6 Comparative Risk Project evaluated 22 environmental problem areas, and identified physical degradation as one of the five greatest risks to ecological health in the region [U.S. Environmental Protection Agency (USEPA) 1990].

Texas Parks and Wildlife Department (TPWD) and TCEQ [formerly Texas Natural Resource Conservation Commission (TNRCC)] conducted intensive studies of least-disturbed, wadeable streams across Texas from 1988 to 1990 (Bayer et al. 1992; Hornig et al. 1995; Linam et al. 2002). More than 70% of the 22 reference streams in east Texas would have received a high or exceptional aquatic life use based upon fish community characteristics. The number was 83% for invertebrates (Bayer et al. 1992). Assessments conducted on wadeable streams in eastern Texas slated to receive wastewater discharges (and mostly adjacent to municipalities) found a lower proportion (ranging from 31% in the ECTP and SCP ecoregions combined to 42% in the TBP) with high or exceptional uses based upon fish community evaluations (Linam et al. 2002). Moring (2001) reported similar results for streams near Houston, Texas. Data from these small stream assessments have periodically raised questions about the ability of urban streams to meet designated uses in Texas. Some have argued that many small, wadeable streams that receive wastewater discharges have inherently lower habitat diversity and that since they are often in urban areas, demonstrate habitat and water quality degradation unrelated to the presence of a discharge. This debate became a regulatory issue when USEPA Region 6 did not approve 1995 State of Texas water quality standards that included a presumption that unclassified (=mostly wadeable) perennial streams in east Texas support only intermediate aquatic life uses, rather than a presumed high use for unclassified streams elsewhere in the state.

Historically, natural resource agencies have directed their efforts at protecting aquatic communities through improvements in water quality. Consequently, monitoring of a variety of constituents is widespread in large streams and rivers and to a lesser extent, in small, wadeable streams. Though acknowledged as an important factor in the conservation of aquatic species, little systematic work has been completed in Texas to evaluate the overall status of physical habitats, the extent of degradation, and to relate that information to biological communities. TCEQ has implemented a robust evaluation of habitat in its assessment of waters receiving regulated point source discharges (TNRCC 1999a), but previously most efforts at habitat evaluation were directed at larger streams and rivers, primarily because of concerns about water allocation issues and the potential loss of instream habitat from declining flows.

Given concerns about habitat degradation and the lack of quantification of its status, our objective was to determine the overall condition of aquatic communities, water quality, and stream habitat throughout east Texas and determine if differences in those factors could be observed between urban and nonurban sites. We were also interested in evaluating whether indices such as the TCEQ benthic community index (TNRCC 1999a) and regionally based Index of Biotic Integrity (IBI) proposed by Linam et al. (2002) were sensitive to physical habitat degradation though they were primarily developed with respect to water quality goals.

The widespread presence of mercury in the environment and its bioaccumulation in living organisms has been a subject of increasing concern to natural resource and public health agencies. Most of the attention has centered on potential impacts on human health from dietary consumption of fish. The USEPA reports that 45 states have issued consumption advisories as a result of elevated mercury concentrations in fish tissue (USEPA 2003). Beginning in the early 1990's, the Texas Department of Health (TDH), TCEQ, and TPWD increased sampling efforts to better delineate the extent of mercury contamination in fish. TDH now bans the consumption of fish from one particularly contaminated coastal bay, and has issued advisories for the Gulf of Mexico and ten water bodies in east Texas (TDH 2003). Eisler (1987), Thompson (1996), and Wolfe et al. (1998) have reviewed literature concerning the effects of mercury on wildlife. A concentration of 0.1 mg/kg in fish tissue was recommended by Eisler (1987) as a screening level to protect sensitive, piscivorous mammal and bird species. TDH uses a risk based screening level of 0.7 mg/kg in edible muscle for human health protection.

Mercury is a trace metal occurring naturally in land and water at various concentrations and in different inorganic and organic compounds. Anthropogenic emissions from incinerators, power plants, and other industrial sources contribute an estimated 4 million kg/year mercury to the atmosphere (Mason et al. 1994). This atmospheric mercury, primarily in a gaseous, elemental state, is transported globally and deposited to soils and water bodies far from anthropogenic sources (Fitzgerald 1995). Under certain chemical and biological conditions, inorganic mercury is converted to methyl mercury by bacteria (Gilmour et al. 1992). Methyl mercury is toxic and is readily accumulated by living organisms and biomagnified as it passes to successively higher trophic levels. Fish deposit most of the mercury absorbed from their diet in axial muscle tissue (fillets). Larger, older, piscivorous fish tend to have higher methyl mercury concentrations in their muscle tissue than smaller piscivores or fish at lower trophic levels (MacCrimmon et al. 1993; Wren et al. 1983).

Most of the analysis of mercury in tissue of freshwater organisms has been of fish collected from reservoirs, lakes, and larger rivers. In addition, much of the tissue analysis in Texas has been for public health screening and utilizes larger, legal sized fish, which are commonly consumed, such as largemouth bass *Micropterus salmoides*. Since this project involved sampling fish communities from a variety of wadeable streams in east Texas, it was decided that analysis of mercury in fish would provide useful information on mercury concentrations in piscivorous fish as well as smaller fish at lower trophic levels inhabiting second through fourth-order streams. Also, the sites to be sampled covered a range of pH, dissolved solids, organic carbon, sulfates, and hardness, parameters which have been found by other investigators to influence methylation of mercury and hence concentrations in fish tissue.

METHODS

Site selection.—During 1998-2000, we sampled 91 sites within the TBP, SCP, and ECTP ecoregions. Streams were limited to those within Texas that were wadeable and second through fourth order (Strahler 1957) on a 1:100,000 scale. Sites were selected using a probability-based approach used in the USEPA Environmental Monitoring and Assessment Program. The selection employed a randomized systematic design with a spatial component (Herlihy et al. 2000; e.g., McCormick et al. 2001) and was accomplished by USEPA in Corvallis, Oregon. The sample population was developed from digital U.S. Geological Survey topographic maps (1:100,000 scale) and limited to reaches depicted as perennial. Probabilities were established so that approximately equal numbers of second-, third-, and fourth-order stream sites would be selected, and independently, that approximately one third of the sites would be contained in each ecoregion. The sample selection was intensified to ensure that at least 30 streams were within corporate city limits (=urban), irrespective of ecoregion or stream order. Oversampling was conducted since it was assumed that some sites would not have significant water, would not be wadeable, would be inaccessible, or landowner access would be denied. If less than half the reach was either wetted or wadeable, the site was not sampled. Sites were sampled during the period of May through October (the majority in June through September), which in east Texas corresponds to a period of warm temperatures and low flow. These conditions are consistent with previous ecoregion studies in Texas (Bayer et al. 1992) and are critical for regulatory considerations and observing steady-state conditions. That period is also advantageous biologically, since fish sampling is more efficient during low flows and invertebrate populations are expected to be well developed. Sample locations were transferred to 1:24,000 topographic maps and teams used these maps, county maps, and global positioning system (GPS) instruments to ensure the correct site was being sampled. In some cases, sites were shifted following criteria outlined by Klemm and Lazorchak (1994). Sites were sampled over a reach that was 40 times the mean wetted channel width as measured at the midpoint (Lyons 1992; Klemm and Lazorchak 1994).

Water chemistry.—Water samples were collected at each site following TNRCC (1994) guidelines, preserved as required, and placed on ice until being delivered to the contract laboratory where they were analyzed for total suspended solids (TSS), total dissolved solids (TDS), volatile suspended solids (VSS), total alkalinity, total hardness, chloride, sulfate, orthophosphorus, total phosphorus, ammonia nitrogen, nitrate, nitrite, and total kjeldahl nitrogen, total organic carbon (TOC), and chlorophyll *a*. A sample collection form was used to track each sample with a unique sample identification number and chain of custody. A multiprobe datalogger (YSI 600 XLM or Hydrolab Recorder) was deployed to sample temperature, conductivity, pH, and dissolved oxygen (DO) every 30 minutes for at least a 24-hour period. The dataloggers were suspended where possible in flowing, but non-turbulent water. They were postcalibrated following deployment to check for drift and the data were accepted following a set of guidelines (TNRCC 1999b). Turbidity was measured in the field using a turbidimeter (HF Scientific Model DRT-15CE). Instruments were calibrated according to manufacturer's instructions.

Physical habitat.—The components of the physical habitat survey consisted of a detailed evaluation of channel cross sections, 11 equally spaced per reach; a thalweg profile; a substrate evaluation; a large woody debris characterization; a riparian and canopy characterization; a discharge measurement; and a rapid habitat assessment. Physical habitat evaluation methods generally followed Klemm and Lazorchak (1994), but were modified as described below, particularly in measuring slopes, sinuosity, and width and depth characteristics at transects. To obtain cross sectional profiles, a graduated tagline was strung across each transect and an autolevel and surveyor's rod were used to develop elevational data across the channel. Elevations were collected at the following points (each measurement=vertical): flood prone height, bankfull, bank, edge of water, bed profile (at each break point of 10% change in slope), thalweg, and water surface. Wetted depths and wetted areas represent the difference between water surface and bed profile elevations. Also measured was bank angle from water's edge (in some cases this was calculated from bed cross sections), depth and angle of undercuts, wetted width of channel, and the width of exposed gravel or sand bars in the channel as described in Klemm and Lazorchak (1994). Wetted width and wetted mean and maximum depths were recorded as zero in dry channels. A semi-permanent monument was set at each site to act as a local benchmark and an elevation established using a GPS unit (Trimble GeoExplorer II). Data were post-processed and in instances where discrepancies occurred, topographic maps were consulted for elevations.

In addition to elevations across the channel, an assessment of substrate and instream (=fish) cover was conducted at each station below bankfull. The area evaluated was equivalent to the space half-way between the previous vertical and the next vertical and the area upstream and downstream reflected by the transect. Substrate size classes (Rosgen 1996) were determined through visual observation with primary and secondary substrates being typed along with the degree of embeddedness. Independently, pebble counts of 10 particles (Rosgen 1996) were made along each transect. Visual observation was used to estimate the proportion of the stream bed and banks occupied by specific cover types. The categories follow Klemm and Lazorchak (1994) and estimates were recorded as cover classes(<10%, 10-40%, >40-75%, and >75%). Subsequently, those estimates were weighted based upon the prevalence of a specific cover type along each transect. Canopy cover was measured at each transect using a Model A spherical densiometer, according to the manufacturer's instructions. Four midstream readings were taken (upstream, downstream, left, and right), along with left bank and right bank. Visual riparian habitat and human influence data estimates followed Klemm and Lazorchak (1994), as did thalweg and woody debris measurements. Wetted widths were measured only at cross sections and midpoints between them. Sinuosity was determined by a combination of methods. GPS data from walking the thalweg were supplemented and crosschecked with compass bearings obtained in the field and through the use of overlays on digital orthophoto quarter-quads (DOQQ). Stream gradient (slope) was determined by relating transect elevations from the physical survey and employing the difference in water surface elevations. Gradient was small for many of the surveyed reaches and in some instances, particularly those with no or very slight flow, slopes between some transects appeared negative based upon the surveyed water surface elevations. By convention, these were set to zero. In a few instances, water level fluctuations resulted in negative slopes. Data from the thalweg profile, which was usually completed in a relatively short period of time, were used to correct water surface elevations. In instances where transects were dry, slope was set to zero. Entrenchment ratio was calculated according to Rosgen (1996). Physical habitat data were summarized following Kaufmann et al. (1999), though modified to account for differing data measurement methodology. Riparian habitat and human influence data were subjected to quality assurance and data inconsistencies were resolved as recommended by Kaufmann et al. (1999). In a few instances, photographic documentation was used to account for missing values.

Drainage area upstream of each site was determined by delineating watersheds at each site using National Elevation Dataset digital elevation data. Land use and land cover data and road density were both determined for the site drainage areas. Land use and land cover data were obtained from the United States Geological Survey and based on 30-meter resolution satellite imagery. Road density data were obtained from the Texas Natural Resource Information System and developed from digitized topographic maps (1:24,000 scale). The former was expressed as a proportion with the latter being expressed in road meters per square kilometer (m/km^2).

Fish assemblage samples.—All available habitats in the reach were sampled with a backpack electrofisher (Smith-Root Type 12) and seines, if feasible, though in some instances, both gear types were not used (e.g., conductivity precluded electrofishing or extensive snags prevented effective seining). Electrofishing proceeded in an upstream direction and any species observed but not captured was noted. Seining was primarily used as a complementary technique in habitats where electrofishing was not as effective, such as deep pools where wading with a backpack electrofisher would be difficult or shallow riffles where staking out a seine and kicking the substrate would more efficiently capture organisms washing downstream. Riffles, runs, and small pools were sampled using either a 1.8 m x 1.8 m x 4.8 mm mesh or 4.6 m x 1.8 m x 4.8 mm delta weave mesh seine, depending on the stream width. Occasionally, deep pools were sampled with a 9.1 m x 1.8 m x 6.4 mm delta weave mesh seine with the goal of collecting larger individuals. Sampling was designed to collect a representative sample of species present in their relative abundances. Minimum total sampling time at each site was 45 minutes and ranged up to 275 minutes, depending on stream size and complexity. All fishes were examined for external deformities, lesions, and tumors and a range of total lengths was generated for each species. Most fishes were preserved in 10% formalin and transported to the laboratory for positive identification. Some larger or easily identified individuals were released. Taxonomic references included Hubbs et al. (1991), Robison and Buchanan (1988), Pflieger (1975), Moore (1968), and Douglas (1974). Common and scientific names follow Robins et al. (1991). Voucher specimens are deposited in the Texas Natural History Collection, Texas Memorial Museum, University of Texas, Austin.

Mercury in fish tissue.—In the course of fish assemblage sampling, tissue was retained for mercury analysis if fish of adequate size and/or number were available. An attempt was made to collect one individual or composite sample of a top predator such as a *Micropterus* or *Lepisosteus* species for axial muscle (fillet) analysis and one composite of a *Lepomis* species for whole fish analysis. At some sites, target species were not available and other species were substituted. For the composites, the largest fish were selected, as similar in size as possible. Immediately after collection, fish were weighed, measured, and wrapped with aluminum foil (dull side in). The foil was labeled with a chain of custody number, placed in a ziploc plastic bag and placed in ice. All fish handling and tissue preparation followed the published Texas tissue sampling guidelines (TNRCC 1999b). Fish to be filleted were processed upon return to the laboratory, using polyethylene cutting boards, stainless steel knives, and latex gloves. Muscle samples consisting of more than one fish were composited proportionate to the total weight of each fish. After initial processing, samples were transferred to laboratory staff, logged in, and placed in an ultralow freezer until analysis. Analyses were conducted by cold vapor atomic absorption spectrophotometry.

Benthic invertebrate samples.—The benthic invertebrate sampling methods used in this study generally follow Klemm and Lazorchak (1994) and employ a randomized, systematic spatial sampling design to reduce bias. Benthic samples were collected from downstream to upstream at nine of the 11 habitat transects—the ones on either end of the reach were excluded. For each transect, a single benthic sample was collected at river left (1/4 point), middle, or right (3/4 point) with the location alternating in order for each transect. Location of the cross-channel starting point for benthic samples was determined randomly prior to arriving at the study site. Individual samples were collected with a sampling net (0.5 m wide by 0.3 m high; 600-μm mesh) by kicking an area of approximately 1 m in front of the net for 20 seconds and allowing the water current to carry substrate inhabitants and debris into the net. This resulted in a sampling area of approximately 0.5 m^2. In pools with little or no current velocity, the collecting method was changed to where the collector simultaneously kicked the substrate and dragged the net though the sampling area. This likewise resulted in a sampling area of approximately 0.5 m^2. Samples were placed into a wash bucket (500-μm mesh) and rinsed to remove fine sediments. Samples were then placed into individual containers, labeled appropriately, and preserved with 95% isopropyl alcohol. At the end of each sampling day, samples were drained and rehydrated with fresh 95% isopropyl in order to maintain proper preservation.

Snag samples were collected by gathering a variety of woody debris along the entire study reach with all habitat types being represented to the extent possible. Woody debris ideally included aged materials with a rough, irregular surface. Each piece of woody debris collected was immediately placed into a 3.8-liter container. Material was collected until the container was full or nearly so. Snag material was preserved similarly to kicknet samples.

Benthic samples were returned to the laboratory where collected materials were rinsed through a 500-μm mesh sieve. Samples collected in 1998 and 1999 were sorted according to a subsampling program that picked the first 100 and 200 organisms, respectively and then the remainder of the sample in its entirety. This was accomplished by placing the sample contents into a grided pan that was then agitated to ensure a random distribution. Each square of the grid was picked based on *a priori* randomization. Individual squares of the grid were picked until the total organism count reached 100 and 200, respectively. The 200-organism subsample was a cumulative count that included the first and second 100 organisms removed from the total sample, but each subsample was stored and processed separately. The remainder of the sample was picked without regard to the randomized grid. The subsampled portions of each sample were stored in individual vials, labeled, and preserved with 70% isopropyl alcohol.

Snag samples were rinsed into a 500 μm-mesh sieve and then each piece handpicked to remove any attached invertebrates. All invertebrates occurring on the snag were removed and stored in 70% isopropyl alcohol. Snag samples were not subsampled. Densities of invertebrates occurring on snag samples were estimated by placing the debris in a large container filled with a known amount of water and then determining the volumetric displacement of the woody debris. Invertebrate densities were expressed as the number of invertebrates per liter of water displaced.

All invertebrates were identified to the lowest practical taxonomic level and counted. Primary literature used to identify invertebrates included Merritt and Cummins (1996), Pennak (1989), Wiggins (1996), Needham et al. (2000), and Westfall and May (1996). Voucher specimens of invertebrates identified during this study are maintained at the lab building, A.E. Wood Fish Hatchery, Texas Parks and Wildlife Department, San Marcos, Texas.

Data analysis.—Fish community data were summarized using regionally derived IBI metrics (Linam et al. 2002). The metrics are presented in tables 1 and 2. Linam and Kleinsasser (1998) was used to classify fish into trophic and tolerance categories. Hubbs et al. (1991) and Conner and Suttkus (1986) were used to determine native status of fish species. Metrics for benthic invertebrate data were calculated according to TNRCC (1999a) methods for evaluating kicknet samples. Data from all transects at a site were considered a single sample for calculating the invertebrate index. We acknowledge that the index was developed for collecting organisms from flowing water, but was applied to transect based and snag samples in this study that came from riffle, glide, and pool habitats (Table 3). For the purposes of evaluating stream condition from a regulatory standpoint, a multimetric habitat quality index (HQI; TNRCC 1999a) was employed that integrates several measures of channel morphology, substrate type, cover, and human disturbance (Table 4). A rapid habitat assessment was also conducted at each site using an earlier but similar habitat index (Table 5) developed by Twidwell and Davis (1989). Ratings for the rapid index were based upon visual observation and walking the site prior to measurement. IBI, benthic index, HQI scores, and dissolved oxygen concentrations were screened using guidelines to evaluate aquatic life for Texas streams. The Texas Surface Water Quality

Standards (TNRCC 1997) provide a framework for protecting aquatic life in public waters. Depending on the nature of a waterbody and its biota, a stream may be assigned limited, intermediate, high, or exceptional aquatic life and would be afforded varying levels of protection based upon a tiered set of water quality criteria, most principally, dissolved oxygen standards. These levels of aquatic life are termed aquatic life use (ALU) subcategories and their ecological characteristics are defined qualitatively in the Texas Surface Water Quality Standards (Table 6; TNRCC 1997). Water quality data other than dissolved oxygen were compared to screening levels developed for regulatory determinations by TCEQ. For analysis, non-detectable concentrations were set at one half the detection limit.

The sampling design allowed us to estimate, with 95% confidence limits, the proportion of stream kilometers within the target population sampled that ranked in each ALU, exceeded water quality criteria, and exceeded screening levels for mercury. Sites were weighted based upon their probability of inclusion in the sample. The weights were then used along with sample strata to estimate confidence intervals using Horvitz-Thompson variance estimation (Diaz-Ramos et al. 1996). Biological, water quality, mercury, and physical habitat data were compared among ecoregions, stream orders, and between urban and nonurban sites by using a surveymeans procedure (SAS Institute, release 8.02) to calculate weighted means and 95% confidence intervals. Since the data have a survey-sampling structure with unequal probabilities, we concluded that comparing means and their associated confidence intervals was more appropriate than using a parametric or nonparametric hypothesis test (Johnson 1999; Ramsey and Schafer 2002). This approach allowed use of the site selection strata and sample weights. To provide greater resolution in making comparisons in which there was some overlap in confidence intervals, we also generated confidence intervals for differences between the means for each pairwise comparison; if zero was located outside the confidence interval, it was concluded the means were different. No adjustments were made for multiple comparisons given our desire to observe trends in the data among ecoregions and stream orders rather than strict hypothesis testing. Though these corrections decrease the chance of concluding false significance, they also increase the chance of overlooking significance given their conservative nature (Perneger 1998). Kruskal-Wallis non-parametric ANOVA with a Tukey's multiple comparison procedure (Kwikstat, release 4.6; $P<0.05$) was used to assess invertebrate stream quality metrics from benthic and snag habitats, and to assess subsampling efforts for benthic samples. Benthic invertebrate metrics were calculated on the basis of sampling sites rather than for individual transects within sites. Metrics for the 100 and 200 specimen subsampling efforts were also calculated in the same fashion. Benthic densities (numbers/0.5 m^2) reflect the mean benthic density among transects for all sites, and snag densities (number/L) are the means among sites.

Spearman correlations were used to test the relationship of unscored biological metrics to water quality analytes, physical habitat variables, rapid habitat, and HQI scores (McCormick et al. 2001). Only significant correlations are presented in tables ($P<0.05$). For invertebrates, only those habitat correlations occurring for one-half or more of the metrics are shown. Forward stepwise regression was used to evaluate the contribution of invertebrate and IBI metrics to the total index scores. Spearman correlations were used to evaluate relationships between HQI and biological index scores and basinwide land use and anthropogenic indicators including road density and proportions of total forest, pasture, residential, residential plus commercial, and row crop and small grain agriculture. For the purposes of comparing IBI scores to landscape and road density, the scores were standardized using a 12-metric total, which was necessary given that ECTP AND SCP had 12 metrics, whereas TBP had only 11. The relative position of each IBI score from TBP was determined within the appropriate ALU subcategory and then adjusted to a comparable position within the corresponding ALU subcategory for the ECTP and SCP 12-metric IBI.

Mercury concentrations in muscle and whole fish were analyzed separately. Muscle results were compared using the pooled data from all species in order to maximize sample size. Whole fish mercury concentrations were compared using data from all species, and longear sunfish *Lepomis megalotis* only, the species most frequently submitted for analysis. For the sites at which more than one species was submitted for whole fish or muscle analysis, only one was selected for statistical analysis (longear sunfish or closely related species for whole fish, and largemouth bass for muscle). Spearman rank correlation was used for determining relationships between mercury concentrations and fish length and weight (mean values from all fish in a composite) and water quality parameters. Only data from longear sunfish were used in the correlations to remove confounding effects caused by species differences.

Data are available on request from the authors.

RESULTS

Stream network.—During 1998-2000, 172 sites were evaluated for sampling (Figure 2), representing 16,277 stream km. Of that target population, second-order streams accounted for 43% of the stream distance or 6,964 km; third-order, 33% or 5,346 km; and fourth-order, 24% or 3,967 km. Potential target stream kilometers were most numerous in the SCP ecoregion (8,189 km) followed by the ECTP (4,389 km) and TBP (3,699 km). Urban streams were a small proportion of the total population and accounted for 413 km. The goal of sampling 90 sites was achieved, with data being collected at 91, representing an estimated 7,764 stream km. Of the streams that were not studied, the largest proportions were eliminated either because no significant water was found at the site (an estimated 3,108 km) or because landowner permission was denied or landowners could not be contacted (2,905 km). Sites were also excluded from consideration because of map errors (greater than fourth order or "x" site on a contour line), inaccessibility, the lack of a defined channel, or water depth precluded wading. Sites were somewhat evenly distributed among ecoregions, with 32 in the TBP, 26 in the ECTP, and 33 in the SCP. Sites that were represented by second-, third-, and fourth-order streams numbered 30, 27, and 34, respectively. The criterion of at least 30 urban streams was accomplished as 34 were sampled, though 23 were located in the TBP. The disproportionate number of streams in that ecoregion resulted from the influence of the Dallas-Fort Worth and San Antonio corporate limits upon site selection, as the largest number of urban stream kilometers were located in those two metropolitan areas.

Stream condition.—Stream condition was evaluated based upon procedures designed for implementing the Texas Surface Water Quality Standards (TNRCC 1997) and employed IBI and benthic community analysis along with measures of habitat and water quality (primarily dissolved oxygen). Fish community data were collected at all 91 sites. Geographical distribution of ALUs determined from regional IBIs are presented in Figure 3. Based upon IBIs, 20.7% of stream kilometers were estimated to have exceptional aquatic life (Table 7). High quality assemblages represented 63.1%; intermediate, 16.2%; and limited <1%. Urban sites had the only fish assemblages that received a limited ALU, had a greater proportion of intermediate stream kilometers than nonurban, and a much lower proportion of exceptional stream kilometers than nonurban. Among urban sites, a lower proportion of stream kilometers were estimated to score exceptional or high than among nonurban ones (61.2 and 84.4%, respectively). The proportion of high and exceptional ALUs decreased with increasing stream order.

Invertebrate community data were collected at all 91 sites. ALU scores derived from kicknet data (=benthic) are presented in Figure 4. Based upon ALU scores (Table 8), 71.8% of stream kilometers were estimated to have high aquatic life among all sites while nearly 73% of nonurban sites had high ALU. By comparison, intermediate life uses were estimated for 23.6% and 22.6% of stream kilometers for all sites and nonurban sites, respectively. However, for the urban land use category, intermediate life uses were dominant (51.6%) followed closely by high life uses (48.4%). ALUs for all ecoregions studied were consistently dominated by high scores ($\geq$70%), and intermediate life uses did not exceed an estimated 25% of stream kilometers. Similarly, high ALU scores also dominated stream kilometer estimates for all three stream orders studied (>68%). Exceptional and limited ALUs were poorly represented.

No stream kilometers were estimated to have exceptional habitat quality based upon the HQI (Figure 5), whereas 2,245 km were estimated to have high quality habitat, compared to 5,000 km, intermediate, and 520 km, limited (Table 9). Urban sites had a higher proportion of stream kilometers that scored intermediate and limited compared to nonurban sites. Streams in the TBP had a higher proportion of limited quality stream kilometers as ranked by the HQI. The proportion of high quality stream kilometers increased with stream order.

Valid dissolved oxygen data were collected from 88 of 91 sites. The majority of sites had dissolved oxygen concentrations reflective of an exceptional ALU, which implies that 24-hour mean values were greater than or equal to 6.0 mg/L and minima were at least 4.0 mg/L (Figure 6). Exceptional sites represented 61.4% of the stream kilometers (4655 km); high, 10% (756 km); intermediate, 9.8% (743 km); and limited, less than 2% (116 km). Stream kilometer estimates are presented in Table 10. Dissolved oxygen concentrations reflective of high or exceptional ALUs were estimated for 71.4% of all sites, with comparable estimates for urban and nonurban stream kilometers. Nonattaining streams accounted for an estimated 1,311 km and represent those streams with 24-hour means and minima less than 3.0 or 2.0 mg/L, respectively. Spearman correlations between dissolved oxygen and habitat

(Table 11) revealed that sites with low dissolved oxygen tended to be low gradient with little or sluggish flow, predominantly pool habitat with a low width/depth ratio, abundant woody debris, and substrate dominated by silt and clay. Five of the 12 nonattaining streams had no flow, two others had stream discharges of less than 0.0028 m^3/sec, and one had a discharge that could not be measured, but which was estimated at less than 0.014 m^3/sec. Streams with 7Q2 values (lowest average seven-day, two-year flow) less than 0.0028 m^3/sec are considered intermittent for regulatory purposes (TNRCC 1997). Four of the nonattaining streams were urban and eight were rural. Nonattaining streams were equally divided among stream order, with the most streams, seven, falling in the SCP ecoregion, generally in the northeastern corner of the study area.

Water samples were collected at all 91 sites, but due to analytical problems during the first summer of sampling (1998), none of the ammonia nitrogen data for that year were usable. In summer 1999, 27 of the 37 sites sampled in 1998 were revisited and sampled for all four nitrogen parameters. Detection limits for total phosphorus were initially higher than requested, ranging from 0.07 mg/L the first year to 0.02 mg/L the last year, complicating comparisons for that parameter. Stream kilometer estimates for TCEQ nutrient screening levels (statewide 85th percentiles; TNRCC 2002a) are outlined in Table 12. In the TBP, ECTP, and SCP respectively, an estimated 12.3% (173 km), 6.2% (120 km), and 2.4% (107 km) of the stream distance would exceed the state screening level of 0.8 mg/L for total phosphorus (Figure 7). In the TBP, orthophosphate was less than the state screening level of 0.5 mg/L in an estimated 75.5% (1066 km) of the population. In ECTP, 85.6% (1660 km) was estimated below the state screening level, compared to 97.6% (4306 km) in SCP (Figure 8). The state screening level for chlorophyll *a* would be exceeded in 33.8% (478 km) of total stream distance in TBP, 10.6% (205 km) in ECTP, and 15.7% (695 km) in SCP (Figure 9). Nitrite plus nitrate was estimated to exceed the state screening level in 25.6% (362 km) of stream distance in TBP, and in 18.8% (364 km) and 2.5% (110 km) in ECTP and SCP respectively (Figure 10). Ammonia nitrogen was estimated to exceed the state screening level in 1.6% of stream kilometers in TBP, 5.2% in SCP, and 9.8% in ECTP.

Biological measures were in agreement for ALU at 50.5% of the sites, with the fish community measures rating the community higher at 31.9% and benthic indices rating higher than fish at 17.6%. Most of the disparities (82.2%) represented a difference of one ALU (e.g., exceptional versus high). Disparities were evident at 50.0% of sites in TBP, 53.8% in ECTP, and 45.4% in SCP. Nonagreement was 47% for nonurban sites and 53% for urban. Second- and third-order streams had similar percentages of streams showing nonagreement, 43% and 44%, respectively, whereas a greater percentage of fourth-order streams demonstrated disparities—59%. A combined biological ALU was developed for each site using the higher of the benthic or fish rank, since the Texas Surface Water Quality Standards protect existing uses (TNRCC 1997). If the associated ALU determined by the dissolved oxygen concentrations was equal or greater than the biologically determined ALU, then it was concluded that water quality was supportive of that use (Figure 11). When all sites were considered, dissolved oxygen levels were supportive of the uses for an estimated 5,345 stream kilometers (70.5%). The proportion of urban and nonurban stream kilometers with water quality supporting the ALU was equivalent, with 70.4 and 70.5%, respectively. Fewer stream kilometers in TBP had dissolved oxygen levels that were supporting compared to the other two ecoregions (62.2% compared to 79.5 in the ECTP and 69.1 in the SCP). Habitat ALUs (determined using HQI) were in agreement with those developed from fish and benthic samples at 35% and 37% of the sites, respectively. Habitat ALUs were normally lower than either the IBI or benthic ALUs. Of the 59 sites at which fish and habitat were not in agreement, habitat values were higher at only four sites. In comparing the benthic and habitat classes, the latter were higher at only six of 57 sites.

Urban, ecoregion, and stream-order influences on water quality.—Mean 24-hour dissolved oxygen ranged from 0.33 to 9.9 mg/L , and 24-hour minima ranged from 0.09 to 9.28 mg/L . Spatial distribution of dissolved oxygen ALUs from means and minima is presented in figures 12 and 13. No significant differences were noted in comparisons of mean or minimum DO between urban and nonurban sites, among ecoregions, or among stream orders (Table 13). Mean 24-hour water temperatures ranged from 13.4 to 31.6 °C. Maximum temperatures ranged from 14.0 to 34.8 °C. Temperatures at urban sites were significantly higher than nonurban sites. Mean temperatures were significantly higher in TBP and ECTP than in SCP, and maximum temperatures were higher in ECTP than SCP, with TBP falling in between. Mean and maximum temperatures were significantly higher in fourth-order streams compared to second order, with third order falling in between. Mean pH ranged from 5.7 to 9.0 in the overall study area, was significantly higher in urban than nonurban sites, and showed a significant, decreasing trend eastward from TBP to SCP (Table 13). Mean pH was significantly higher in fourth-order streams than in second-order streams, with third order falling in between.

Sulfate, total hardness, and total alkalinity showed significant differences between urban and nonurban sites, with urban sites having higher concentrations for all three parameters (Table 12; figures 14 and 15). Among ecoregions, TDS was significantly higher in TBP and ECTP than in SCP, and chloride (Figure 16) was significantly higher in ECTP than SCP, with TBP falling in between. Sulfate, total hardness and total alkalinity showed a significant, decreasing trend eastward from TBP to SCP. In stream order comparisons, TDS, total alkalinity, and total hardness were significantly higher in third and fourth-order streams than in second. Chloride and sulfate were significantly higher in fourth-order than second-order streams with third-order streams falling in between. Orthophosphate and total phosphorus were significantly higher in third order than second order, with fourth-order streams not significantly different from either. No significant differences were observed among any of the nitrogen species.

Urban, ecoregion, and stream-order influences on physical habitat.—Comparisons of means for a wide array of habitat variables showed that the greatest number of differences were apparent between urban and nonurban sites, but with ecoregion and stream order groupings demonstrating fewer (Table 14). Differences in urban and nonurban streams were apparent in variables relating to channel morphology, canopy, cover, and riparian disturbance. Urban streams were characterized by less sinuosity, canopy cover, large woody debris, natural instream cover, and more nonagricultural riparian disturbance. Both the HQI and rapid habitat index were significantly lower in urban streams than in nonurban ones. Urban streams also had larger substrate types and more fast water habitats, which may reflect ecoregion influence since those were also attributes of the TBP where the most of the urban streams were located. Ecoregion differences were most apparent among measures of substrate size with the ECTP and SCP having a higher proportion of sand and fines. Most of the differences in stream order were predictably tied to variables relating to channel morphology or stream size. Second- and third-order streams tended to be similar with means from fourth-order streams being significantly different. Overall, fourth-order streams had significantly greater drainage area, discharge, width, width-depth product, and less mid-stream canopy. Third- and fourth-order streams also had a higher percentage of fine substrate composed of silt and clay.

HQI representation of habitat.—As noted, none of the 91 sites evaluated in this study received an exceptional ALU based upon the habitat index. HQI integrates nine aspects of stream habitat which are scored and summed to develop an ALU ranking. Though the index is applied statewide with no regional adjustment of scores, data that relate to three of the nine metrics demonstrated significant variation among ecoregions (Table 14). These included the proportions of coarse gravel, natural cover, and riffle habitat, which suggests the potential for a regional bias to the index. However, HQI means of total scores were not significantly different among ecoregions, but the index did detect significant differences between urban and nonurban reaches. HQI was not significantly correlated with a number of landscape measures, including row crop agriculture, residential development, residential and commercial development, and road density. HQI was correlated positively with the proportion of forested land in the drainage (Figure 17) and negatively with the proportion of pasture (Figure 18).

Urban, ecoregion, and stream-order influences on fish community structure.—Linam et al. (2002) developed regional IBIs primarily based upon applicability to water quality determinations. Two different indices were developed that apply to sites in the study area, one for the TBP and another for the ECTP and SCP (tables 1 and 2). The regional indices contain equivalent metrics, except the index for TBP eliminates the number of intolerant species and the criteria for total number of species, number of native cyprinid species, number of benthic invertivore species, and number of sunfish species are lower, whereas the catch rate criteria for seining are higher. Raw values for many of the fish community metrics included in the regional indices were significantly different relative to urban versus nonurban streams, with ecoregion and stream order influences apparent for some parameters (Table 15). Means for the number of sunfish species (SUN), the number of individuals captured per minute of electrofishing (MIN), and percent of individuals as invertivores (INVERT) were different between urban and nonurban sites, but demonstrated no difference among ecoregions or stream orders. Total number of fish species (TNOS), number of benthic invertivore species (BENT), and number of intolerant species (INTSP) had means that were different relative to urban status and ecoregion (TBP $\neq$ ECTP and SCP). Percent of individuals as tolerant species excluding western mosquitofish (ETOL) and number of individuals captured per seine haul (SEIN) had means that differed for land use, ecoregion, and stream order. No differences were observed for number of native cyprinid species (CYPR), percent of individuals as omnivores (OMNI), percent of individuals as piscivores (PISC), and percent of individuals non-native to state (NONST). TBP had a lower proportion of individuals with disease or other anomalies (DIS).

Overall, urban sites had significantly lower ALUs as determined by regional IBIs, fewer total species, and fewer benthic invertivore, sunfish, and intolerant species. Urban sites had higher catch rates and more tolerant individuals. Among ecoregions, no differences in ALU were observed. TBP sites had fewer total, benthic invertivore, and intolerant species, and a lower percentage of individuals with anomalies. TBP and ECTP sites had higher seine catch rates and a higher percentage of tolerant individuals. Second-order streams had a lower seine catch rate and a lower percentage of tolerant individuals.

IBI metric response to water quality and physical habitat.—Significant Spearman rank correlation coefficients between selected water quality and habitat measures (limited to those that appear sensitive to habitat and water quality degradation) and fish community unscored metric values are presented in tables 16 and 17. TNOS, CYPR, BENT, SUN, INTSP INVERT, PISC, SEIN, and MIN are assumed to decrease with degradation, whereas ETOL, OMNI, NONST, and DIS are assumed to increase (Linam et al. (2002). Most of the regional IBI metrics appeared to follow those anticipated response patterns. TNOS correlated poorly with most water quality measures, demonstrating negative correlations with total dissolved solids, sulfate, and mean pH and a positive one for turbidity. With habitat measures, TNOS was positively correlated with sinuosity, the percentage of small substrates, fish cover measures, and both habitat quality indices. TNOS was negatively correlated with bed stability and measures of riparian disturbance. Like TNOS, BENT was positively correlated with turbidity, but negatively correlated with sulfate, total dissolved solids, and mean pH. It was also negatively correlated with measures of nitrogen and temperature. BENT also correlated with many of the same habitat measures as TNOS, but also had positive relationships with canopy cover and large woody debris. Benthic species were negatively correlated with the proportion of riffle and pool habitat, but positively correlated with glides. CYPR was correlated with few water quality and habitat measures, having positive relationships with turbidity, discharge, sinuosity, and both habitat indices. SUN was negatively correlated with dissolved oxygen, nitrates, sulfates, glide habitat, and discharge. SUN was positively correlated with percent pools and fish cover measures. INTSP was negatively correlated to sulfate and mean pH, pools, larger substrates, and nonagricultural disturbances and had a positive relationship with discharge, glide habitat, sinuosity, smaller substrates, canopy, cover, and both habitat indices. This metric was also positively correlated with agricultural disturbance, which may relate to the prevalence of pastureland compared to intense row crop agriculture in the drainage nets of many sites. ETOL had the highest correlations with water quality variables, including positive relationships with ammonia, sulfates, total dissolved solids, nitrogen measures, and suspended solids. Relationships between ETOL and habitat variables were fewer, but included negative correlations with sinuosity, and riparian canopy cover. NONST was negatively correlated with total phosphorus, sinuosity, small substrates, canopy measures, and cover. INVERT was correlated with relatively few measures, positively for turbidity, small substrates, and agricultural disturbance, whereas negative relationships were observed for bed stability and nonagricultural disturbance. OMNI was positively correlated with nitrogen and suspended solids, discharge, larger substrates, and bed stability and negatively correlated with pools, fine substrates, and leaf litter. PISC was positively correlated with total kjeldahl nitrogen, volatile suspended solids, slow water habitat and pools, and various measures of woody debris and cover and negatively correlated with discharge, dissolved oxygen, sulfate, and mean pH. DIS was not correlated to any water quality variable and only one habitat measure, demonstrating a negative relationship to agricultural disturbance. The final metric combines catch rates from seining (SEIN) and electrofishing (MIN) and averages the scores for total number of individuals metric (TIND). The unscored raw values for this metric demonstrated response patterns that were somewhat contradictory to the assumed response. Both measures were negatively correlated with variables that signify increasing habitat complexity, including canopy measures, cover, and sinuosity, which may relate to the increased difficulty of collecting in more complex systems.

Spearman correlations and scatter plots comparing the regional IBIs to various landscape and habitat measures demonstrated predictable trends (figures 19 through 22). Regional IBI scores were positively correlated with HQI scores and the proportion of forest cover in the drainage area. IBI scores were negatively correlated with road density and the proportion of residential and residential and commercial development in a basin. Correlations with row crop agriculture were not significant.

Metric contribution to IBI.—Stepwise regression of the scored metric values against the total index score for both of the regional IBIs indicated that seven metrics explained about 90% of the variation in the data (Table 18). In both indices, richness metrics (e.g., BENT) entered the model early in the analysis. Generally, the trophic metrics OMNI and INVERT explained little of the data variation and were included late in the process. An exception was PISC, which entered both models by the sixth or seventh step.

Urban, ecoregion, and stream-order influences on invertebrate assemblages.—Taxa richness (TAXA) was significantly greater for snag samples collected from nonurban streams compared to urban streams, but no significant difference was observed for benthic samples (Table 19). Densities (N) were not significantly different between urban and nonurban sites for either benthic or snag habitats. Analysis of other invertebrate metrics showed they were highly variable with respect to significance between urban and nonurban systems for both snag and benthic collections. No significant differences were observed among ALU scores, number of non-insect taxa (NOIN), or percent Hydropsychidae of total Trichoptera (HYDR) for benthic or snag habitats. By comparison, significant differences for the other metrics were observed among snag samples in some instances but not among benthic samples, and vice versa. Hilsenhoff Biotic Index (HBI), percent Chironomidae (CHIR), and percent dominant taxon (DTAX)—primarily Chironomidae—were all significantly greater at urban sites. Ratio of intolerant to tolerant taxa (INTO), percent dominant functional feeding group (DFFG), percent predators (PRED), number of Ephemeroptera, Plecoptera, and Trichoptera (EPT) taxa, and percent Elmidae (ELM) all were greater for nonurban sites. In all instances, mean HBI scores across all ecoregions, stream orders, and land use categories fell into either the fair (5.51-6.50) or good (4.51-5.50) categories.

Differences among invertebrate communities in the three ecoregions studied were not pronounced. Although a few significant differences were observed, there was no clear pattern, suggesting there were no biological differences among the samples. No significant differences were observed for HBI, NOIN, and INTO. In addition, few differences were observed among benthic invertebrate community metrics based on stream order, and are probably not biologically significant.

Metric contribution to the invertebrate index.—Stepwise regression of the scored metrics against the total index scores for benthic and snag data demonstrated very similar results. More than 90% of the data variation was explained by eight of the 12 metrics assessed for each habitat type (Table 20). The only differences between benthic and snag data were that TAXA was not included in the equation for the former until late in the analysis compared to DTAX in the latter. However, the point at which the various metrics entered the model was strikingly different for the two habitat types. For instance, EPT was first to enter the model for benthic samples, but this metric did not enter the snag model until 91% of the variation was explained. In comparison, TAXA was the first metric assessed in the snag samples, but did not enter the benthic model until more than 97% of the variation was explained. Similarly, HBI entered the model at 63% variation explained for snags, but much later for the benthic data. Both ELM and HYDR entered both models relatively early.

Invertebrate metrics.—The collector-gatherer functional feeding group was dominant among benthic and snag collections (62 and 64 sites, respectively). Filterer-collector was the next most abundant functional feeding group among these data sets (15 benthic and 13 snag sites). Dominant predator and scraper functional feeding groups did not exceed 12 sites each. The family Chironomidae was the dominant taxon for both benthic (59 sites, 64.8%) and snag (46 sites, 50.5%) samples collected from the streams sampled, respectively. However, 23 genera and one family were represented as dominant taxa among all streams sampled. Riffle beetles *Stenelmis* spp. were the second-most dominant taxa among both benthic and snag samples (7 and 17 sites, respectively). Asian clam *Corbicula fluminea* and the mayfly *Caenis* spp. were dominant taxa in benthic samples at five sites, each. Another riffle beetle *Heterelmis* sp. was the dominant taxon in snag samples at six sites.

Invertebrate metric response to water quality and physical habitat.—Although invertebrate metrics and habitat descriptors produced variable Spearman correlation results, some trends and patterns were apparent (Table 21). In general, invertebrate metrics were not significantly correlated with many of the habitat variables, including mean width and length, pool depth, and substrate types. Such poor correlation with these habitat variables may be, in part, an artifact of the large number of sampling sites in this study that were dominated by pool habitats with fine substrates. However, percent pools of reach was negatively correlated with EPT, INTO, HYDR and ALU, but positively correlated with HBI, CHIR, DTAX, PRED, and NOIN. By comparison, percent glides of reach was negatively correlated with HBI, DTAX, PRED and CHIR, but positively correlated with TAXA, EPT, INTO, HYDR, ELM, and ALU. HQI and rapid habitat scores were positively correlated with ALU scores, N, TAXA, and EPT suggesting that scores for these metrics and respective indices increase mutually. Both HQI and rapid habitat indices were negatively correlated with HBI, DTAX, AND PRED. Drainage area was positively correlated with N, TAXA, EPT, DFFG, INTO, and ALU scores. All of these metrics reflect higher quality habitats as they increase. By contrast, drainage area was negatively correlated with HBI, CHIR, and PRED, all of which suggest disturbance or perturbation as they increase. Stream discharge was positively correlated with most metrics calculated, but was negatively correlated with CHIR, HBI, DTAX, PRED, and NOIN.

Invertebrate metrics including N, CHIR, NOIN, and DTAX were generally negatively correlated with agricultural disturbances. By comparison, metrics that were positively correlated with non-agricultural disturbances include CHIR, DTAX, and PRED. EPT was negatively correlated with non-agricultural disturbances. INTO was negatively correlated with non-agricultural disturbances, but was positively correlated with agricultural disturbances. Cover classes yielded highly variable correlation results. In general, TAXA from snags was positively correlated with most cover classes, whereas benthic densities were correlated negatively with brush, large cover, and large woody cover. Some invertebrate metrics including ELM, DFFG (mostly collector-gatherer), and INTO generally were positively correlated with most cover variables. The majority of invertebrate metrics were negatively correlated with the various canopy cover variables, although a few metrics were positively correlated. This suggests that the observed correlations may not be biologically significant. Similarly, most of the woody debris variables were positively correlated with the metrics although a few were negatively correlated. Correlation of invertebrate metrics with water quality variables showed few clear trends and other comparisons were conflicting between benthic and snag data sets.

Spearman correlations between the invertebrate index and basinwide landscape measures (e.g., percent forest, road density, residential development, and row crop agriculture) demonstrated no discernable trends and were largely not significant.

Invertebrate subsample analysis.—When the subsamples were evaluated in their entirety without regard to categorical descriptors, significant differences among benthic subsampling levels were observed for TAXA, NOIN, and ALU score (Table 22). For benthic density (N) and TAXA, the values for the entire sample data were significantly larger than for either subsampling effort, but the two subsamples were not significantly different. However, for the ALU score, the 100-specimen subsample had a significantly smaller score than that of the completely picked sample, but neither were significantly different from the 200-specimen subsample. Moreover, the overall assessment shows that the completely picked samples yielded a "high" ALU mean score while those of the two subsamples produced only intermediate ALU mean scores. For the 100-specimen subsample, the percentage of sites yielding high, intermediate and limited ALUs was 52.3%, 43.1% and 4.61%, respectively, but no sites yielded an exceptional ALU. By comparison, the 200–specimen subsample yielded 1.5% exceptional ALU while the percentage of sites yielding high, intermediate and limited ALUs was 53.8%, 41.5%, and 3.1%, respectively. The net result of both subsampling efforts is that they produced fewer high ALU sites and more intermediate ALU sites than did the completely picked samples (65.9% high ALU, 30.8% intermediate ALU). There also were some significant differences between metrics calculated for benthic and snag habitats. Significant differences were observed for N, TAXA, EPT, PRED, NOIN, and ALU scores with snag samples having considerably smaller values in all instances compared to benthic samples. Conversely, a significantly higher percentage of Elmidae were found in snag habitat.

Prevalence of mercury in fish tissue.—Collection of fish for mercury analysis was attempted at all 91 sites. Samples for whole fish analysis were collected at 87 of those sites (a total of 100 whole fish samples). Since time constraints did not allow extra effort beyond that expended in fish community sampling to target fish for tissue analysis, piscivorous fish of sufficient size for muscle analysis were not obtained at most sites. A total of 31 muscle samples were collected and analyzed from 30 sites. A total of 10 species were submitted for muscle analysis and 12 species for whole fish.

Mercury concentrations in whole fish ranged from <0.02 to 0.895 mg/kg and concentrations in muscle ranged from 0.052 to 0.866 mg/kg. Weighted means with associated standard errors and statistically significant differences are summarized in Table 13. In whole fish as well as muscle, fish from nonurban sites had significantly higher concentrations than those from urban sites. In muscle and whole fish (all species pooled) mercury concentrations were significantly higher in ecoregion ECTP than TBP and higher in SCP than ECTP, in effect demonstrating an increasing trend eastward (figures 23 and 24). In whole longear sunfish samples, ECTP and TBP were not significantly different but SCP was higher than both. In stream order comparisons, second-order sites had higher values than fourth order in whole fish (all species pooled) and third were also higher than fourth in whole longear sunfish. Mercury in muscle showed no significant differences among stream orders.

Stream kilometer estimates of mercury concentrations in whole fish tissue are shown in Table 23. Muscle sample sizes were too small to allow useful stream kilometer estimates for mercury at predator protection and human health thresholds. Correlations of mercury concentrations in whole longear sunfish with water quality parameters and fish size are summarized in Table 24. Mercury tended to increase with increasing fish size ($P<0.05$). Significant, negative correlations were found between mercury levels and pH, sulfate, total hardness, total alkalinity, total dissolved solids, and total nitrogen.

DISCUSSION

Stream condition.—Lower biotic integrity as reflected by fish assemblages in urban streams concurs with observations by other workers (Weaver and Garman 1994, Wang et al. 1997; Steedman 1988; Moring 2001). Much as Moring (2001) observed in assessing streams in the Houston-Galveston, Texas area, urban streams in this study tended to have more simplified habitat. Though ecoregional influences were apparent for some habitat variables, sinuosity, measures of riparian canopy, and the HQI were reduced in urban areas compared to nonurban ones without respect to ecoregion. Identifying specific disturbance factors that negatively influenced IBI scores was difficult, but negative correlations were observed for species richness, benthic invertivore species richness, and intolerant species richness with nonagricultural disturbance. Clearly, in urban or adjacent areas, habitat modifications may occur through channel rectification, road and bridge construction, suburban development, and increases in impermeable cover. A number of urban reaches in this study appear to have been straightened previously and a few had extensive armored sections reinforced with concrete, with walls and riprap being more prevalent. Channelizing streams removes bends, reducing stream length and increasing stream gradient (Hubbard et al. 1993). The effect is to remove the diversity of depth, velocity, and cover in a reach (Hubbard et al. 1993). The resulting system may have a relatively low biological diversity compared to natural streams (Huggins and Moss 1975). Clearly, while drastic changes such as channel modification can have direct effects on streams, urbanization can have more subtle effects as land use changes and may appear to be a low intensity disturbance (Weaver and Garman 1994). However, urbanization can alter watershed hydrology and influence channel equilibrium and flow dynamics (Karr et al. 1983). In a study of fish assemblages in an urbanizing stream, Weaver and Garman (1994) observed that substrate-oriented species were notably reduced in abundance over time and there was a general shift in community structure at headwater sites toward species characteristic of downstream, higher order reaches.

Despite the observed lower fish assemblage integrity in urban streams, the majority of the estimated stream kilometers in urban areas demonstrated high or exceptional ALUs. Similarly, Matthews and Gelwick (1990) observed that relatively diverse communities could be found in urban streams, particularly given that fish species in southwestern low-gradient streams have evolved in moderately to highly turbid waters with low flows, intense heating, and potentially low dissolved oxygen levels. Water quality was generally good among urban streams in this study and demonstrated no particular trend relative to urban status, though it should be noted that instrument deployments were short term and only conventional parameters were characterized through single grab samples. Ecoregional differences observed for species richness were not unexpected and concur with observations by previous observers (Hubbs et al. 1991, Conner and Suttkus 1986, Linam et al. 2002).

The overall general pattern in ALU designations based on invertebrate data reported here is similar to that reported by other studies that have shown urban streams to generally have a higher proportion of degraded habitat compared to nonurban streams (Jones and Clark 1987; Lenat and Crawford 1994; Lammert and Allan 1999; Gordon and Majumder 2000). Overall, the majority (~70%) of estimated stream kilometers based on streams sampled in this study was found to have a high ALU. This finding is comparable to that reported by Bayer et al. (1992) although that study generally showed a greater percentage of high or exceptional ALU stream segments based on invertebrate data (~83%), which makes sense given that the streams they studied were least disturbed and constitute reference conditions. In this study, urban streams had approximately equal proportions of intermediate and high ALU, though the former was slightly greater. High ALU was clearly dominant for nonurban streams. These collective designations show that streams in urban areas have a degraded condition in comparison to those in nonurban areas. The effects of modified hydrology accompanying urbanization exert the earliest and, at least initially, the strongest deleterious influences on freshwater ecosystems (Horner et al. 1996). Moreover, Barbour (1996) stated that aquatic systems reflect the condition of their watersheds with impacts from different land use patterns becoming cumulative as drainage areas become larger.

Evaluation of regional IBIs.—The regional IBIs developed by Linam et al. (2002) appear sensitive to declines in habitat quality related to urban and suburban development. Though the raw metric scores reflected differences between TBP and the other two ecoregions, the metrics were calibrated by Linam et al. (2002) to adjust for those differences in the pair of applicable regional indices (i.e., one for TBP and another for ECTP and SCP), meaning that the actual metric score represents attainable values within the individual ecoregions. This appeared to be reflected in the regional IBI ALU rankings, which detected differences between urban and nonurban sites, and not ecoregion. Most of the metrics responded as expected to perceived habitat degradation, though a few should be evaluated further and perhaps recalibrated. Two of them, TIND and DIS, are perceived to be responsive to water quality perturbations. Perhaps the most problematic metric was related to the number of individuals collected (MIN and SEIN combined to form TIND). In an overview of the IBI, Karr et al. (1986) suggested this metric could be sensitive to water quality perturbations with the expected response that numbers would decrease as stream conditions degraded. As noted in this study, water quality was generally good at the majority of sites and the raw, unscored catch rates were negatively correlated with measures of increasing channel complexity, cover, and canopy, suggesting that collecting fishes becomes more difficult as the cover, depth, and sinuosity of the stream increases. McCormick et al. (2001) experienced problems calibrating a metric related to catch and eliminated it from the index they developed. Though this metric may require additional calibration, it can be argued that it is an important biological measure that is related to degraded water quality. For instance, several studies have demonstrated a reduction in number of individuals downstream from chlorinated wastewater outfalls (Kleinsasser and Linam 1992; Lewis et al. 1981). Karr et al. (1984) noted a decrease in individuals at two stream sites exposed to municipal effluent, but higher fish abundances at channelized, poor quality sites on the same stream. In this study, little variation was observed in the DIS rankings; however, as noted above, the water quality observed in this study was largely intact. Karr (1981) originally developed the "percent disease individuals" metric to address a prevalence of anomalies that may occur when fish assemblages are subjected to pollutants. These individual condition metrics focus on chronic exposure to chemical contamination and metrics of this nature have been successfully implemented in fish indices (Barbour et al. 1995). Consequently, in evaluations of streams with severe water quality problems, DIS might prove more useful in discriminating affected sites. Raw values for the INVERT metric were correlated to habitat variables, but the scored values were all given the highest ranking in the ECTP and SCP ecoregions, perhaps suggesting that the metric should be recalibrated with different expectation criteria.

Evaluation of invertebrate community index.—The effects of urbanization on stream invertebrate communities was clearly shown in this study by the reduction in EPT and increases in HBI, CHIR, and DTAX for urban streams. Although TAXA and N were not significantly different among urban and nonurban streams, invertebrate densities in urban streams were comprised of a greater percentage of tolerant taxa including Chironomidae and Oligochaeta. Despite the distinct differences observed among most metrics between urban and nonurban streams, ALU scores were not significantly different suggesting that these scores may not be sufficiently sensitive to the impacts of urbanization on stream invertebrate communities. Comparable results were found for the snag samples. Similar to this study, Stepenuck (1999) compared impacts of urban land use on invertebrate communities with samples from snags and riffles in 43 Wisconsin streams. She found that, as watershed imperviousness related to urbanization increased, the number of tolerant taxa and HBI increased thus indicating that stream quality declined with increased urbanization. Stepenuck (1999) also reported that functional feeding group metrics indicated a shift in invertebrate composition from little to highly urbanized sites. For example percent collector-gatherer increased while percent filterer, percent scraper, and percent shredder decreased with watershed imperviousness (Stepenuck 1999). In this study, the collector-gatherer group was dominant at ≥ 62 of the sites sampled, but our finding that the percent DFFG was significantly less for urban streams is in contrast to that reported by Stepenuck (1999). The contribution of the various metrics for benthic and snag habitats analyzed using stepwise regression showed that the individual metrics clearly have different roles in how they affect the model for each habitat type evaluated in this study.

Although some significant differences were observed in invertebrate metrics for ecoregions, the lack of a consistent pattern among those differences suggests they may not be biologically significant. This is particularly true since some of the metrics including TAXA, NOIN, ELM, and INTO demonstrated few if any significant differences among ecoregions. Furthermore, any true differences in metrics may have been masked by the range of stream conditions among the ecoregions where there is no clearly defined break at their respective borders and streams sampled near those borders may exhibit considerable similarity in physical structure and ecological functioning. Differences among stream order likewise were not pronounced and observed differences probably are not significant. This suggests that any differences in benthic and snag habitats among stream orders were not sufficiently different to alter invertebrate community structure.

Rapid bioassessment studies using invertebrates are well known for their variability due to temporal and spatial variation (Hannaford and Resh 1995, Linke et al. 1999, Fries and Bowles 2002). This study is no exception. However, the large sample size employed here suggests that metric selection for streams in eastern Texas should be re-evaluated and revised as appropriate. Some of the metrics used in this study should be interpreted cautiously. For instance, in this study, CHIR and DTAX measured nearly the same thing as the majority of sites sampled had high population densities of these insects. Also, HYDR appears to have a limited role in east Texas streams since this is a dominant trichopteran family in this region. For this metric to be truly useful, it must be modified to include genus level information to distinguish among ecologically tolerant hydropsychids such as *Cheumatopsyche* spp. in comparison to the other commonly occurring, but more sensitive taxa such as *Macrostemum carolina*. Although Hewlett (2000) concluded that genus-level identifications offered no substantial advantage over family-level identifications, the findings of this study suggest otherwise. Concerning the ELM metric, high percentages of a sample in the family Elmidae are considered to reflect stream impairment according to the invertebrate index (TNRCC 1999a). This is based upon the rationale that *Stenelmis* sp. is relatively tolerant to pollution and may become dominant in situations where a moderate tolerance to organic enrichment may confer an advantage (TNRCC 1999a). However, in this study, a higher percentage of Elmidae were recorded for non-urban streams, suggesting this metric may need to be re-evaluated for east Texas streams. Other more sensitive genera commonly collected during this study were *Ancyronyx, Heterelmis, Hexacylloepus, Macrelmis, Macronychus, Microcylloepus, and Neoelmis*. All of these taxa have tolerance values of 4 or less compared to a tolerance value of 7 for *Stenelmis*. An additional metric that may require recalibration for east Texas streams is the HBI. Although significant differences in HBI scores were recorded between urban and nonurban streams for both benthic and snag habitats, overall scores for this metric were surprisingly low in this study. The majority of mean HBI scores fell in the fair category ((5.51-6.50) and a few scores were in the good category (4.51-5.50). No mean HBI scores were in the very good or excellent categories. This suggests that the HBI is too sensitive for east Texas streams under its current configuration (Hilsenhoff 1987), and that some of the taxa designated as tolerant in these systems may be indeed more intolerant than previously thought.

Use of taxa richness as an indicator of stream habitat quality has been critically debated as to its utility (Larsen and Herlihy 1998; Courtemanch 1996; Vinson and Hawkins 1996). However, the literature has generally supported the use of this metric for stream habitat characterization in relation to anthropogenic disturbance, and it is widely used for this purpose (Resh and Jackson 1993; Larsen and Herlihy 1998). In this study, taxa richness by itself was not sufficiently sensitive for detecting differences among land use type, ecoregions, or stream order. However, TAXA, when coupled with other metrics such as EPT and INTO, can allow insight to the relative disturbance of stream habitat and community structure. Taxa richness normally increases with increasing sample size, and the nine benthic samples collected per site in this study likely missed many rare taxa that were present. Although Cao et al. (2001) warned against excluding rare species from bioassessment studies, the underrepresentation of such rare species likely does not substantially alter the outcome of the various percentage-based metrics presented here due to their lack of sensitivity.

With respect to functional feeding group and percent dominant functional feeding group metrics, collector-gatherers were dominant in virtually all east Texas streams regardless of whether or not they are impacted and thus offer little insight into potential anthropogenic disturbance. A similar argument can be made for the number of non-insect taxa present at a site. Scores had little value in distinguishing between urban and nonurban sites suggesting that it may require a finer level of calibration or more restrictive groupings to be effective for its intended purpose. Other metrics did demonstrate utility in distinguishing between urban and nonurban streams. These include EPT, INTO, ELM, CHIR, and HBI. Although ratio metrics typically exhibit inherent variability and therefore may have little discriminatory power with respect to detecting stream quality perturbations, they offer a reasonable basis of comparison when considered with other data such as fish IBIs, water quality, and physical habitat descriptors. Justification of cause-and-effect relationship in studies such as this, i.e., that urban streams are more impacted than nonurban streams, are generally confounded by a lack of randomization and replication (Beyers 1998). However, the combination of most metrics used herein and consideration of water quality data and fish community structure make for a reasonable basis of comparison between urban and nonurban streams.

Moring (2001) compared stream habitat and biological integrity scores computed from 31 stream reaches in southeastern Texas and found that, in general, reaches which generally had higher stream-habitat integrity scores were in drainage areas that were heavily forested and had fewer people per square mile. Urban reaches generally had more simplified stream habitat conditions and lower biological integrity scores (Moring 2001). With respect to invertebrate metrics, Moring (2001) also found that medians for percent Chironomidae and HBI were significantly higher for urban-agricultural areas, and number of people per square mile was negatively correlated with percent EPT taxa and taxa richness. Greater EPT taxa and number of taxa were associated with those reaches and associated drainage areas that are well forested, have more stable riparian zones, and have more in-channel structures such as woody snags and undercut banks. Moring (2001) also noted that benthic invertebrate data typically provide better site-specific information about a site than fish community data by itself. The results of Moring (2001) generally corroborate those of this study. Our finding of a significant difference in HBI between urban and nonurban streams differs from that of Zweig (2000). Zweig (2000) suggested that this index is sensitive only to organic enrichment, the impairment it was created to detect, and not to habitat alteration. However, the HBI appears sufficiently sensitive to physical habitat alteration as well in east Texas streams.

Spearman rank correlations.—The poor correlation of certain habitat variables with invertebrate metrics may be an artifact of the large number of sampling sites in this study that were dominated by pool habitats with fine substrates. Another likely reason was that the cover metrics were expressed as a fraction of the stream reach, which doesn't necessarily imply that transect-based sampling would detect relationships since the cover might not exist where the sample was taken. The positive correlation of stream drainage area with metrics such as TAXA and INTO, and negative correlations with HBI, CHIR, and DTAX suggests that streams with larger drainage areas may be less sensitive to certain anthropogenic disturbance, or at least have a better capacity to buffer such disturbances.

Correlation of metrics with percent pools of reach and percent glides of reach produced nearly opposite results. Because higher HBI scores reflect increasing tolerance of organic pollution by the invertebrate community, the positive correlation of HBI with percent pools of reach but negative correlation with percent glides of reach suggests that high scores of this metric are a result of more tolerant taxa inhabiting pool habitat in east Texas streams. Likewise, the positive correlation of DTAX, CHIR, NOIN, and PRED with percent pools of reach likely reflects the increased abundance of non-insect taxa and Chironomidae that occur in these habitats. The opposite condition was observed for ALU scores and INTO where higher values were more reflective of glide habitat. The positive correlation of EPT, ELM, and HYDR with percent glides of reach is not unexpected since the organisms comprising these metrics favor flowing water habitat.

Habitat quality and rapid habitat indices were positively correlated with ALU scores and negatively correlated with HBI. Both outcomes are expected given that as stream quality improves, HBI should decrease and ALU should increase. Discharge, not surprisingly, was negatively correlated with HBI, CHIR, and DTAX, and positively correlated with EPT, INTO, HYDR, and ELM. Interestingly, the most invertebrate metrics were negatively correlated with the various canopy cover variables. Most of the woody debris variables were positively correlated with the metrics although some were negatively correlated. Similar observations were recorded for cover classes and riparian disturbance variables.

Snag versus benthic samples.—Although snag samples had significantly smaller values for most metrics, the relative similarity between benthic and snag habitat results suggests that snag samples alone could be used to assess stream quality when properly calibrated for ecoregions and when used in conjunction with other components such as water quality and fish IBIs. In this study, snag samples had significantly lower TAXA, PRED, NOIN, and EPT values compared to benthic samples, but ELM and INTO were higher for snag samples. Benthic samples generally yielded higher ALU scores in comparison to the snag samples. Other metrics did not differ significantly between benthic and snag habitat samples. Stepenuck (1999) used discriminant analyses to show that dominant species occurring in riffle and snag habitats differed. Unlike this study, however, Stepenuck (1999) did not find a significant difference between taxa richness from snags and riffles. Stepenuck (1999) concluded that invertebrate communities found in snags could be used to indicate decreased stream quality when compared to communities found in riffles at the same location. Similarly, Hewlett (2000) noted that single habitat sampling was sufficient for biological monitoring indicating that snag samples could be used to adequately characterize habitat quality of a stream, if using finer taxonomic resolution of the invertebrates and calibrating comparisons on the basis of a regional reference standard. Using only snag habitat samples to assess invertebrate community structure in relation to anthropogenic disturbance certainly merits consideration because this approach would not only result in time savings in the field and laboratory, but substantial cost savings as well.

Benthic subsampling.—The analysis of benthic subsampling conducted here clearly shows that both the 100 and 200 subsampling efforts yielded metrics that largely were not significantly different from the completely picked benthic samples. Although some metrics (TAXA, NOIN, and EPT) were under represented in the subsamples, the considerable savings of time, effort, and costs may be sufficient justification for using 100 specimen subsampling protocols, particularly when used in conjunction with other metrics such as water quality and fish IBIs. The use of subsamples in rapid bioassessments has been widely debated in the published literature. For example, Vinson and Hawkins (1996) stated that identifying less than 200 individuals will greatly underestimate the true richness of an assemblage, and taxa richness estimates based on less than 150 individuals may result in loss of sensitivity and the subsequent inability to detect real differences among collections. However, Vinson and Hawkins (1996) also indicated that the best estimate for species richness falls between 300 and 500 individuals and is a range similar to or less than the number of specimens removed from most of the completely picked benthic samples in this study. Barbour and Gerritsen (1996) and Courtemanch (1996) both placed subsampling efforts in context with desired study objectives and appropriate caveats, and they noted that the scientific validity of subsampling depends on the specific study objectives. A fixed count subsampling regime, such as used in this study, generally yields better estimates of numerical species richness than does fixed fraction counts (Barbour and Gerritsen 1996). Species richness generally increases with sample size, but reaches an asymptote between 100 and 900 organisms depending on total overall richness in a sample (May 1975). However, Barbour et al. (1995) reported that a 100-organism subsample is the optimal subsampling size for calculating metrics used to assess the condition of benthic assemblages in Florida. The results of this study likewise indicate that 100-specimen subsampling fractions of benthic samples collected in east Texas can safely be employed to characterize stream habitat quality, particularly if used in conjunction with water quality data and fish IBIs. The primary trade-off of such subsampling is that taxa richness will be under represented, but this deficiency could be augmented with snag habitat sampling or with multihabitat sampling (Turak et al. 1999, Moulton et al. 2002). If no other supporting data are collected, however, then the more conservative approach of basing the invertebrate indices on completely sorted and identified benthic samples should be used to avoid introducing error into the analysis.

Water quality data.—The large proportion of sites with high or exceptional ALUs based on dissolved oxygen is notable, particularly since the measurements were made during summertime, low-flow conditions. In their study of least disturbed streams, Bayer et al. (1992) reported 75% of streams in the same three ecoregions as scoring high or exceptional, with three of 24 sites intermediate and three nonattaining. They reported no significant differences for mean or minimum dissolved oxygen among ecoregions, but observed considerable variation in SCP with 24-hour means ranging from 1.5 to 7.6 mg/L (n=10). TBP demonstrated the least variability, with means ranging from 4.8 to 7.5 mg/L (n=8), whereas ECTP means ranged from 0.3 to 6.9 mg/L (n=6). The relationship of low dissolved oxygen concentrations with low stream gradients and stream discharge was also observed in that study (Hornig et al. 1995).

Using that data, TCEQ implemented dissolved oxygen criteria for nontidal streams in eastern Texas that consider stream flow, bed slope, and canopy (Hornig et al. 1995). The majority of sites sampled had nutrient concentrations less than the state screening level. No significant ecoregional or landuse differences were found with nitrogen or phosphorus parameters, though total phosphorus and orthophosphate were significantly higher in third and fourth order sites. The reason for this is not clear, though it may result from the preponderance of the sites with the highest phosphorus values receiving treated domestic wastewater (TNRCC 2002b) and all but one of those being third or fourth order streams. One rural, fourth-order site in SCP had markedly higher values for nitrogen and phosphorus than the other streams and is listed on the draft Texas 303(d) list for bacteria levels, with nutrients and depressed dissolved oxygen listed as concerns (TNRCC 2002b). Total alkalinity, total hardness, pH, TDS, chloride, and sulfate decreased significantly from west to east, presumably because of soil chemistry characteristics as it is consistent with the gradation of clayey, calcareous soils of the TBP to the more acid, sandy soils of the SCP. The same phenomenon may explain the higher values for these parameters (excluding chloride and TDS) in urban than nonurban areas, since more urban sites were located in the TBP. This geographic trend is also reflected in data from other agencies (USGS 2002; TNRCC 1999c).

Though this study found no significant difference between urban and nonurban sites for many water quality parameters, the data reflect single samples collected during low flow, steady-state conditions. As such, they provide no information on episodic water quality fluctuations that may occur during stormwater runoff events. In a study of the Trinity River basin, Texas, Land and Shipp (1996) reported that total nitrogen and phosphorus concentrations increased with increasing streamflow. Storm runoff events in urban areas contribute high sediment loads to streams (Pitt 1995). The increased organic loading which often accompanies high suspended sediment concentrations in stormwater runoff can result in temporarily depressed dissolved oxygen concentrations, a phenomenon that would not have been observed in this study.

Physical habitat data and HQI.—The fact that no streams in this study received an exceptional ALU for habitat mirrors rankings based upon data from Bayer et al. (1992). In that study, which evaluated only least disturbed sites, 58% would have rated high using the HQI and 42% intermediate, with no exceptional or limited. The lack of exceptional ALUs in either study could suggest habitat degradation at even the best sites, but could also imply that the scoring criteria for the ALUs may need calibration to determine attainable uses in the ecoregions. A positive, significant correlation with the proportion of forest suggests that the index detects higher quality sites at a broad level, though HQI was not correlated with basinwide landscape measures that reflect anthropogenic disturbance (e.g., road density or residential development). This also suggests the potential need for additional calibration. Given the trends of increased canopy, fewer riffles, smaller substrate particle sizes, and more natural cover as one moves from west to east, several metrics are potentially affected by naturally occurring conditions that vary among the ecoregions studied. Regionalization of the index should be evaluated and potentially could be implemented, if justified, with the physical habitat data collected in this study.

Mercury in fish tissue.— The mercury concentrations found in this study fell within the range of other published data. Twidwell (2000) reported mercury concentrations in largemouth bass muscle in east Texas reservoirs ranging from 0.043 to 2.10 mg/kg. Mills and Luedke (2003) found concentrations in largemouth bass muscle from 0.024 to 1.64 mg/kg in east Texas reservoirs. Snodgrass et al. (2000) analyzed mercury in whole fish from southeastern depression wetlands. They reported concentrations ranging from <0.01 to 1.75 mg/kg in lake chubsucker *Erimyzon sucetta,* 0.13 to 1.53 mg/kg in mud sunfish *Acantharchus pomotis*, and <0.01 to 1.90 mg/kg in redfin pickerel *Esox americanus*.

Since the human health screening level is based on concentrations in edible muscle, and only 31 fillet samples were analyzed in this study, confidence intervals were too large to allow useful population estimates for concentrations posing a risk to human health. Estimates of stream distances expected to exceed the predator protection level in whole fish indicate that TBP probably has few streams with concentrations posing a threat to avian predators (4.9 km out of 1272 km total). ECTP has more (845 km out of 1938 km total), and SCP is the highest with most fish predicted to exceed the threshold. It appears that further studies would be advisable in ECTP and SCP to further quantify the risk to wildlife consuming fish in small streams. The higher concentrations found in nonurban streams may be reflective of the large number of urban streams sampled in ecoregion TBP, an area west of the regions of the state which typically show higher mercury concentrations. The higher concentrations found in lower order streams may relate to the greater proportion of streams in SCP being second order.

The positive correlation noted in this study between mercury concentration and fish size was also reported by Twidwell (2000). Mills and Luedke (2003) reported weak correlations between mercury and largemouth bass size, but strong correlations with age, and noted that the relatively narrow range in the size of fish collected may have weakened the size correlation. As larger fish are frequently older, it is generally assumed that they will have had more time to accumulate mercury in muscle tissue than younger fish. Although fish age was not determined in this study, the positive correlation between longear sunfish size and whole body mercury concentration may reflect such a relationship between age and increasing mercury concentrations accumulating during a fish's life span.

In correlations between mercury concentration and water quality parameters, there were some similarities to the reports of other authors. The significant, negative correlations we found between mercury levels and pH, sulfate, total hardness, total alkalinity, and total dissolved solids were also noted by Twidwell (2000). Mills and Luedke (2003) found that in reservoirs with relatively low levels of mercury bioaccumulation, there were significant, negative correlations between mercury concentrations in fish tissue and TDS, total alkalinity, total hardness, and sulfate. A positive correlation was reported with TOC. However, Mills and Luedke did not find these relationships in reservoirs with higher mercury concentrations in fish. Low pH increases solubility of mercury and can increase methylation rates (Xun et al. 1987). Low calcium concentrations (reflected in low alkalinity and hardness) may increase uptake of methylmercury by fish (Rodgers and Beamish 1983).

Recent atmospheric mercury deposition data were compared for monitoring stations in Fort Worth and Longview, Texas (NADP 2003) to evaluate if differing deposition by region might relate to the differences observed in fish tissue concentrations among ecoregions. Atmospheric deposition was not significantly different between the two monitoring sites (paired sample *t* test, P=0.88) though the period of record for Fort Worth is only since August 2001. The extent to which naturally occurring, inorganic mercury from rock and soils may have influenced fish tissue concentrations is unknown.

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TABLE 1. —Index of Biotic Integrity scoring criteria for Texas Blackland Prairies (TBP) from Linam et al. (2002).

Metric	Acronyms	Scoring Criteria		
		5	**3**	**1**
1. Total number of fish species	TNOS		Varies with drainage area	
2. Number of native cyprinid species	CYPR	>3	2-3	<2
3. Number of benthic invertivore species	BENT	>1	1	0
4. Number of sunfish species	SUN	>3	2-3	<2
5. % of individuals as tolerant species (excluding western mosquitofish)	ETOL	<26%	26-50%	>50%
6. % of individuals as omnivores	OMNI	<9%	9-16%	>16%
7. % of individuals as invertivores	INVERT	>65%	33-65%	<33%
8. % of individuals as piscivores	PISC	>9%	5-9%	<5%
9. Number of individuals in sample	TIND			
a. Number of individuals/seine haul	SEIN	>87	36-87	<36
b. Number of ind/min electrofishing	MIN	>7.1	3.3-7.1	<3.3
10. % of individuals as non-native species	NONST	<1.4%	1.4-2.7%	>2.7%
11. % of individuals with disease or other anomaly	DIS	<0.6%	0.6-1.0%	>1.0%

AQUATIC LIFE USE: ≥49 Exceptional; 41-48 High; 35-40 Intermediate; <35 Limited

TABLE 2. —Index of Biotic Integrity scoring criteria for the East Central Texas Plains (ECTP) and South Central Plains (SCP) ecoregions from Linam et al. (2002).

Metric	Acronyms	Scoring Criteria		
		5	**3**	**1**
1. Total number of fish species	TNOS		Varies with drainage area	
2. Number of native cyprinid species	CYPR	>4	2-4	<2
3. Number of benthic invertivore species	BENT	>4	3-4	<3
4. Number of sunfish species	SUN	>4	3-4	<3
5. Number of intolerant species	INTSP	>3	2-3	<2
6. % of individuals as tolerant species (excluding western mosquitofish)	ETOL	<26%	26-50%	>50%
7. % of individuals as omnivores	OMNI	<9%	9-16%	>16%
8. % of individuals as invertivores	INVERT	>65%	33-65%	<33%
9. % of individuals as piscivores	PISC	>9%	5-9%	<5%
10. Number of individuals in sample	TIND			
a. Number of individuals/seine haul	SEIN	>28	14-28	<14
b. Number of ind/min electrofishing	MIN	>7.3	3.6-7.3	<3.6
11. % of individuals as non-native species	NONST	<1.4%	1.4-2.7%	>2.7%
12. % of individuals with disease or other anomaly	DIS	<0.6%	0.6-1.0%	>1.0%

AQUATIC LIFE USE: ≥52 Exceptional; 42-51 High; 36-41 Intermediate; <36 Limited

TABLE 3. — Metrics and scoring criteria for kick samples, benthic invertebrates (TNRCC 1999).

Metric	Acronym	Scoring Criteria			
		4	3	2	1
Taxa richness	TAXA	>21	15-21	8-14	<8
EPT[1] taxa abundance	EPT	>9	9-7	6-4	<4
Hilsenhoff biotic index	HBI	<3.77	3.77-4.52	4.53-5.27	>5.27
Percent Chironomidae	CHIR	0.79-4.10	4.11-9.48	9.49-16.19	<0.79 or >16.19
Percent dominant taxon	DTAX	<22.15	22.15-31.01	31.02-39.88	>39.88
Percent dominant FFG[2]	DFFG	<36.5	36.50-45.30	45.31-54.12	>54.12
Percent predators	PRED	4.73-15.20	15.21-25.67	25.68-36.14	<4.73 or >36.14
Ratio of intolerant:tolerant taxa	INTO	>4.79	3.21-4.79	1.63-3.20	<1.63
% of total Trichoptera as Hydropsychidae	HYDR	<25.5	25.51-50.50	50.51-75.50	>75.50 or none
# of non-insect taxa	NOIN	> 5	5-4	3-2	< 2
% collector-gatherers	COLG	8.00-19.23	19.24-30.46	30.47-41.68	< 8.00 or >41.68
% of total number as Elmidae	ELM	0.88-10.04	10.05-20.08	20.09-30.12	< 0.88 or >30.12

AQUATIC LIFE USE: >36 Exceptional; 29–36 High; 22–28 Intermediate; <22 Limited

[1]EPT=Ephemeroptera, Plecoptera, Trichoptera

[2] FFG=Functional feeding group

TABLE 4. —Habitat quality index (HQI) metrics from TNRCC (1999a).

Habitat Parameter	Scoring Criteria			
Available instream cover	**Abundant** >50% of substrate favorable for colonization and fish cover; good mix of several stable (not new fall or transient) cover types such as snags, cobble, undercut banks, macrophytes	**Common** 30-50% of substrate supports stable habitat; adequate habitat for maintenance of populations; may be limited in the number of different habitat types	**Rare** 10-29.9% of substrate supports stable habitat; habitat availability less than desirable; substrate frequently disturbed or removed	**Absent** <10% of substrate supports stable habitat; lack of habitat is obvious; substrate unstable or lacking
	4	3	2	1
Bottom substrate stability	**Stable** >50% gravel or larger substrate, i.e., gravel, cobble, boulders; dominant substrate type is gravel or larger	**Moderately Stable** 30-50% gravel or larger substrate; dominant substrate type is mix of gravel with some finer sediments	**Moderately Unstable** 10-29.9% gravel or larger substrate; dominant substrate type is finer than gravel, but may still be a mix of sizes	**Unstable** <10% gravel or larger substrate; substrate is uniform sand, silt, clay or bedrock
	4	3	2	1
Number of riffles (To be counted, riffles must extend >50% the width of the channel and be at least as long as the channel width)	**Abundant** > 5 riffles	**Common** 2-4 riffles	**Rare** 1 riffle	**Absent** No riffles
	4	3	2	1
Dimensions of largest pool	**Large** Pool covers more than 50% of the channel width; maximum depth is >1 meter	**Moderate** Pool covers approximately 50% or slightly less of the channel width; maximum depth is 0.5-1 meter	**Small** Pool covers approximately 25% of the channel width; maximum depth is <0.5 meter	**Absent** No existing pools; only shallow auxiliary pockets
	4	3	2	1
Channel flow status	**High** Water reaches the base of both lower banks; < 5% of channel substrate is exposed	**Moderate** Water fills >75% of the channel; or <25% of channel substrate is exposed	**Low** Water fills 25-75% of the available channel and/or riffle substrates are mostly exposed	**No Flow** Very little water in the channel and mostly present in standing pools; or stream is dry
	3	2	1	0

Table 4. —Continued.

Habitat parameter	Scoring Category			
Bank stability	**Stable** Little evidence (<10%) of erosion or bank failure; bank angles average <30°	**Moderately Stable** Some evidence (10-29.9%) of erosion or bank failure; small areas of erosion mostly healed over; bank angles average 30-39.9°	**Moderately Unstable** Evidence of erosion or bank failure is common (30-50%); high potential of erosion during flooding; bank angles average 40-60°	**Unstable** Large and frequent evidence (>50%) of erosion or bank failure; raw areas frequent along steep banks; bank angles average >60°
	3	2	1	0
Channel sinuosity	**High** > 2 well-defined bends with deep outside areas (cut banks) and shallow inside areas (point bars) present	**Moderate** 1 well-defined bend or > 3 moderately-defined bends present	**Low** <3 moderately-defined bends or only poorly-defined bends present	**None** Straight channel; may be channelized
	3	2	1	0
Riparian buffer vegetation	**Extensive** Width of natural buffer is >20 meters	**Wide** Width of natural buffer is 10.1-20 meters	**Moderate** Width of natural buffer is 5-10 meters	**Narrow** Width of natural buffer is <5 meters
	3	2	1	0
Aesthetics of reach	**Wilderness** Outstanding natural beauty; usually wooded or unpastured area; water clarity is usually exceptional	**Natural Area** Trees and/or native vegetation are common; some development evident (from fields, pastures, dwellings); water clarity may be slightly turbid	**Common Setting** Not offensive; area is developed, but uncluttered such as in an urban park; water clarity may be turbid or discolored	**Offensive** Stream does not enhance the aesthetics of the area; cluttered; highly developed; may be a dumping area; water clarity is usually turbid or discolored
	3	2	1	0

HQI Total Score: ≥ 26 Exceptional; 20-25 High; 14-19 Intermediate; ≤13 Limited

TABLE 5. —Rapid habitat assessment parameters and rating criteria (Twidwell and Davis 1989).

Rating Parameter	Scoring Criteria			
Instream cover	Abundant (≥50%) (4)	Common (30<50%) (3)	Rare (10<29.9%) (2)	Absent (<10%) (0)
Riffle/runs	Abundant (≥5) (4)	Common (2-4) (3)	Rare (1) (2)	Absent (0)
Pool depth	Large and deep Max depth > 4 feet (4)	Moderate Max depth 2-4 feet (3)	Small Max depth < 2 feet (2)	No pools Shallow pockets (1)
Bank stability (rate each attribute separately and average)	Stable Little evidence (<10%) of erosion Side slopes < 30° (3)	Moderately stable Some evidence (10<30%) of erosion Side slopes 30-40° (2)	Moderately unstable Moderate frequency (30<50%)of erosion. Side slopes 40-60° (1)	Unstable Frequent (≥50%) eroded areas. Side slopes > 60° (0)
Riparian cover	Extensive Width of natural cover > 350 feet (3)	Wide Width of natural cover 150-350 feet (2)	Moderate Width of natural cover 15-150 feet (1)	Narrow Width of natural cover >15 feet (0)
Flow fluctuation	Minor Little or none from base flow (3)	Moderate Debris along middle portion of banks (2)	Severe Evidence of debris high on banks (1)	Severe Intermittent stream (0)
Channel sinuosity	High >2 well defined outside bends (3)	Moderate 1 well developed or ≥3 moderately defined bends (2)	Low <3 moderately defined or only poorly defined bends (1)	None Straight channel; possibly altered (0)
Bottom substrate	Stable ≥50% cobbles, rubble, or gravel (3)	Moderately stable 30<50% gravel or larger substrate (2)	Moderately unstable 10<30% gravel or larger substrate (1)	Unstable <10% gravel or larger substrate; bottom uniform sand, clay, silt, or bedrock (0)
Aesthetics	Wilderness Outstanding natural beauty; usually wooded or unpastured; water clarity exceptional (3)	Natural area Trees or native vegetation common; some development evident; water clarity discolored (2)	Common setting Not offensive, developed but uncluttered; water may be colored or turbid (1)	Offensive Stream does not enhance area; cluttered, highly developed, dumping area; water discolored (0)

Rapid Habitat Total Score: ≥26 Exceptional; 25-21 High; 20-15 Intermediate; ≤14 Limited

TABLE 6. —Aquatic life use subcategories as defined in the Texas Surface Water Quality Standards (TNRCC 1995).

Aquatic Life Use Subcategory	Dissolved Oxygen Criteria, mg/L			Aquatic Life Attributes					
	Freshwater mean/ minimum	Freshwater in Spring mean/ minimum	Saltwater mean/ minimum	Habitat Character- istics	Species Assemblage	Sensitive Species	Diversity	Species Richness	Trophic Structure
Exceptional	6.0/4.0	6.0/5.0	5.0/4.0	Outstanding natural variability	Exceptional or unusual	Abundant	Exceptionally high	Exceptionally high	Balanced
High	5.0/3.0	5.5/4.5	4.0/3.0	Highly diverse	Usual asso- ciation of regionally expected species	Present	High	High	Balanced to slightly imbalanced
Intermediate	4.0/3.0	5.0/4.0	3.0/2.0	Moderately diverse	Some expected species	Very low in abundance	Moderate	Moderate	Moderately imbalanced
Limited	3.0/2.0	4.0/3.0		Uniform	Most regionally expected species absent	Absent	Low	Low	Severely imbalanced

- Dissolved oxygen means are applied as a minimum average over a 24-hour period.
- Daily minima are not to extend beyond 8 hours per 24-hour day. Lower dissolved oxygen minima may apply on a site-specific basis, when natural daily fluctuations below the mean are greater than the difference between the mean and minima of the appropriate criteria.
- Spring criteria to protect fish spawning periods are applied during that portion of the first half of the year when water temperatures are 63°F to 73°F.
- Quantitative criteria to support aquatic life attributes are described in the standards implementation procedures.
- Dissolved oxygen analyses and computer models to establish effluent limits for permitted discharges will normally be applied to mean criteria at steady-state, critical conditions.
- Determination of standards attainment for dissolved oxygen criteria is specified in §307.9(d)(6) (relating to Determination of Standards Attainment).

TABLE 7. —Stream distance (km) estimates of aquatic life uses for all sites based on regional index of biotic integrity scores (LCL=lower 95% confidence limit; UCL=upper 95% confidence limit).

	LCL	% km	UCL	LCL	Stream km	UCL
All sites						
Exceptional	9.0	20.7	32.4	698	1607	2517
High	49.5	62.9	76.4	3840	4885	5930
Intermediate	6.9	16.2	25.4	534	1255	1976
Limited	0.0	0.2	0.5	0	17	42
Nonurban sites						
Exceptional	9.2	21.3	33.5	686	1598	2510
High	49.1	63.1	77.1	3682	4728	5774
Intermediate	6.0	15.6	25.2	448	1167	1886
Limited	0.0	0.0	0.0	0	0	0
Urban sites						
Exceptional	0.0	3.4	8.1	0	9	22
High	39.3	57.8	76.4	107	157	207
Intermediate	14.5	32.4	50.2	39	88	136
Limited	0.0	6.4	15.1	0	17	41
Texas Blackland Prairies						
Exceptional	0.0	18.3	42.4	0	258	599
High	33.6	62.1	90.6	474	877	1281
Intermediate	0.0	18.4	40.8	0	260	576
Limited	0.0	1.2	3.1	0	17	44
East Central Texas Plains						
Exceptional	0.4	12.2	24.1	8	237	466
High	39.1	59.2	79.3	758	1148	1537
Intermediate	10.4	28.6	46.7	202	554	906
Limited	0.0	0.0	0.0	0	0	0
South Central Plains						
Exceptional	6.8	25.2	43.7	298	1112	1926
High	44.6	64.8	85.0	1970	2860	3751
Intermediate	0.0	10.0	21.9	0	441	965
Limited	0.0	0.0	0.0	0	0	0
Second-order streams						
Exceptional	0.0	15.8	35.7	0	510	1153
High	52.1	74.9	97.8	1681	2418	3154
Intermediate	0.0	9.3	23.6	0	299	762
Limited	0.0	0.0	0.0	0	0	0
Third-order streams						
Exceptional	4.5	26.5	48.6	114	678	1241
High	34.0	57.6	81.2	869	1471	2073
Intermediate	0.0	15.2	31.4	0	387	801
Limited	0.0	0.7	1.7	0	17	43
Fourth-order streams						
Exceptional	5.4	21.1	36.8	108	420	732
High	30.4	50.2	70.0	605	997	1389
Intermediate	11.3	28.7	46.0	224	569	914
Limited	0.0	0.0	0.0	0	0	0

TABLE 8. —Stream distance (km) estimates of aquatic life uses for all sites based on benthic collections (LCL=lower 95% confidence limit; UCL=upper 95% confidence limit).

	LCL	% km	UCL	LCL	Stream km	UCL
All sites						
Exceptional	0.00	1.34	3.96	0	104	308
High	58.96	71.76	84.57	4578	5572	6567
Intermediate	11.77	23.60	35.44	914	1833	2752
Limited	0.00	3.29	9.62	0	256	747
Nonurban sites						
Exceptional	0.00	1.39	4.11	0	104	308
High	59.32	72.61	85.89	4445	5441	6436
Intermediate	10.32	22.59	34.86	773	1693	2612
Limited	0.00	3.41	9.97	0	256	748
Urban sites						
Exceptional	0.00	0.00	0.00	0	0	0
High	29.61	48.41	67.20	80	131	183
Intermediate	32.80	51.59	70.39	89	140	191
Limited	0.00	0.00	0.00	0	0	0
Texas Blackland Prairies						
Exceptional	0.00	7.36	21.51	0	104	304
High	47.55	72.63	97.71	672	1026	1381
Intermediate	0.00	20.01	42.55	0	283	601
Limited	0.00	0.00	0.00	0	0	0
East Central Texas Plains						
Exceptional	0.00	0.00	0.00	0	0	0
High	56.48	75.15	93.82	1095	1457	1819
Intermediate	6.18	24.85	43.52	120	482	844
Limited	0.00	0.00	0.00	0	0	0
South Central Plains						
Exceptional	0.00	0.00	0.00	0	0	0
High	50.44	70.00	89.56	2226	3089	3952
Intermediate	6.26	24.21	42.15	276	1068	1860
Limited	0.00	5.79	16.84	0	256	743
Second-order streams						
Exceptional	0.00	0.00	0.00	0	0	0
High	44.88	68.89	92.89	1448	2222	2997
Intermediate	1.87	23.19	44.51	60	748	1436
Limited	0.00	7.93	22.93	0	256	740
Third-order streams						
Exceptional	0.00	0.00	0.00	0	0	0
High	54.26	75.08	95.90	1385	1917	2448
Intermediate	4.10	24.92	45.74	105	636	1168
Limited	0.00	0.00	0.00	0	0	0
Fourth-order streams						
Exceptional	0.00	5.24	15.24	0	104	303
High	54.46	72.17	89.89	1081	1433	1785
Intermediate	6.38	22.59	38.80	127	448	770
Limited	0.00	0.00	0.00	0	0	0

TABLE 9. —Stream distance (km) estimates of aquatic life uses for all sites based on Habitat Quality Index (LCL=lower 95% confidence limit; UCL=upper 95% confidence limit).

	LCL	% km	UCL	LCL	Stream km	UCL
All sites						
Exceptional	0.00	0.00	0.00	0	0	0
High	16.42	28.91	41.40	1275	2245	3215
Intermediate	51.17	64.39	77.62	3973	5000	6027
Limited	0.29	6.70	13.11	23	520	1018
Nonurban sites						
Exceptional	0.00	0.00	0.00	0	0	0
High	16.60	29.58	42.55	1244	2216	3189
Intermediate	50.50	64.21	77.91	3784	4811	5838
Limited	0.00	6.22	12.84	0	466	962
Urban sites						
Exceptional	0.00	0.00	0.00	0	0	0
High	0.00	10.50	21.75	0	29	59
Intermediate	52.04	69.53	87.03	141	189	236
Limited	4.41	19.97	35.53	12	54	97
Texas Blackland Prairies						
Exceptional	0.00	0.00	0.00	0	0	0
High	0.57	22.73	44.89	8	321	634
Intermediate	33.88	61.71	89.54	479	872	1265
Limited	0.00	15.56	37.91	0	220	536
East Central Texas Plains						
Exceptional	0.00	0.00	0.00	0	0	0
High	12.00	32.33	52.66	233	627	1021
Intermediate	41.21	61.37	81.53	799	1190	1580
Limited	0.00	6.30	14.66	0	122	284
South Central Plains						
Exceptional	0.00	0.00	0.00	0	0	0
High	10.49	29.38	48.27	463	1297	2131
Intermediate	46.91	66.58	86.25	2070	2938	3807
Limited	0.00	4.04	11.56	0	178	510
Second-order streams						
Exceptional	0.00	0.00	0.00	0	0	0
High	0.00	20.16	40.72	0	650	1314
Intermediate	57.88	78.51	99.14	1867	2533	3198
Limited	0.00	1.33	2.88	0	43	93
Third-order streams						
Exceptional	0.00	0.00	0.00	0	0	0
High	9.59	31.56	53.53	245	806	1367
Intermediate	30.88	54.53	78.19	788	1392	1996
Limited	0.00	13.91	31.37	0	355	801
Fourth-order streams						
Exceptional	0.00	0.00	0.00	0	0	0
High	19.96	39.71	59.47	396	788	1181
Intermediate	34.31	54.13	73.96	681	1075	1468
Limited	0.00	6.15	14.50	0	122	288

TABLE 10. —Stream distance (km) estimates of aquatic life uses for all sites based on 24-mean and minimum dissolved oxygen concentrations (LCL=lower 95% confidence limit; UCL=upper 95% confidence limit).

	LCL	% km	UCL	LCL	Stream km	UCL
All sites						
Exceptional	47.7	61.4	75.1	3614	4655	5696
High	3.1	10.0	16.8	239	756	1272
Intermediate	1.3	9.8	18.3	101	743	1384
Limited	0.0	1.5	4.1	0	116	309
Nonattaining	5.9	17.3	28.7	450	1311	2172
Nonurban sites						
Exceptional	47.9	62.1	76.3	3510	4550	5589
High	2.2	9.2	16.2	162	675	1188
Intermediate	1.1	9.9	18.7	83	725	1368
Limited	0.0	1.3	4.0	0	98	290
Nonattaining	5.6	17.4	29.2	413	1275	2137
Urban sites						
Exceptional	22.5	41.0	59.5	58	106	154
High	13.5	31.2	48.9	35	81	126
Intermediate	0.0	6.7	16.4	0	17	42
Limited	0.4	7.1	13.8	1	18	36
Nonattaining	0.8	14.0	27.2	2	36	70
Texas Blackland Prairies						
Exceptional	20.6	50.1	79.5	289	700	1112
High	0.0	12.1	27.2	0	170	380
Intermediate	0.0	11.3	29.9	0	158	418
Limited	0.0	0.4	1.1	0	5	15
Nonattaining	0.0	26.2	54.5	0	366	762
East Central Texas Plains						
Exceptional	43.3	63.5	83.8	839	1232	1625
High	1.5	19.1	36.8	29	371	712
Intermediate	0.0	9.1	19.2	0	176	373
Limited	0.0	5.1	14.9	0	99	288
Nonattaining	0.0	3.2	9.3	0	61	179
South Central Plains						
Exceptional	43.7	64.2	84.7	1853	2723	3594
High	0.0	5.1	11.8	0	215	500
Intermediate	0.0	9.6	22.5	0	408	956
Limited	0.0	0.3	0.6	0	12	27
Nonattaining	3.3	20.8	38.4	139	884	1629
Second-order streams						
Exceptional	45.4	69.2	93.0	1465	2232	3000
High	0.0	6.3	15.5	0	203	500
Intermediate	0.0	8.0	22.3	0	257	720
Limited	0.0	0.3	0.7	0	9	23
Nonattaining	0.0	16.3	36.2	0	525	1168
Third-order streams						
Exceptional	31.8	56.1	80.5	756	1333	1911
High	0.0	4.7	13.1	0	111	311
Intermediate	0.0	12.8	29.7	0	304	705
Limited	0.0	4.3	12.4	0	102	293
Nonattaining	0.5	22.1	43.7	11	524	1037
Fourth-order streams						
Exceptional	35.5	55.0	74.5	704	1090	1476
High	5.4	22.3	39.3	106	442	777
Intermediate	0.0	9.1	18.8	0	181	372
Limited	0.0	0.2	0.7	0	5	15
Nonattaining	0.0	13.3	27.0	0	263	534

TABLE 11.—Spearman rank correlation coefficients between 24-hour minimum and mean dissolved oxygen concentrations and stream habitat variables. Only significant correlations are presented (*P*<0.05). Dissolved oxygen concentrations did not demonstrate significant correlations with measures of riparian disturbance.

	Correlation Coefficient (r_s)	
Habitat Parameter	Mean Dissolved Oxygen	Minimum Dissolved Oxygen
Discharge	0.30	0.45
Reach slope	0.29	0.23
Mean width/depth ratio	0.34	
Mean bankfull width	0.25	
Mean bankfull height	0.21	
Entrenchment ratio	-0.22	
Mean residual depth	-0.23	
Percent riffle	0.30	0.26
Percent pool	-0.52	-0.55
Percent slow water (glide + pool)	-0.31	
Substrate fines	-0.36	-0.31
Mean $\log_{10}$ substrate diameter	0.22	
Erodible $\log_{10}$ substrate diameter	0.25	
Mean embeddedness	-0.28	
Large woody debris (#/100 m)	-0.37	
Large woody debris (volume)	-0.39	
Mean midstream canopy	-0.29	
Mean bank canopy	-0.27	
Riparian canopy > 0.3 m DBH	-0.29	
All types cover	-0.22	
Brush and small debris cover	-0.32	
Large woody debris cover	-0.23	
Mean undercut distance cover		0.22
Overhanging vegetation cover	0.22	
HQI		0.26

TABLE 12. —Stream distance (km) estimates for selected water quality parameters (LCL=lower 95% confidence limit; UCL= upper 95% confidence limit) at state 85th percentile screening levels (TCEQ 2002).

Screening Level	LCL	% km	UCL	LCL	Stream km	UCL
			Total phosphorus			
Texas Blackland Prairies						
≤0.8 mg/L	65.3	87.7	100	919	1235	1550
>0.8 mg/L	0	12.3	34.7	0	173	489
East Central Texas Plains						
≤0.8 mg/L	85.3	93.8	100	1653	1819	1984
>0.8 mg/L	0	6.2	14.7	0	120	285
South Central Plains						
≤0.8 mg/L	93.0	97.6	100	4104	4306	4509
>0.8 mg/L	0	2.4	7.0	0	107	309
			Orthophosphate			
Texas Blackland Prairies						
≤0.5 mg/L	47.2	75.5	100.0	668	1066	1465
>0.5 mg/L	0	24.5	52.8	0	347	745
East Central Texas Plains						
≤0.5 mg/L	71.8	85.6	99.4	1392	1660	1927
>0.5 mg/L	0.6	14.4	28.2	11	279	546
South Central Plains						
≤0.5 mg/L	93.0	97.6	100.0	4104	4306	4509
>0.5 mg/L	0	2.4	7.0	0	107	309
			Chlorophyll *a*			
Texas Blackland Prairies						
≤11.6 μg/L	37.5	66.2	94.9	530	935	1341
>11.6 μg/L	5.1	33.8	62.5	72	478	883
East Central Texas Plains						
≤11.6 μg/L	74.3	89.4	100	1441	1733	2025
>11.6 μg/L	0.0	10.6	25.7	0	205	497
South Central Plains						
≤11.6 μg/L	69.3	84.2	99.2	3057	3719	4380
>11.6 μg/L	0.7	15.7	30.7	33	695	1356
			Nitrate plus nitrite			
Texas Blackland Prairies						
≤2.76 mg/L	48.2	74.4	100	680	1051	1421
>2.76 mg/L	0.0	25.6	51.8	0	362	732
East Central Texas Plains						
≤2.76 mg/L	63.4	81.2	99.0	1229	1574	1919
>2.76 mg/L	1.0	18.8	36.6	20	364	709
South Central Plains						
≤2.76 mg/L	92.9	97.5	100	4101	4303	4506
>2.76 mg/L	0.0	2.5	7.1	0	110	313
			Ammonia nitrogen			
Texas Blackland Prairies						
≤0.17	95.9	98.4	100	1355	1390	1425
>0.17	0	1.6	4.1	0	23	58
East Central Texas Plains						
≤0.17	76.5	90.2	100	1217	1436	1654
>0.17	0	9.8	23.5	0	156	375
South Central Plains						
≤0.17	84.8	94.8	100	2795	3127	3459
>0.17	0	5.2	15.2	0	170	502

TABLE 13. —Comparison of water quality parameters and mercury in fish tissue for significance among land use types, ecoregions, and stream orders. Differing letters (e.g., a, b) indicate significantly different means (P<0.05). Numbers are weighted means (standard error).

	Land Use		**Ecoregion**			**Stream order**		
Parameter	Urban	Nonurban	TBP	ECTP	SCP	2	3	4
24-hour mean DO (mg/L)	6.08 (0.46) a	5.97 (0.40) a	5.39 (0.94) a	6.23 (0.30) a	6.06 (0.59) a	6.48 (0.70) a	5.34 (0.65) a	5.91 (0.41) a
24-hour minimum DO (mg/L)	4.67 (0.43) a	5.21 (0.39) a	4.37 (0.87) a	5.53 (0.30) a	5.31 (0.60) a	5.84 (0.70) a	4.56 (0.63) a	4.89 (0.39) a
Mean temperature (Celsius)	27.24 (0.42) a	25.49 (0.57) b	26.75 (0.62) a	27.47 (0.44) a	24.27 (0.86) b	24.07 (0.92) a	25.66 (0.96) ab	27.82 (0.51) b
Max temperature (Celsius)	28.99 (0.55) a	26.88 (0.65) b	27.91 (0.84) ab	28.84 (0.47) a	25.78 (1.00) b	25.59 (1.08) a	26.93 (1.07) ab	29.20 (0.58) b
pH	7.52 (0.10) a	7.10 (0.10) b	7.71 (0.11) a	7.34 (0.13) b	6.82 (0.12) c	6.81 (0.15) a	7.19 (0.12) ab	7.52 (0.12) b
Turbidity (NTU)	15.61 (4.52) a	17.56 (1.63) a	14.28 (2.76) a	21.68 (3.50) a	16.69 (2.12) a	16.37 (2.77) a	17.23 (2.47) a	19.65 (2.76) a
TSS (mg/L)	14.61 (3.13) a	15.96 (1.75) a	14.56 (2.84) a	19.76 (3.07) a	14.66 (2.50) a	15.50 (3.16) a	15.31 (2.78) a	17.38 (2.24) a
TDS (mg/L)	302.48 (36.02) a	226.69 (26.20) a	416.01 (51.11) a	309.02 (42.75) a	134.58 (20.50) b	138.32 (25.31) a	249.01 (43.87) b	351.95 (47.61) b
VSS (mg/L)	7.08 (0.62) a	7.86 (0.59) a	8.09 (1.45) a	8.50 (0.76) a	7.46 (0.83) a	8.04 (1.06) a	6.99 (0.89) a	8.57 (0.84) a
TOC (mg/L)	4.75 (0.31) a	5.65 (0.71) a	4.59 (0.77) a	4.83 (0.44) a	6.30 (1.15) a	6.18 (1.56) a	5.62 (0.50) a	4.71 (0.54) a
Total alkalinity (mg/L)	131.91 (10.50) a	83.88 (10.37) b	176.26 (10.90) a	120.42 (19.70) b	41.22 (5.64) c	51.39 (11.59) a	97.91 (18.59) b	125.23 (15.41) b
Total hardness (mg/L)	176.47 (21.51) a	101.20 (12.86) b	221.87 (15.05) a	144.42 (21.54) b	48.21 (7.97) c	59.22 (14.00) a	116.47 (21.49) b	160.07 (20.60) b
Chloride (mg/L)	36.40 (7.12) a	41.09 (6.21) a	60.64 (17.49) ab	63.70 (11.91) a	24.62 (5.25) b	21.01 (5.82) a	44.56 (11.39) ab	68.62 (11.87) b
Sulfate (mg/L)	72.22 (16.76) a	29.97 (6.60) b	88.37 (21.11) a	38.87 (5.81) b	9.97 (4.74) c	15.14 (5.56) a	36.06 (11.59) ab	52.02 (14.98) b
Ammonia N (mg/L)	0.06 (0.02) a	0.03 (0.01) a	0.02 (0.01) a	0.04 (0.02) a	0.03 (0.02) a	0.02 (0.01) a	0.05 (0.02) a	0.02 (0.01) a
TKN (mg/L)	0.61 (0.05) a	0.56 (0.06) a	0.72 (0.12) a	0.60 (0.04) a	0.49 (0.09) a	0.53 (0.12) a	0.59 (0.10) a	0.56 (0.05) a
Nitrate (mg/L)	0.49 (0.12) a	1.13 (0.51) a	1.64 (0.72) a	1.18 (0.51) a	0.90 (0.80) a	0.42 (0.31) a	0.83 (0.43) a	2.57 (1.71) a
Total N (mg/L)	1.11 (0.14) a	1.69 (0.52) a	2.37 (0.70) a	1.78 (0.51) a	1.39 (0.81) a	0.95 (0.32) a	1.43 (0.45) a	3.13 (1.71) a
Ortho P (mg/L)	0.05 (0.02) a	0.13 (0.05) a	0.21 (0.10) a	0.15 (0.06) a	0.09 (0.08) a	0.01 (0) a	0.15 (0.07) b	0.28 (0.17) ab
Total P (mg/L)	0.08 (0.03) a	0.18 (0.06) a	0.28 (0.11) a	0.19 (0.06) a	0.14 (0.08) a	0.05 (0.01) a	0.22 (0.07) b	0.34 (0.18) ab
Chlorophyll *a* (mg/L)	0.01 (0) a	0.01 (0) a	0.02 (0.01) a	0.01 (0.01) a	0.01 (0) a	0.01 (0) a	0.01 (0.01) a	0.01 (0) a
Mercury in fish tissue (mg/kg)								
Muscle, all species	0.21 (0.06) a	0.41 (0.07) b	0.10 (0.02) a	0.28 (0.05) b	0.55 (0.09) c	0.44 (0.12) a	0.37 (0.09) a	0.34 (0.08) a
Whole fish, all species	0.06 (0) a	0.17 (0.03) b	0.06 (0.01) a	0.10 (0.02) b	0.23 (0.05) c	0.24 (0.07) a	0.13 (0.02) ab	0.09 (0.01) b
Whole longear sunfish	0.07 (0) a	0.14 (0.02) b	0.06 (0.01) a	0.09 (0.02) a	0.20 (0.03) b	0.20 (0.04) a	0.13 (0.02) a	0.08 (0.01) b

TABLE 14. —Comparison of habitat measures for significance among land use types, ecoregions, and stream order. Differing letters (e.g., a, b) indicate significantly different means (P<0.05). Riparian disturbance, riparian characterization, residual pool metrics, and bed stability are derived after Kaufman et al. (1999). Numbers are weighted means (standard error).

	Land Use		Ecoregion			Stream order		
Parameter	Urban	Nonurban	TBP	ECTP	SCP	2	3	4
Morphology								
Length of sample reach (m)	283.90 (18.74) a	258.68 (16.60) a	302.56 (43.08) a	280.31 (21.86) a	236.69 (21.26) a	208.28 (21.67) a	261.59 (22.96) a	340.29 (28.82) b
Drainage area (km^2)	179.47 (29.76) a	488.23 (101.08) b	882.37 (264.03) a	919.29 (254.57) a	153.71 (40.28) b	42.43 (8.67) a	408.58 (129.83) b	1272.83 (230.26) c
Discharge (m^3/sec)	0.15 (0.05) a	0.25 (0.06) a	0.55 (0.22) a	0.24 (0.06) a	0.14 (0.04) a	0.13 (0.04) a	0.16 (0.06) a	0.52 (0.15) b
Sinuosity	1.14 (0.04) a	1.46 (0.13) b	1.18 (0.05) a	1.36 (0.11) a	1.58 (0.21) a	1.61 (0.27) a	1.27 (0.05) a	1.42 (0.15) a
Reach slope (%)	0.17 (0.03) a	0.12 (0.02) a	0.13 (0.03) a	0.09 (0.02) a	0.12 (0.03) a	0.17 (0.04) a	0.08 (0.02) a	0.09 (0.02) a
Mean wetted width (m)	8.35 (0.51) a	7.05 (0.51) a	9.21 (1.51) a	8.50 (0.87) a	5.81 (0.46) b	5.30 (0.50) a	7.19 (0.89) a	9.91 (0.79) b
Mean thalweg depth (cm)	58.88 (5.76) a	58.79 (3.40) a	67.23 (7.61) ab	70.32 (7.65) a	51.02 (3.08) b	49.82 (4.98) a	61.36 (5.48) ab	70.06 (4.40) b
Mean bank angle (degrees)	43.68 (3.09) a	45.10 (2.50) a	47.38 (6.06) a	51.37 (4.83) a	41.52 (3.00) a	48.68 (4.14) a	43.33 (4.64) a	41.35 (2.75) a
Mean width / depth ratio (m/m)	25.50 (2.29) a	16.55 (1.98) b	24.18 (9.21) a	16.95 (1.94) a	14.48 (1.09) a	14.89 (1.39) a	18.88 (5.39) a	17.48 (1.41) a
Mean width * depth product (m^2)	5.73 (0.77) a	4.87 (0.56) a	6.63 (1.14) a	7.03 (1.33) a	3.42 (0.45) b	3.20 (0.81) a	4.68 (0.61) a	7.96 (0.89) b
Mean bankfull width (m)	21.17 (2.99)a	14.20 (1.32) b	22.64 (4.91) a	16.37 (1.78) a	10.98 (0.90) b	9.86 (0.91) a	15.23 (2.73) ab	20.89 (1.96) b
Mean bankfull height (m)	2.48 (0.19) a	2.02 (0.10) b	2.67 (0.18) a	2.32 (0.14) a	1.70 (0.11) b	1.65 (0.15) a	2.14 (0.15) b	2.51 (0.10) b
Mean flood prone area width (m)	176.38 (46.79) a	519.64 (115.10) b	372.14 (162.84) a	573.78 (196.78) a	521.96 (167.13) a	233.79 (111.92) a	623.76 (202.97) ab	803.29 (258.50) b
Mean flood prone area height (m)	4.92 (0.39) a	4.01 (0.21) b	5.33 (0.37) a	4.55 (0.30) a	3.41 (0.23) b	3.30 (0.30) a	4.29 (0.31) b	4.93 (0.24) b
Entrenchment ratio	9.24 (2.22) a	42.10 10.38) b	24.34 (12.37) a	34.28 (9.07) a	49.20 (16.67) a	30.91 (17.25) a	50.83 (17.58) a	44.56 (14.15) a
Percent riffle	10.47 (2.06) a	4.63 (0.90) b	8.46 (2.27) a	2.78 (0.83) b	4.58 (1.28) ab	6.27 (1.55) a	3.55 (1.33) a	4.18 (1.39) a
Percent glide	31.88 (5.00) a	43.15 (4.96) a	37.78 (6.97) a	47.06 (7.06) a	42.45 (7.46) a	47.23 (8.69) a	35.72 (8.01) a	44.51 (5.41) a
Percent pool	57.19 (5.45) a	51.63 (5.35) a	53.22 (8.32) a	50.00 (6.91) a	52.19 (8.11) a	46.40 (9.48) a	60.20 (8.50) a	49.88 (5.64) a
Mean residual depth (cm)	38.38 (4.72) a	31.70 (2.12) a	37.61 (4.11) a	37.87 (5.37) ab	27.50 (1.91) b	29.34 (3.80) a	31.15 (3.20) a	37.15 (2.29) a
Residual pools >100 cm deep (n)	1.01 (0.21) a	0.80 (0.12) a	0.90 (0.22) ab	1.17 (0.22) a	0.62 (0.16) b	0.58 (0.19) a	0.69 (0.17) a	1.34 (0.21) b
Residual pools >75 cm deep (n)	1.53 (0.24) a	1.53 (0.18) a	1.44 (0.29) a	1.89 (0.27) a	1.40 (0.26) a	1.32 (0.29) a	1.53 (0.35) a	1.86 (0.18) a
Maximum residual depth (cm)	130.55 (13.72) a	113.28 (6.87) a	129.64 (15.53) a	127.08 (15.11) a	103.04 (7.47) a	106.21 (11.41) ab	105.91 (10.42) a	136.61 (10.27) b
Mean residual pool area (m^2)	17.94 (3.04) a	14.99 (1.68) a	18.53 (3.07) a	22.75 (4.30) a	10.64 (1.22) b	11.18 (2.57) a	14.21 (2.10) a	22.60 (2.82) b
Substrate								
Substrate fines (%)	21.52 (4.01) a	33.40 (4.69) a	30.18 (9.00) a	43.80 (6.85) a	29.14 (6.68) a	19.47 (6.56) a	45.29 (8.13) b	39.14 (5.75) b
Substrate sand (%)	18.34 (3.59) a	43.95 (4.84) b	17.35 (5.15) a	38.19 (6.82) b	53.42 (6.77) b	53.45 (8.50) a	35.33 (7.02) a	36.10 (5.42) a
Substrate sand and fines (%)	39.86 (5.32) a	77.36 (4.19) b	47.53 (9.22) a	81.98 (6.33) b	82.56 (5.43) b	72.91 (7.57) a	80.62 (6.09) a	75.25 (6.08) a
Substrate ≤ fine gravel (%)	67.38 (4.77) a	87.45 (3.43) b	66.70 (7.37) a	88.67 (5.69) b	92.32 (4.33) b	85.62 (6.51) a	86.83 (5.38) a	88.46 (3.12) a
Substrate ≥ coarse gravel (%)	32.62 (4.77) a	12.55 (3.43) b	33.30 (7.37) a	11.33 (5.69) b	7.68 (4.33) b	14.38 (6.51) a	13.17 (5.38) a	11.54 (3.12) a
Substrate bedrock (%)	8.07 (3.40) a	2.22 (1.57) a	12.51 (7.33) a	0.00 (0.00) a	0.25 (0.13) a	0.77 (0.38) a	6.31 (4.40) a	0.09 (0.06) a
Mean embeddedness (%)	70.52 (5.09) a	89.17 (3.40) b	71.54 (7.24) a	94.38 (2.84) b	91.38 (4.95) b	86.79 (6.76) a	90.24 (4.57) a	89.11 (3.24) a
Substrate mean log diameter (mm)	0.51 (0.22) a	-0.47 (0.18) b	0.32 (0.41) a	-0.73 (0.26) b	-0.55 (0.24) ab	-0.13 (0.30) a	-0.64 (0.30) a	-0.68 (0.17) a
Relative bed stability	-0.59 (0.23) a	-1.33 (0.15) b	-0.84 (0.39) a	-1.57 (0.22) a	-1.34 (0.20) a	-1.13 (0.23) a	-1.40 (0.32) a	-1.48 (0.14) a
Log erodible substrate diameter (mm)	1.15 (0.10) a	0.89 (0.08) a	1.16 (0.09) a	0.83 (0.11) b	0.84 (0.13) b	1.00 (0.13) a	0.83 (0.16) a	0.80 (0.11) a

TABLE 14. —Continued.

	Land Use		Ecoregion			Stream order		
Parameter	Urban	Nonurban	TBP	ECTP	SCP	2	3	4
Canopy and woody debris								
Mean bank canopy cover (%)	73.43 (4.91) a	91.87 (1.51) b	84.94 (5.79) a	89.86 (2.07) a	93.83 (1.47) a	93.50 (1.63) a	91.38 (3.43) ab	87.31 (2.25) b
Mean midstream canopy cover (%)	55.58 (6.14) a	83.79 (2.58) b	73.01 (8.19) ab	77.35 (4.46) a	88.34 (2.66) b	88.64 (2.75) a	84.88 (5.10) a	70.67 (4.29) b
Riparian canopy cover > 5m	0.41 (0.05) a	0.56 (0.03) b	0.56 (0.07) a	0.52 (0.03) a	0.57 (0.03) a	0.59 (0.04) a	0.54 (0.05) a	0.53 (0.03) a
Riparian woody cover	0.64 (0.09) a	0.77 (0.04) a	0.78 (0.09) a	0.72 (0.04) a	0.78 (0.06) a	0.86 (0.07) a	0.70 (0.06) ab	0.70 (0.04) b
Riparian mid-layer woody cover	0.59 (0.08) a	0.73 (0.04) a	0.72 (0.09) a	0.67 (0.04) a	0.74 (0.05) a	0.80 (0.06) a	0.65 (0.06) a	0.67 (0.04) a
Riparian ground layer	0.27 (0.03) a	0.28 (0.02) a	0.31 (0.05) a	0.30 (0.02) a	0.26 (0.03) a	0.30 (0.03) a	0.27 (0.03) a	0.25 (0.03) a
Riparian three-layers (reach fraction)	0.73 (0.06) a	0.92 (0.02) b	0.89 (0.04) a	0.93 (0.02) a	0.91 (0.02) a	0.93 (0.02) a	0.90 (0.03) a	0.89 (0.03) a
Canopy, mid-layer (reach fraction)	0.74 (0.07) a	0.93 (0.02) b	0.90 (0.04) a	0.94 (0.02) a	0.93 (0.02) a	0.95 (0.02) a	0.91 (0.03) a	0.91 (0.03) a
Riparian Canopy > 0.3 m DBH	0.20 (0.04) a	0.27 (0.02) a	0.27 (0.05) a	0.27 (0.02) a	0.27 (0.02) a	0.28 (0.02) a	0.27 (0.03) a	0.24 (0.02) a
Bare ground	0.37 (0.03) a	0.50 (0.03) b	0.37 (0.04) a	0.48 (0.03) b	0.54 (0.04) b	0.53 (0.05) a	0.50 (0.03) a	0.44 (0.03) a
Large woody debris (#/100 m)	2.97 (0.70) a	5.92 (0.86) b	7.40 (2.77) a	5.34 (0.92) a	5.52 (1.04) a	4.35 (1.14) a	6.90 (1.69) a	6.81 (1.29) a
Cover (%)								
All types cover	13.47 (1.39) a	25.55 (1.86) b	16.06 (2.54) a	28.48 (3.05) b	26.55 (2.61) b	22.23(2.46) a	26.56(3.65) a	27.98(3.00) a
Natural cover	12.09 (1.30) a	24.99 (1.84) b	15.39 (2.44) a	28.35 (3.06) b	25.79 (2.59) b	21.80 (2.45) a	25.43 (3.63) a	27.84 (3.01) a
Filamentous algae	1.89 (0.70) a	1.79 (1.20) a	6.92 (5.91) a	0.82 (0.43) a	0.58 (0.26) a	0.48 (0.30) a	4.12 (3.40) a	0.92 (0.40) a
Macrophytes	2.09 (0.90) a	1.95 (1.23) a	0.44 (0.21) a	1.72 (1.03) a	2.55 (2.02) a	2.90 (2.68) a	0.16 (0.09) a	2.73 (1.37) a
Brush and small debris	4.11 (0.94) a	15.35 (1.66) b	7.01 (1.95) a	18.71(2.94) b	15.85(2.33) b	10.72(1.53) a	18.00(3.49) ab	17.93(2.81) b
Large woody debris	1.10 (0.35) a	3.26 (0.38) b	3.06 (1.02) a	3.58 (0.80) a	3.04 (0.45) a	2.43 (0.49) a	3.39 (0.68) a	4.13 (0.77) a
Leaf litter	0.28 (0.15) a	4.56 (1.40) b	0.30 (0.19) a	1.33 (0.31) a	7.08 (2.26) b	4.74 (1.66) a	6.08 (3.41) a	1.73 (0.61) a
Boulders	1.16 (0.40) a	0.03 (0.02) a	0.29 (0.12) a	0.06 (0.04) a	0.00 (0.00) a	0.08 (0.04) a	0.09 (0.05) a	0.04 (0.04) a
Overhanging vegetation	5.02 (0.70) a	5.15 (1.01) a	3.80 (0.85) a	4.81 (1.39) a	5.72 (1.56) a	7.11 (2.14) a	3.06 (0.54) a	4.64 (0.84) a
Undercut banks	0.70 (0.17) a	1.20 (0.18) a	1.23 (0.27) a	1.19 (0.24) a	1.17 (0.27) a	1.46 (0.33) a	0.89 (0.23) a	1.10 (0.21) a
Artificial structures	1.38 (0.43) a	0.56 (0.24) a	0.66 (0.29) a	0.13 (0.06) a	0.77 (0.39) a	0.43 (0.27) a	1.13 (0.58) a	0.14 (0.06) a
Riparian disturbance								
Riparian disturbance all types	1.66 (0.21) a	0.74 (0.11) b	1.28 (0.18) a	0.76 (0.12) b	0.61 (0.17) b	0.81 (0.21) a	0.82 (0.19) a	0.63 (0.11) a
Riparian disturbance non-agricultural	1.56 (0.23) a	0.31 (0.09) b	0.61 (0.20) a	0.08 (0.03) b	0.39 (0.13) a	0.48 (0.14) a	0.40 (0.17) ab	0.10 (0.04) b
Riparian disturbance agricultural	0.10 (0.04) a	0.42 (0.07) b	0.67 (0.09) a	0.67 (0.11) a	0.22 (0.08) b	0.33 (0.11) a	0.42 (0.11) a	0.54 (0.10) a
Riparian disturbance walls	0.23 (0.08) a	0.01 (0) b	0.04 (0.02) a	0.01 (0.01) a	0.01 (0.01) a	0.03 (0.01) a	0.01 (0.01) a	0.01 (0.01) a
Riparian disturbance pipes	0.06 (0.02) a	0.01 (0.01) b	0.01 (0.00) a	0.01 (0.01) a	0.02 (0.02) a	0.03 (0.02) a	0.00 (0.00) a	0.00 (0.00) a
Habitat indices								
Rapid habitat index	15.04 (0.56) a	16.48 (0.36) b	15.70 (0.78) a	17.09 (0.61) a	16.37 (0.48) a	16.07 (0.49) a	15.99 (0.66) a	17.59 (0.65) a
Habitat quality index	16.12 (0.56) a	18.01 (0.37) b	17.14 (0.97) a	17.69 (0.60) a	18.31 (0.47) a	18.16 (0.48) a	17.33 (0.75) a	18.37 (0.62) a
Number of sites	34	57	32	26	33	30	27	34

TABLE 15.—Comparison of fish community metrics for significance among land use types, ecoregions, and stream order. Differing letters (e.g., a, b) indicate significantly different means (P<0.05). Numbers are weighted means (standard error).

Metric	Land Use		Ecoregion			Stream order		
	Urban	Nonurban	TBP	ECTP	SCP	2	3	4
Total number of fish species	12.96 (0.71) a	17.51 (0.67) b	14.00 (0.95) a	17.88 (0.86) b	18.18 (0.97) b	18.18 (1.11) a	16.12 (1.11) a	17.57 (0.90) a
Number of native cyprinid species	3.42 (0.30) a	3.93 (0.27) a	3.66 (0.30) a	4.27 (0.26) a	3.84 (0.44) a	3.94 (0.58) a	3.70 (0.26) a	4.15 (0.24) a
Number of benthic invertivore species	0.83 (0.22) a	3.00 (0.32) b	1.46 (0.18) a	2.64 (0.20) b	3.52 (0.50) b	3.24 (0.61) a	2.75 (0.39) a	2.63 (0.38) a
Number of sunfish species	3.54 (0.21) a	4.73 (0.20) b	3.86 (0.46) a	4.42 (0.28) ab	5.06 (0.26) b	4.84 (0.29) a	4.60 (0.43) a	4.54 (0.29) a
Number of intolerant species	0.67 (0.16) a	2.11 (0.24) b	1.30 (0.23) a	2.18 (0.23) b	2.24 (0.38) b	2.27 (0.49) a	1.69 (0.25) a	2.17 (0.19) a
% of individuals as tolerant species (excluding western mosquitofish)	33.12 (3.49) a	17.51 (1.93) b	25.38 (4.87) a	24.53 (3.89) a	12.86 (2.05) b	12.87 (2.46) a	21.79 (3.46) b	21.66 (3.54) b
% of individuals as omnivores	4.20 (0.80) a	2.51 (0.43) a	2.41 (0.37) a	3.11 (1.13) a	2.39 (0.53) a	2.87 (0.66) a	2.61 (0.93) a	2.04 (0.33) a
% of individuals as invertivores	74.20 (4.36) a	86.78 (1.45) b	82.55 (5.27)a	88.98 (1.50)a	86.39 (1.67)a	84.43 (2.17) a	85.83 (3.03) ab	90.09 (1.07) b
% of individuals as piscivores	9.31 (1.74) a	8.13 (0.66) a	6.54 (1.14) a	7.85 (1.08) a	8.84 (0.90) a	8.94 (1.17) a	8.21 (0.92) a	6.88 (0.97) a
Number of individuals/seine haul	29.81 (6.15) a	15.89 (2.64) b	32.30 (8.16) a	20.73 (3.86) a	9.10 (1.98) b	9.00 (2.54) a	22.73 (5.93) b	21.24 (3.05) b
Number of individuals/min electrofishing	13.35 (1.70) a	6.84 (0.65) b	8.63 (2.81) a	7.42 (0.77) a	6.46 (0.63) a	5.89 (0.61) a	8.47 (1.54) a	7.19 (0.76) a
% of individuals as non-native species	0.65 (0.33) a	0.81 (0.49) a	0.35 (0.13) a	0.14 (0.08) a	1.25 (0.82) a	0.85 (0.73) a	0.13 (0.07) a	1.60 (1.36) a
% of individuals with disease or other anomaly	0.23 (0.08) a	0.30 (0.09) a	0.04 (0.02) a	0.17 (0.05) b	0.45 (0.14) b	0.36 (0.17) a	0.27 (0.10) a	0.24 (0.11) a
Aquatic life use	2.58 (0.12) a	3.06 (0.09) b	2.97 (0.19) a	2.84 (0.12) a	3.15 (0.12) a	3.07 (0.13) a	3.10 (0.16) a	2.92 (0.14) a
Number of sites	34	57	32	26	33	30	27	34

TABLE 16.—Spearman rank correlation coefficients ($P<0.05$) between fish assemblage metrics and water quality parameters. Metric acronyms are defined in tables 1 and 2.

Parameter	TNOS	CYPR	BENT	SUN	INTSP	ETOL	OMNI	INVERT	PISC	SEIN	MIN	NONST	DIS
Mean dissolved oxygen				-0.29					-0.30				
Minimum dissolved oxygen				-0.22					-0.22				
Ammonia						0.23							
Nitrate				-0.27			0.29				-0.23		
Total kjeldahl nitrogen			-0.25			0.27			0.26				
Total nitrogen			-0.26			0.33	0.24						
Total phosphorus												-0.31	
Sulfate	-0.23		-0.35	-0.26	-0.25	0.40			-0.22	0.31			
Total dissolved solids	-0.22		-0.31			0.28				0.31			
Total suspended solids						0.22	0.25				-0.42		
Volatile suspended solids						0.22	0.26		0.22		-0.29		
Turbidity	0.33	0.30	0.36					0.25			-0.31		
Mean pH	-0.21		-0.40	-0.33	-0.23	0.31	0.22		-0.32	0.52	0.24		
Max temperature			-0.29			0.29				0.38			
Mean temperature			-0.24			0.31				0.42			

TABLE 17.—Spearman rank correlation coefficients (P<0.05) between fish assemblage metrics and habitat parameters. Metric acronyms are defined in tables 1 and 2. Derivation of habitat and disturbance values are described in Kaufmann et al. (1999).

Parameter	TNOS	CYPR	BENT	SUN	INTSP	ETOL	OMNI	INVERT	PISC	SEIN	MIN	NONST	DIS
Drainage area						0.23				0.23			
Discharge		0.30		-0.22	0.32		0.22		-0.24		-0.32		
Channel sinuosity	0.34	0.23	0.37		0.39	-0.29				-0.49	-0.29	-0.22	
Reach slope								-0.21			0.26	0.22	
Wetted width										0.45			
Mean thalweg depth					0.23						-0.42		
Mean residual pool depth											-0.22		
Mean width/depth ratio	-0.21		-0.26	-0.24	-0.32					0.39	0.51		
Entrenchment ratio	0.28		0.30	0.37	0.35				0.29	-0.38	-0.29		
Percent riffle habitat			-0.24										
Percent glide habitat			0.28	-0.26	0.32		0.26		-0.21				
Percent pool habitat				0.29	-0.25		-0.23		0.26				
Substrate percent fines				0.23				0.21					
Substrate percent sand			0.27		0.27	-0.27				-0.22	-0.23	-0.23	
Substrate percent sand and fines	0.29		0.48	0.26	0.41		-0.22	0.22		-0.27	-0.37	-0.31	
Substrate ≤16 mm	0.27		0.51		0.44		-0.21	0.22		-0.29	-0.39	-0.33	
Substrate ≥ 16 mm	-0.25		-0.51		-0.44		0.22	-0.23		0.30	0.40	0.34	
Substrate percent bedrock	-0.23		-0.29		-0.40			-0.26			0.31		
Mean embeddedness	0.24		0.42		0.37				0.24	-0.36	-0.44	-0.38	
Substrate $\log_{10}$ mean diameter	-0.24		-0.41	-0.25	-0.34		0.24	-0.28			0.32	0.26	
Relative bed stability	-0.24		-0.43		-0.40			-0.35			0.29		
Mean percent canopy cover midstream			0.38		0.30					-0.56	-0.47	-0.24	
Mean percent canopy cover at banks			0.33							-0.41	-0.40		
Riparian canopy cover			0.32			-0.25				-0.26	-0.30		
Riparian woody cover											-0.26		
Riparian 3-layers presence (% reach)											-0.29		
Large woody debris pieces			0.26		0.32				0.22	-0.34	-0.41		
Large woody debris volume	0.23		0.34		0.36				0.22	-0.43	-0.44		
Fish cover all types	0.40		0.40	0.42	0.42				0.32	-0.34	-0.35		
Fish cover natural	0.41		0.39	0.41	0.40				0.30	-0.33	-0.35		
Fish cover brush and small debris	0.45		0.52	0.39	0.49				0.27	-0.34	-0.37	-0.23	
Fish cover large woody debris	0.22		0.35		0.37					-0.27	-0.42		
Fish cover leaf litter	0.26		0.50	0.26	0.33	-0.24	-0.28		0.23	-0.28		-0.23	
Fish cover artificial structures			-0.25										
Riparian disturbance all types	-0.21		-0.22										
Riparian disturbance non-agricultural	-0.28	-0.26	-0.32		-0.25			-0.31			0.25		
Riparian disturbance agricultural					0.22			0.23					-0.25
Riparian disturbance walls, revetments	-0.24		-0.33		-0.21								
Rapid habitat index	0.38	0.39	0.21	0.30	0.30								
Habitat quality index	0.31	0.36	0.25		0.31					-0.28			

TABLE 18.—Metric contribution to the two regional IBIs. Analysis is based on a forward stepwise regression of scored metrics versus total score. The values across the bottom explain the proportion of variance explained by the combination of metrics. The metrics are displayed in decreasing order of significance. Metric acronyms are defined in tables 1 and 2.

Texas Blackland Prairies

SUN	SUN	SUN	SUN	NONST	NONST	NONST	NONST	BENT	BENT	BENT
	BENT	BENT	NONST	BENT	BENT	BENT	BENT	NONST	NONST	NONST
		NONST	BENT	SUN	SUN	SUN	SUN	CYPR	ETOL	ETOL
			CYPR	CYPR	CYPR	CYPR	CYPR	ETOL	PISC	PISC
				ETOL	ETOL	ETOL	ETOL	PISC	CYPR	CYPR
					TNOS	TNOS	TNOS	SUN	TIND	TIND
						PISC	PISC	DIS	TNOS	TNOS
							DIS	TNOS	SUN	SUN
								TIND	INVERT	INVERT
									DIS	DIS
										OMNI
0.26	0.55	0.72	0.82	0.85	0.88	0.90	0.94	0.96	0.98	1.00

East Central Texas Plains/South Central Plains

BENT	BENT	BENT	BENT	BENT	BENT	BENT	CYPR	BENT	PISC	PISC	PISC
	TNOS	TNOS	SUN	CYPR	CYPR	CYPR	BENT	PISC	BENT	BENT	BENT
		SUN	CYPR	SUN	PISC	PISC	PISC	ETOL	SUN	SUN	SUN
			TNOS	PISC	SUN	ETOL	ETOL	CYPR	TIND	TIND	TIND
				TNOS	ETOL	SUN	SUN	SUN	ETOL	ETOL	ETOL
					TNOS	DIS	DIS	DIS	DIS	DIS	DIS
						TNOS	NONST	NONST	INTSP	INTSP	INTSP
							TNOS	TIND	CYPR	CYPR	CYPR
								TNOS	TNOS	TNOS	TNOS
									NONST	NONST	NONST
										OMNI	OMNI
											INVERT
0.56	0.73	0.78	0.81	0.84	0.88	0.91	0.94	0.96	0.98	1.00	1.00

TABLE 19.—Comparison of invertebrate community metrics among land use types, ecoregions, and stream order. Differing letters (e.g., a, b) indicate significantly different means. Numbers are weighted means (standard error). Metric acronyms are defined in Table 3.

		Land Use		Ecoregion			Stream Order		
Metric	**Sample Type**	**Urban**	**Nonurban**	**TBP**	**ECTP**	**SCP**	**2**	**3**	**4**
N	**Benthic**[1]	257.35 (39.94)a	168.54 (32.92)a	192.23 (42.96)a	151.80 (33.68)a	173.78 (51.95)a	161.10 (70.43)a	152.44 (28.03)a	213.50 (35.12)a
	Snag[2]	479.40 (74.88)a	412.51(48.33)a	463.15 (109.25)a	493.92 (82.17)a	364.66 (62.39)a	223.29 (37.72)a	499.40 (78.55)b	617.42 (95.27)b
TAXA	**Benthic**	29.58 (1.32)a	32.74 (1.95)a	35.30 (3.73)a	30.12 (2.97)a	32.88 (2.74)a	30.24 (3.72)a	32.36 (2.23)a	36.87 (2.72)a
	Snag	13.05 (0.93)a	16.21 (0.92)b	16.49 (2.10)a, b	20.47 (1.17)a	14.06 (1.09)b	13.16 (1.36)a	15.20 (0.97)a	22.04 (1.37)b
DTAX	**Benthic**	53.78 (2.68)a	39.76 (2.10)b	30.89 (3.05)a	44.34 (3.44)b	41.45 (2.83)b	39.68 (2.85)a	41.19 (4.19)a	39.95 (3.51)a
	Snag	58.91 (3.04)a	46.51 (2.03)b	48.01 (5.33)a	43.96 (2.30)a	47.91 (2.85)a	46.56 (3.28)a, b	51.51 (3.55)a	41.69 (2.52)b
DFFG	**Benthic**	35.57 (1.09)a	40.57 (1.62)b	43.71 (2.36)a	39.63 (2.07)a	39.66 (2.42)a	39.13 (2.92)a	39.21 (2.09)a	43.96 (2.64)a
	Snag	31.73 (1.28)a	39.00 (1.20)b	43.32 (2.39)a	39.33 (2.18)a, b	37.03 (1.46)b	37.11 (1.83)a	38.25 (2.12)a	42.05 (1.98)a
PRED	**Benthic**	18.26 (1.31)a	23.28 (2.07)b	14.47 (3.18)a	20.14 (2.33)a, b	27.18 (2.93)b	26.45 (3.32)a	20.32 (2.74)a	21.28 (3.94)a
	Snag	20.22 (1.06)a	13.12 (1.06)b	11.92 (2.69)a	13.69 (1.31)a	13.70 (1.45)a	14.20 (1.74)a	12.19 (1.71)a	13.55 (1.72)a
COLG	**Benthic**	31.07 (1.33)a	33.15 (1.52)a	39.57 (3.53)a	35.58 (1.63)a	29.89 (1.85)b	28.28 (2.09)a	37.46 (2.27)b	35.23 (2.28)b
	Snag	28.62 (1.43)a	34.25 (1.51)b	40.84 (3.07)a	32.30 (2.05)b	32.66 (1.95)b	30.94 (2.48)a	35.40 (2.29)a	37.40 (2.26)a
EPT	**Benthic**	5.77 (0.50)a	8.57 (0.64)b	10.32 (2.09)a	9.13 (0.82)a	7.60 (0.72)a	7.11 (0.89)a	8.03 (0.86)a, b	11.26 (1.28)b
	Snag	3.72 (0.41)a	5.64 (0.45)b	5.64 (0.99)a, b	7.42 (0.54)a	4.74 (0.59)b	4.52 (0.57)a	5.47 (0.77)a, b	7.43 (0.79)b
HBI	**Benthic**	6.14 (0.08)a	5.63 (0.09)b	5.39 (0.32)a	5.57 (0.17)a	5.77 (0.09)a	5.68 (0.14)a	5.74 (0.15)a	5.49 (0.18)a
	Snag	5.98 (0.14)a	5.60 (0.12)b	5.61 (0.36)a	5.38 (0.18)a	5.72 (0.14)a	5.60 (0.20)a	5.81 (0.20)a	5.38 (0.15)a
INTO	**Benthic**	0.20 (0.04)a	0.71 (0.11)b	1.20 (0.35)a	0.69 (0.18)a	0.53 (0.10)a	0.52 (0.10)a	0.65 (0.20)a	1.02 (0.24)a
	Snag	0.33 (0.09)a	0.99 (0.19)b	1.38 (0.44)a	1.58 (0.49)a	0.57 (0.16)a	0.68 (0.28)a	0.93 (0.34)a	1.50 (0.33)a
NOIN	**Benthic**	7.60 (0.53)a	6.47 (0.57)a	6.70 (1.05)a	5.47 (0.61)a	6.91 (0.85)a	5.57 (1.01)a	7.35 (0.93)a	6.97 (0.53)a
	Snag	2.73 (0.32)a	2.22 (0.28)a	2.84 (0.78)a	2.67 (0.31)a	1.85 (0.35)a	1.58 (0.41)a	2.47 (0.47)a, b	2.99 (0.37)b
ELM	**Benthic**	4.41 (1.03)a	12.19 (1.79)b	13.87 (1.73)a, b	17.13 (2.86)a	9.01 (2.15)b	7.11 (1.95)a	14.83 (3.55)a, b	15.99 (2.77)b
	Snag	8.31 (2.57)a	29.66 (3.41)b	31.04 (8.49)a	27.03 (3.60)a	29.06 (4.83)a	27.80 (6.00)a	27.91 (5.94)a	32.02 (3.54)a
HYDR	**Benthic**	38.14 (7.35)a	30.63 (5.05)a	22.55 (6.95)a	31.26 (7.35)a	33.40 (7.56)a	37.33 (9.410)a	25.18 (7.01)a	27.78 (6.01)a
	Snag	34.36 (7.63)a	41.34 (6.16)a	27.77 (9.93)a	60.06 (6.96)b	37.04 (9.23)a, b	42.73 (11.63)a	31.55 (8.38)a	50.74 (7.82)a
CHIR	**Benthic**	47.56 (3.75)a	30.14 (2.73)b	15.77 (2.93)a	31.57 (4.39)b	35.18 (3.57)b	31.97 (3.82)a	30.87 (5.87)a	28.58 (3.37)a
	Snag	55.54 (3.82)a	30.49 (2.95)b	22.09 (3.46)a	30.25 (4.03)a, b	34.83 (4.31)b	34.69 (5.35)a	31.74 (4.93)a	25.50 (3.13)a
ALU score	**Benthic**	28.76 (0.52)a	29.53 (0.49)a	31.68 (1.01)a	29.62 (0.70)a, b	28.75 (0.67)b	28.53 (0.89)a	30.00 (0.57)a	30.43 (0.84)a
	Snag	25.17 (0.45)a	25.69 (0.55)a	26.19 (1.45)a, b	27.59 (0.93)a	24.66 (0.62)b	24.92 (0.98)a	25.55 (0.87)a	27.05 (0.54)a

[1]Mean number/0.5 m^2 per transect among sites sampled

[2]Mean number/L among sampling sites

TABLE 20.—Forward stepwise regression of scored invertebrate metrics for the total invertebrate community index. Analysis is based on a forward stepwise regression of scored metrics versus total score. The values across the bottom explain the proportion of variance explained by the combination of metrics. The metrics are displayed in decreasing order of significance. Metric acronyms are defined in Table 3.

Benthic											
EPT	EPT	EPT	EPT	EPT	EPT	EPT	EPT	ELM	ELM	ELM	ELM
	HYDR	HYDR	HYDR	HYDR	DTAX	DTAX	ELM	EPT	HYDR	HYDR	HYDR
		ELM	ELM	DTAX	HYDR	HYDR	HYDR	HYDR	EPT	PRED	PRED
			DTAX	ELM	ELM	ELM	DTAX	DTAX	NOIN	DTAX	DTAX
				COLG	COLG	COLG	COLG	NOIN	PRED	EPT	EPT
					NOIN	NOIN	NOIN	COLG	DTAX	COLG	COLG
						PRED	PRED	HBI	COLG	HBI	HBI
							HBI	PRED	HBI	DFFG	DFFG
								DFFG	DFFG	NOIN	NOIN
									CHIR	CHIR	CHIR
										TAXA	TAXA
											INTO
0.37	0.49	0.59	0.69	0.75	0.82	0.87	0.91	0.94	0.97	0.99	1.00
Snag											
TAXA	TAXA	ELM	TAXA	TAXA	TAXA	HBI	HBI	HBI	HBI	ELM	ELM
	ELM	TAXA	ELM	HBI	HBI	TAXA	ELM	ELM	ELM	HYDR	HYDR
		HBI	HBI	HYDR	HYDR	ELM	HYDR	HYDR	HYDR	HBI	HBI
			HYDR	COLG	ELM	COLG	NOIN	NOIN	DFFG	DFFG	DFFG
				ELM	COLG	HYDR	COLG	COLG	COLG	NOIN	NOIN
					DFFG	DFFG	TAXA	TAXA	NOIN	PRED	PRED
						NOIN	EPT	EPT	PRED	COLG	COLG
							DFFG	DFFG	TAXA	CHIR	CHIR
								PRED	EPT	EPT	EPT
									CHIR	TAXA	TAXA
										DTAX	DTAX
											INTO
0.38	0.55	0.63	0.72	0.81	0.84	0.87	0.91	0.93	0.96	0.99	1.00

TABLE 21.—Spearman rank correlation coefficients (P<0.05) for invertebrate metrics based on collections from benthic and snag habitats in relation to habitat and water quality parameters. Metric acronyms are defined in the Table 3.

Parameter	Sample Type	N	TAXA	EPT	HBI	CHIR	DTAX	DFFG	PRED	INTO	HYDR	NOIN	COLG	ELM	ALU
Morphology															
Drainage area	Benthic			0.39	-0.29	-0.24		0.35	-0.27	0.32			0.31	0.32	
	Snag	0.34	0.52	0.41	-0.24	-0.34	-0.28	0.35	-0.26	0.38			0.23	0.27	0.30
Discharge	Benthic			0.40	-0.28		-0.25		-0.32	0.22	0.34	-0.21	0.24	0.37	
	Snag	0.24	0.40	0.62	-0.46	-0.28	-0.34	0.22	-0.28	0.36	0.51			0.25	
Channel sinuosity	Benthic	-0.25		0.22		-0.23	-0.36			0.27				0.32	
	Snag			0.25		-0.24	-0.22	0.28	-0.31		0.26			0.32	
Mean width/depth ratio	Benthic	0.31			0.23		0.25	-0.27		-0.29				-0.35	
	Snag							-0.21	0.22				-0.21	-0.39	
Percent pools of reach	Benthic			-0.33	0.26		0.35		0.26	-0.22	-0.41	0.21			-0.21
	Snag			-0.44	0.44	0.22	0.22		0.27	-0.32	-0.40				
Percent glides of reach	Benthic			0.38	-0.27		-0.37		-0.29	0.23	0.40			0.21	0.22
	Snag		0.22	0.51	-0.43	-0.24	-0.23		-0.32	0.32	0.43				
Canopy & Woody Debris															
Mean percent canopy-midstream	Benthic	-0.39			-0.32	-0.24	-0.29			0.33				0.31	
	Snag								-0.32				0.21	0.43	
Mean percent canopy-banks	Benthic	-0.32			-0.24	-0.26	-0.26			0.24				0.24	
	Snag								-0.29					0.37	
Riparian canopy & midlayer present	Benthic	-0.26			-0.25	-0.26	-0.24			0.24				0.26	
	Snag				-0.25	-0.23			-0.26	0.31				0.22	
Woody debris in/above bankfull channel	Benthic				-0.24			0.26		0.24			0.26	0.40	
	Snag	0.25	0.28	0.31	-0.24	-0.22		0.26	-0.29		0.31		0.30	0.51	
Woody debris in bankfull channel	Benthic	-0.23			-0.25		-0.22	0.28		0.26			0.24	0.41	
	Snag	0.28	0.21	0.27	-0.21			0.22	-0.30		0.26		0.29	0.55	

Table 21.—Continued

Parameter	Sample Type	N	TAXA	EPT	HBI	CHIR	DTAX	DFFG	PRED	INTO	HYDR	NOIN	COLG	ELM	ALU
<u>**Cover**</u>															
All types	Benthic							0.21		0.24			0.27	0.33	
	Snag		0.26								0.26			0.34	
Natural types	Benthic							0.23		0.26			0.29	0.34	
	Snag		0.27				-0.22	0.21			0.27			0.35	
Large	Benthic	-0.22						0.23		0.24			0.22	0.29	
	Snag			0.25	-0.26				-0.26		0.32			0.31	
Filamentous algae	Benthic	0.45	0.43					-0.29		-0.22		0.39		-0.25	
	Snag								0.23			0.28		-0.28	0.21
Brush & small debris	Benthic	-0.26			-0.27		-0.23	0.23		0.33			0.22	0.40	
	Snag		0.31	0.26				0.23	-0.23		0.26		0.21	0.48	
Large woody debris	Benthic	-0.29			-0.28	-0.23	-0.25	0.35		0.34			0.29	0.40	
	Snag		0.26	0.34	-0.28	-0.25	-0.21	0.31	-0.30	0.21	0.30		0.22	0.47	
<u>**Riparian Disturbance**</u>															
Non-agricultural disturbance	Benthic			-0.40		0.27	0.26			-0.23				-0.23	-0.21
	Snag		-0.22	-0.29		0.25	0.25	-0.21	0.21	-0.24				-0.24	
Agricultural disturbance	Benthic	-0.23				-0.27	-0.21	0.33		0.27		-0.34			
	Snag					-0.29				0.29					
<u>**Habitat Indices**</u>															
HQI	Benthic		0.25	0.41	-0.26		-0.22			0.29	0.33			0.33	0.30
	Snag	0.21	0.31	0.45	-0.32		-0.33	0.23	-0.24		0.36				
Rapid habitat index	Benthic		0.30	0.35					-0.23		0.28	0.22	0.24	0.27	0.23
	Snag	0.26	0.40	0.47	-0.23	-0.24	-0.31	0.23	-0.32		0.41				
<u>**Water Quality**</u>															
Turbidity	Benthic	-0.26	-0.24					0.25				-0.21		0.24	-0.23
	Snag								-0.22						
Dissolved oxygen-minimum	Benthic			0.22	-0.29		-0.23			0.25	0.24	-0.24			
	Snag			0.25	-0.32		-0.25				0.22	-0.34			

TABLE 22.—Analyses of invertebrate metrics from invertebrate samples collected from benthic and snag habitats, and an analysis of subsampling efforts for benthic samples. Values are means (standard error). Differing letters (e.g., a, b) indicate significantly different means (Kruskal-Wallis test with Tukey's multiple comparison test, P<0.05). Metric acronyms are defined in Table 3.

	Habitat sample type		Benthic subsample type[1]		
Metric	Benthic[2]	Snag[3]	Complete Sample	100 Subsample	200 Subsample
N	248.77 (33.15)a	508.55 (47.61)b			
TAXA	32.36 (1.21)a	16.78 (0.70)b	32.36 (1.21)a	23.22 (1.29)b	25.20 (1.44)b
EPT	8.05 (0.47)a	5.32 (0.34)b	8.05 (0.47)a	6.41 (0.52)a	6.88 (0.56)a
HBI	5.78 (0.07)a	5.68 (0.08)a	5.78 (0.07)a	5.12 (0.22)a	5.13 (0.22)a
CHIR	36.91 (2.33)a	37.68 (2.49)a	36.91 (2.33)a	31.67 (2.57)a	31.47 (2.60)a
DTAX	45.65 (1.84)a	49.23 (1.78)a	45.65 (1.84)a	38.84 (2.36)a	39.12 (2.40)a
DFFG	39.02 (1.08)a	37.19 (1.14)a	39.02 (1.08)a	35.47 (1.84)a	35.74 (1.89)a
PRED	20.86 (1.35)a	15.46 (0.82)b	20.86 (1.35)a	18.61 (1.72)a	18.36 (1.77)a
INTO	0.58 (0.09)a	0.92 (0.16)a	0.58 (0.09)a	0.50 (0.07)a	0.52 (0.08)a
HYDR	32.79 (3.70)a	41.51 (4.40)a	32.79 (3.70)a	32.08 (4.28)a	32.04 (4.19)a
NOIN	7.03 (0.35)a	2.75 (0.22)b	7.03 (0.35)a	5.01 (0.35)b	5.41 (0.37)b
COLG	33.29 (1.02)a	32.29 (1.17)a	33.29 (1.02)a	29.96 (1.61)a	29.86 (1.64)a
ELM	10.70 (1.32)a	22.03 (1.17)b	10.70 (1.32)a	8.90 (1.28)a	8.99 (1.32)a
ALU	29.50 (0.32)a	26.02 (0.34)b	29.50 (0.32)a	25.55 (1.21)b	25.78 (1.31)ab

[1]Includes only samples subjected to subsampling protocols
[2]Mean number/0.5 m^2 per transect among sites sampled
[3]Mean number/L among sampling sites

TABLE 23.—Stream distance (km) estimates for mercury in all species of whole fish (LCL= lower 95% confidence limit; UCL= upper 95% confidence limit). Although human health screening levels are intended for comparison to muscle tissue rather than whole fish, estimates are included here for comparison purposes.

Ecoregion	Hg, mg/kg	LCL	Stream km	UCL
TBP	<0.1[a]	1257	1267	1277
	≥0.1	0	4.9	15.0
	<0.7[b]	1272	1272	1272
	≥0.7		none predicted	
	Total		1272	
ECTP	<0.1	689	1093	1498
	≥0.1	441	845	1249
	<0.7	1938	1938	1938
	≥0.7		none predicted	
	Total		1938	
SCP	<0.1	0	488	1036
	≥0.1	2779	3327	3875
	<0.7	3074	3559	4044
	≥0.7	0	256	741
	Total		3815	

[a] Predator protection level (Eisler 1987)
[b] Texas Department of Health screening level for edible muscle tissue (fillet)

TABLE 24.—Spearman rank correlations of mercury concentration in whole longear sunfish composites with water quality parameters and fish size.

Parameter	r_s	P
pH	-0.54974	<0.0001
Sulfate	-0.52600	<0.0001
Total hardness	-0.50639	<0.0001
Total alkalinity	-0.45359	0.0002
Total dissolved solids	-0.40854	0.0008
Composite mean fish length	0.33278	0.0072
Composite mean fish weight	0.27167	0.0299
Total nitrogen	-0.25187	0.0447
Chloride	-0.18224	0.1495
Dissolved oxygen, mean	-0.17456	0.1712
Volatile suspended solids	-0.16729	0.1864
Total kjeldahl nitrogen	-0.15365	0.2254
Ammonia nitrogen	-0.13580	0.3051
Dissolved oxygen, minimum	-0.11631	0.3640
Chlorophyll *a*	0.08080	0.5256
Total organic carbon	0.07579	0.5517
Total suspended solids	-0.06361	0.6175
Turbidity	0.04634	0.7162
Total phosphorus	-0.04059	0.7521
Orthophosphate	0.03352	0.7926

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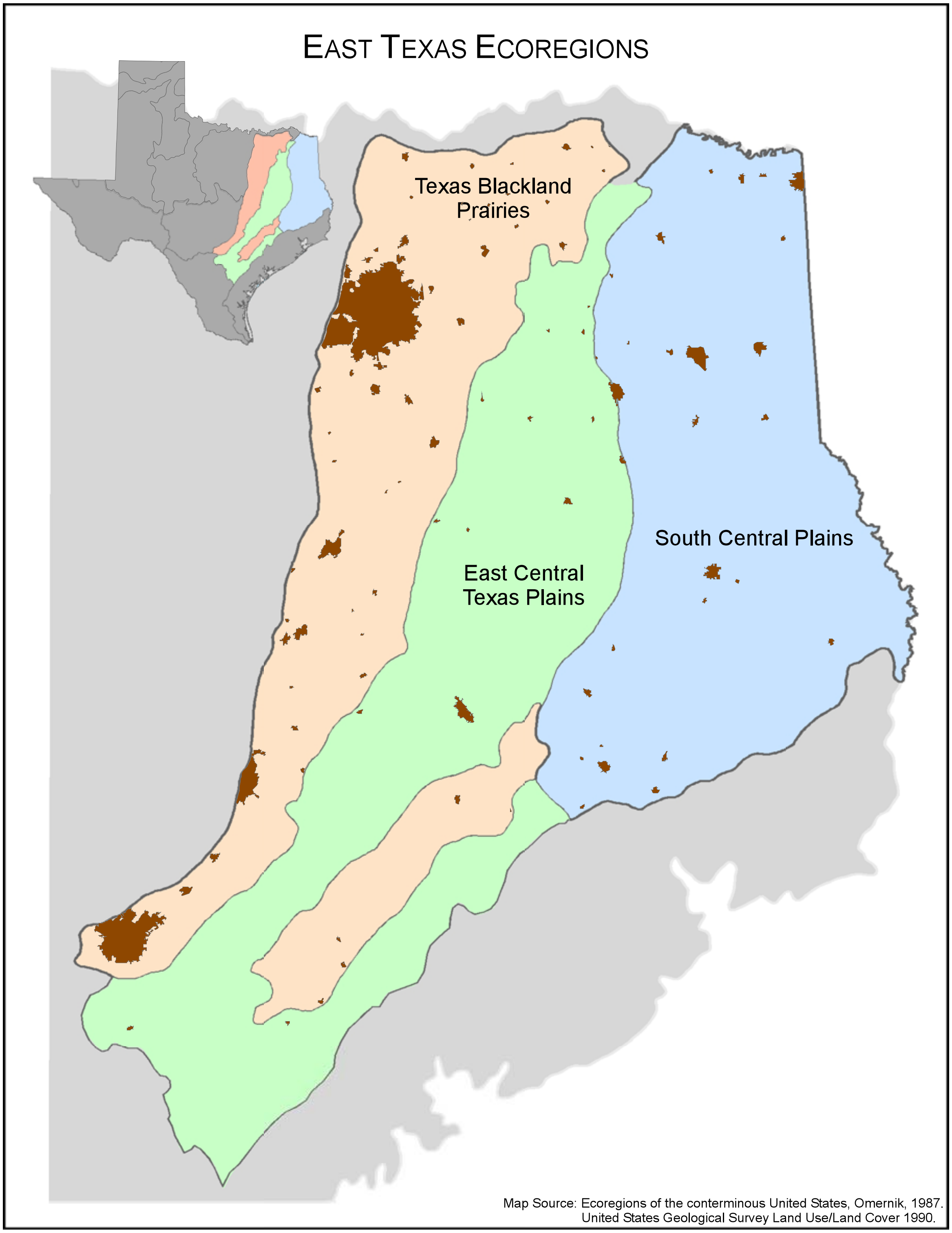

FIGURE 1.—Map of the Texas Blackland Prairies (TBP), East Central Texas Plains (ECTP), and South Central Plains (SCP) ecoregions including city limits (shaded brown areas).

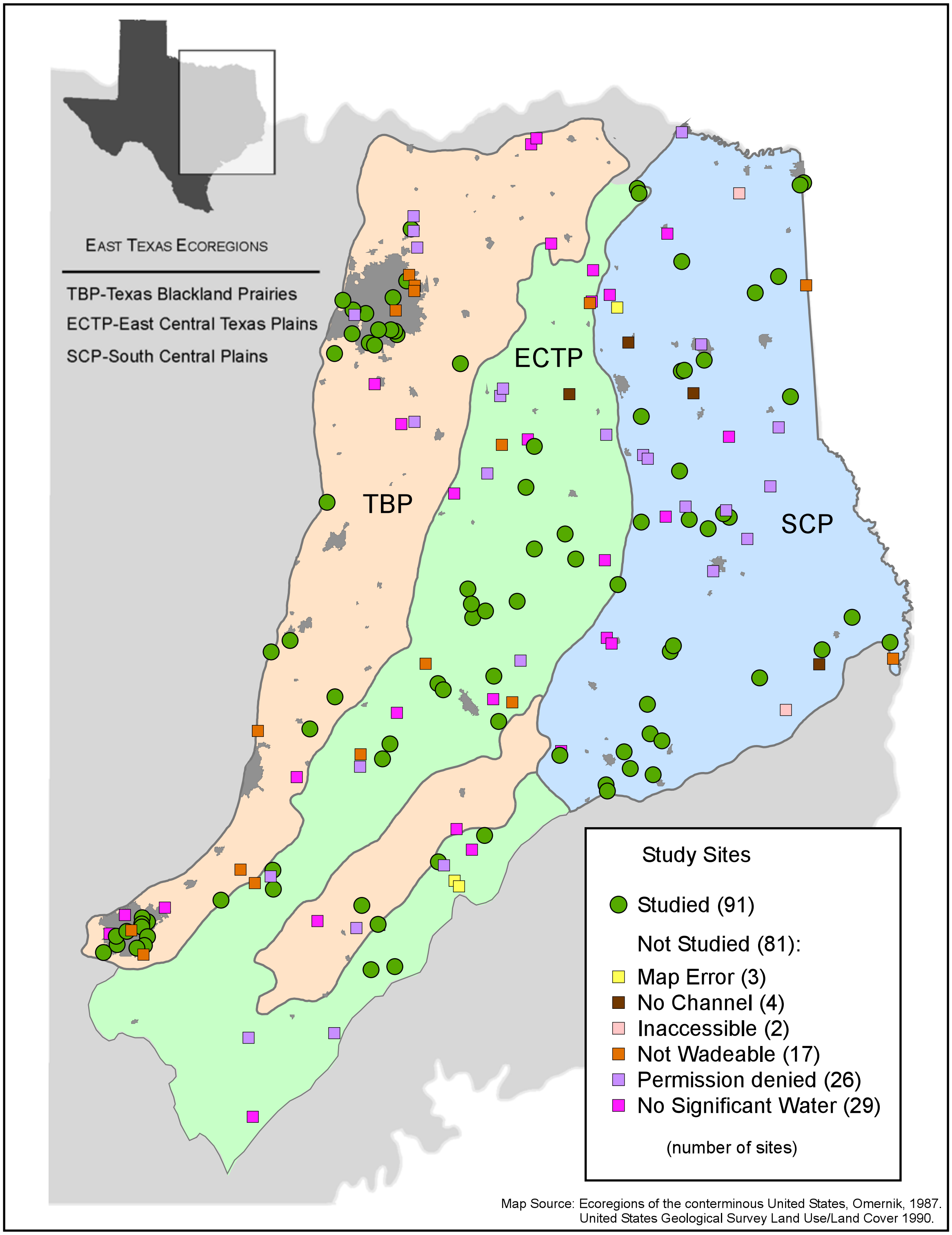

FIGURE 2.—Map of the sites evaluated for data collection in the study including causes for site elimination.

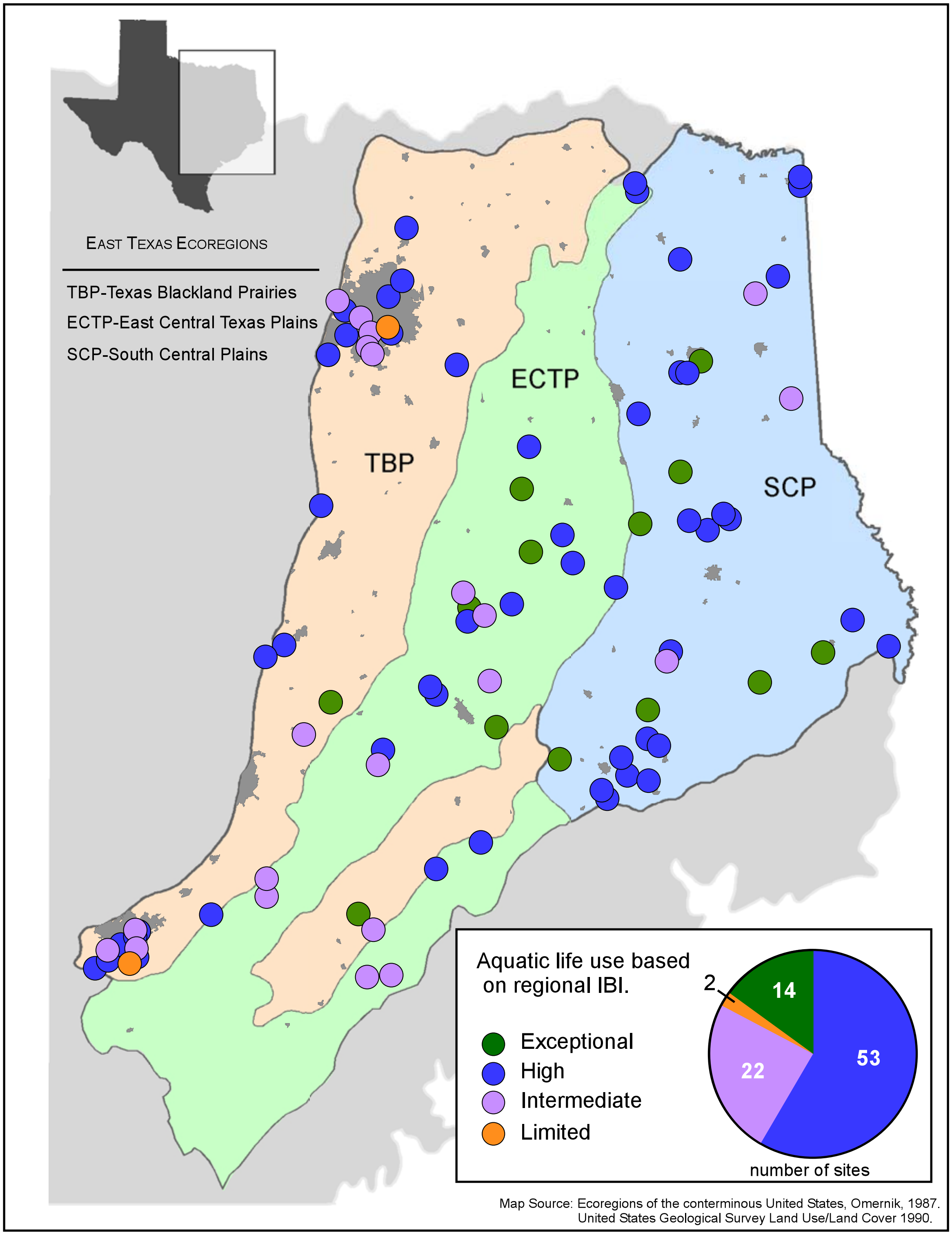

FIGURE 3.—Aquatic life use (ALU) based on regionally developed Index of Biotic Integrity (IBI) scores (Linam et al. 2002).

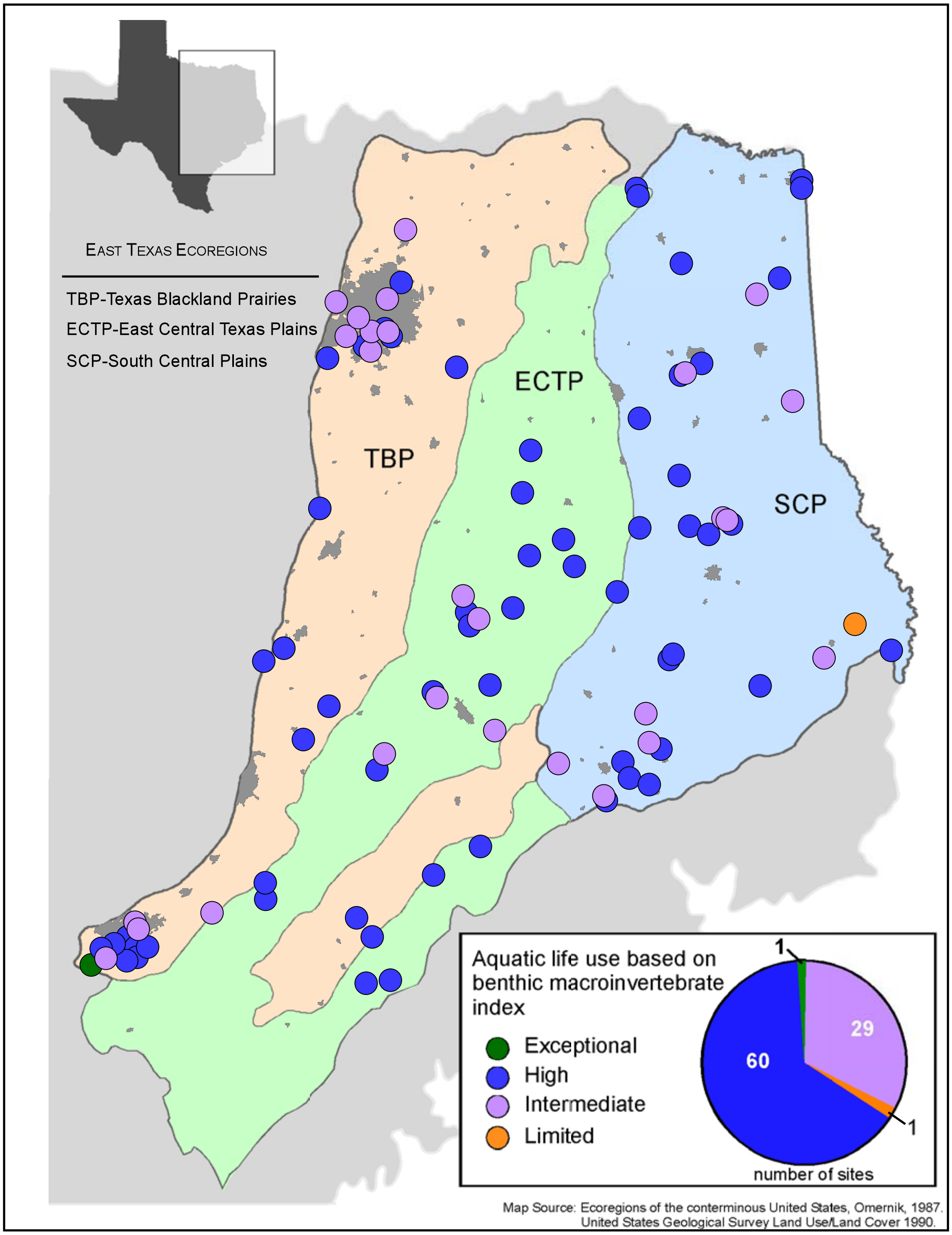

FIGURE 4.—Aquatic live use (ALU) based on benthic invertebrate index (TNRCC 1999a). Ranks represent only kicknet samples.

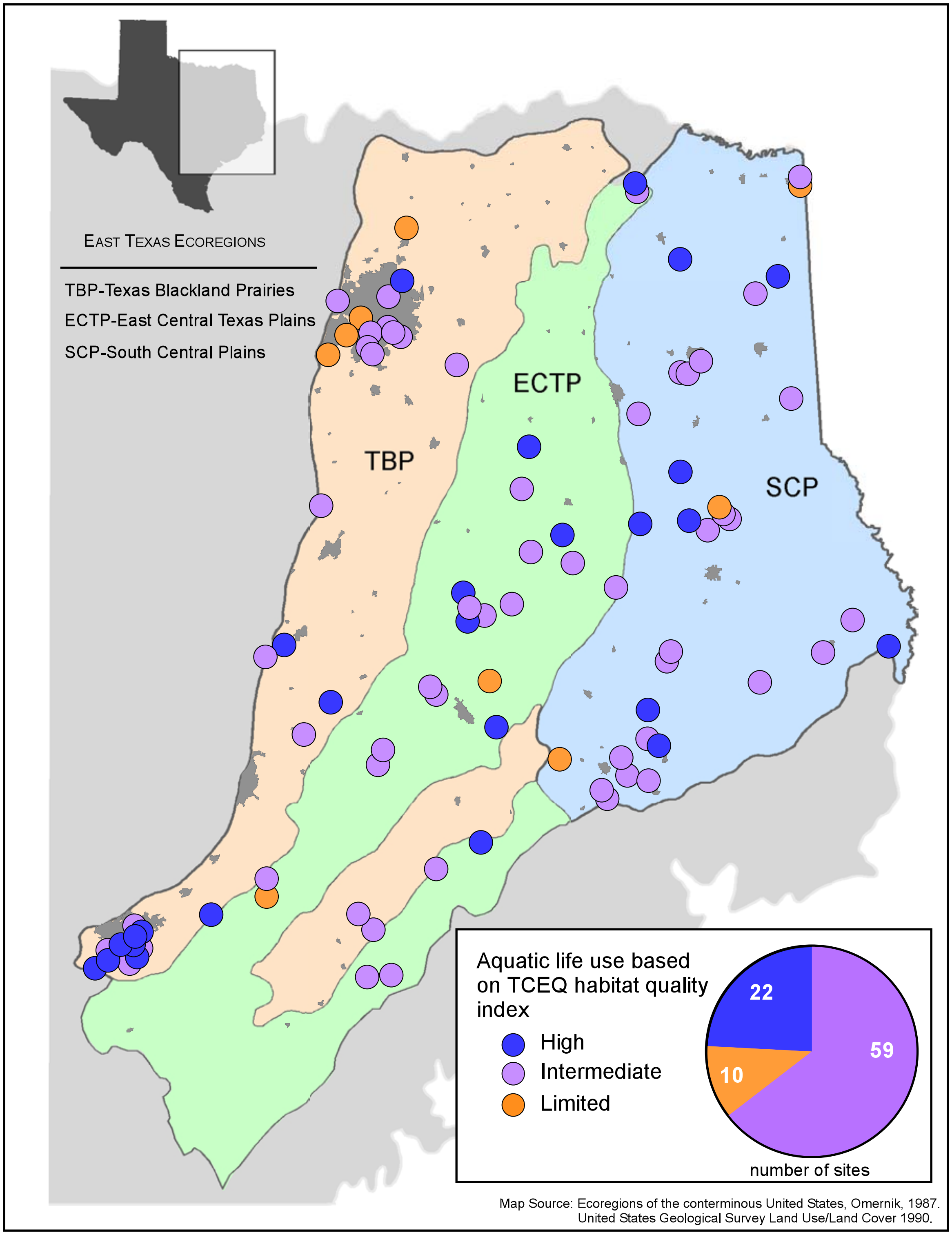

FIGURE 5.—Aquatic life use (ALU) based upon habitat quality index (HQI; TNRCC 1991a).

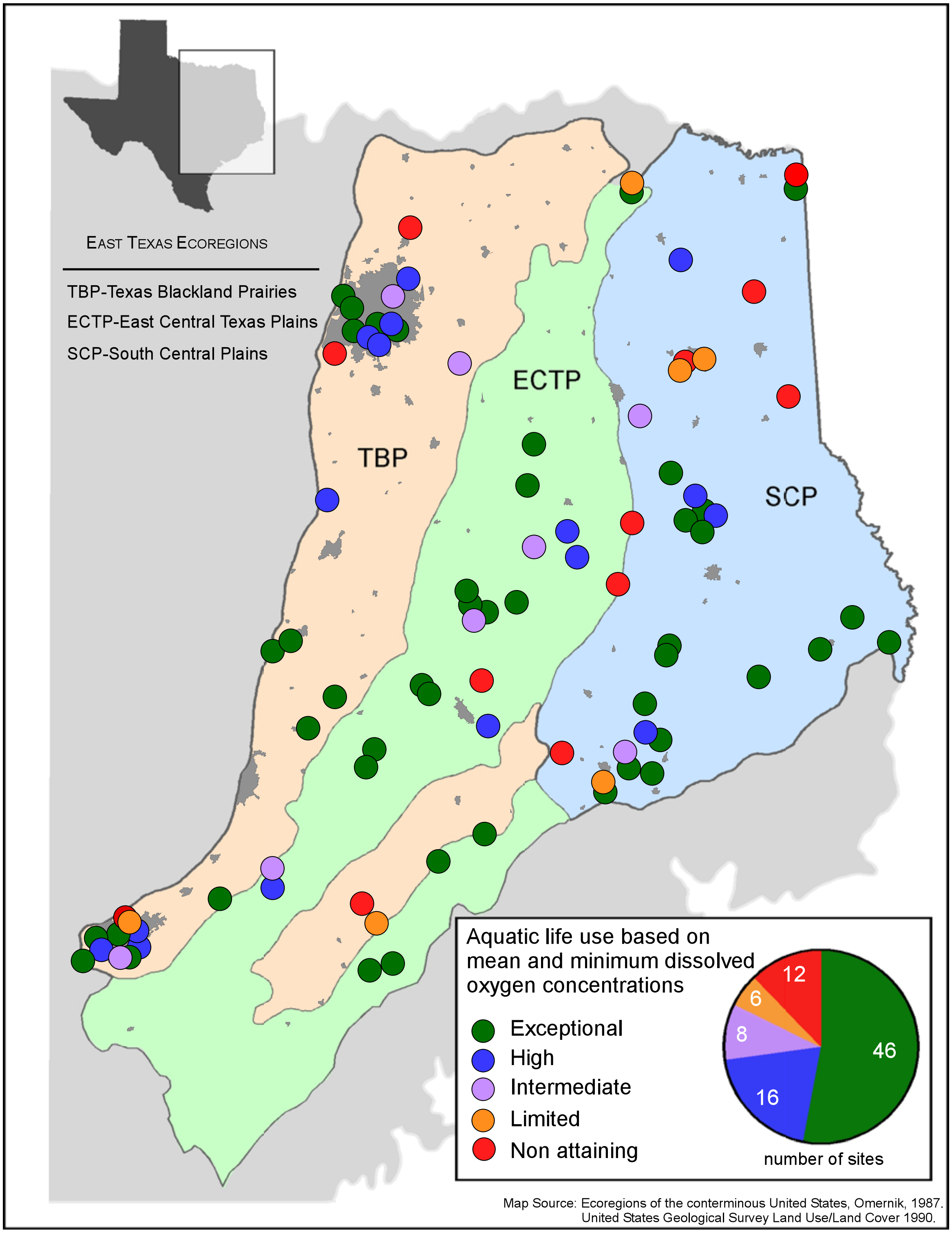

FIGURE 6.—Aquatic life use (ALU) based on mean and minimum dissolved oxygen concentrations. Rankings are based upon 24-hour measurements, with lower of the mean or minimum determining the use (TNRCC 1995).

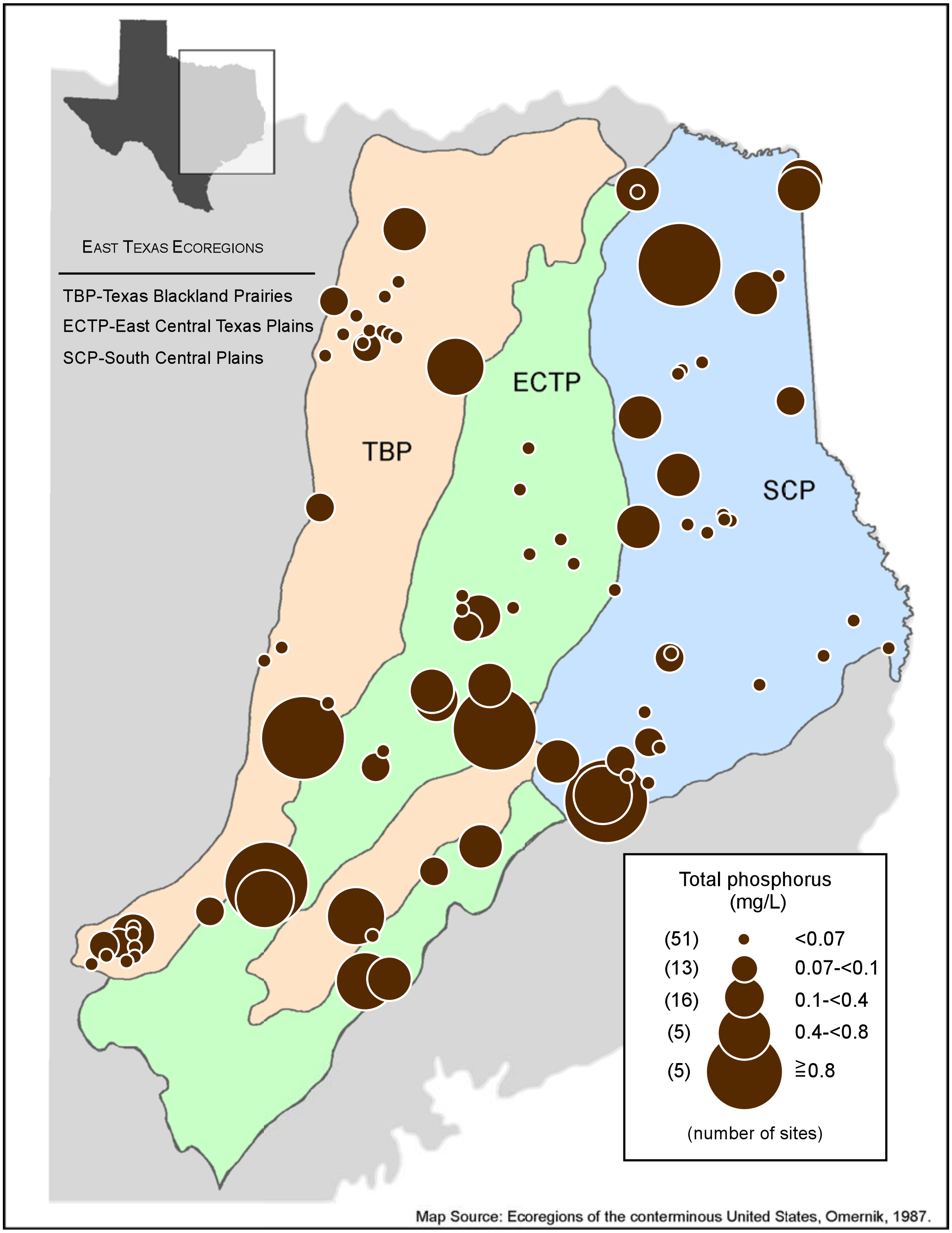

Figure 7.—Total phosphorus concentrations from analysis of single water samples from each site. The state 85th percentile screening level was 0.8 mg/L.

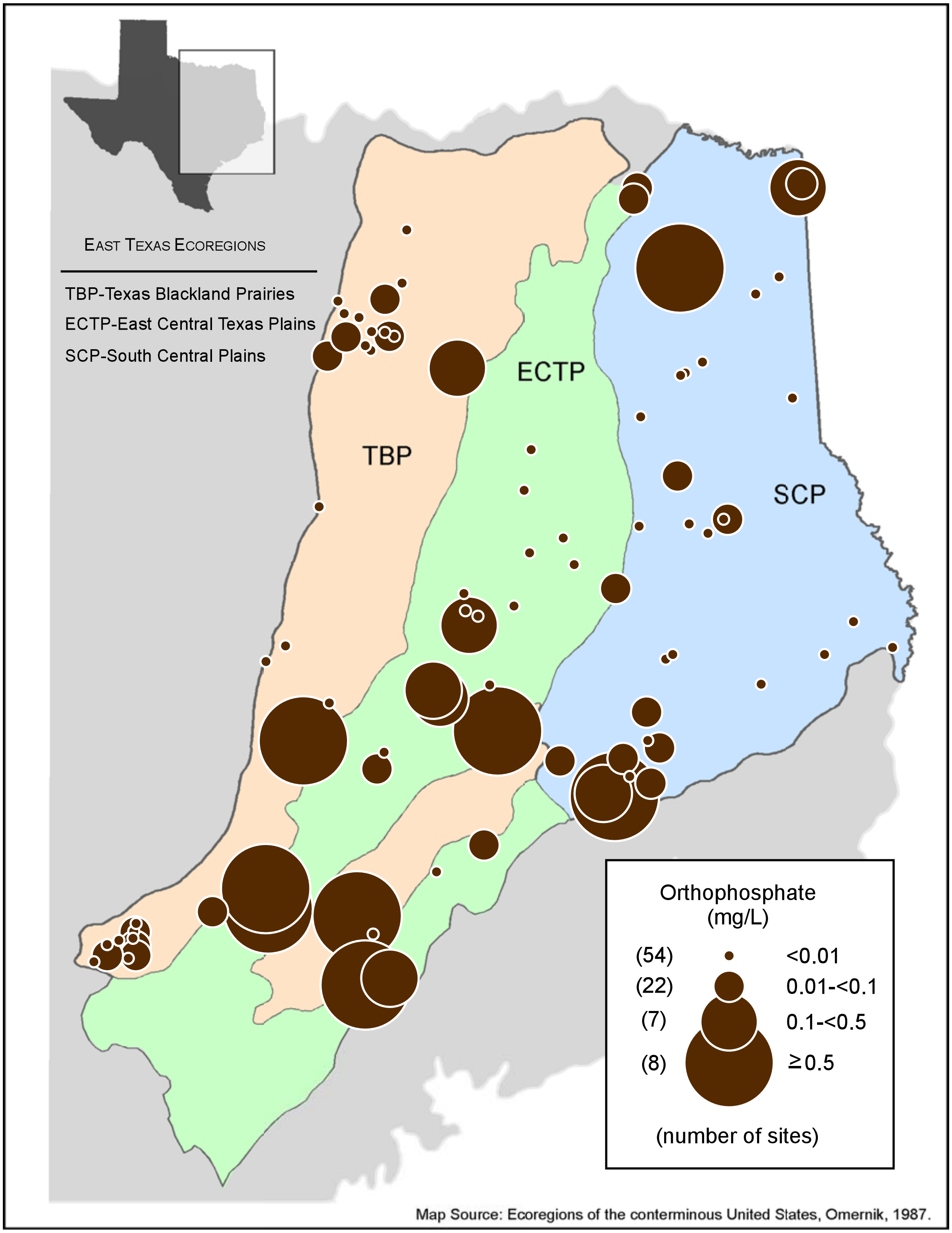

FIGURE 8.—Orthophosphate concentration from analysis of single water samples from each site. The state 85th percentile screening level was 0.5mg\L.

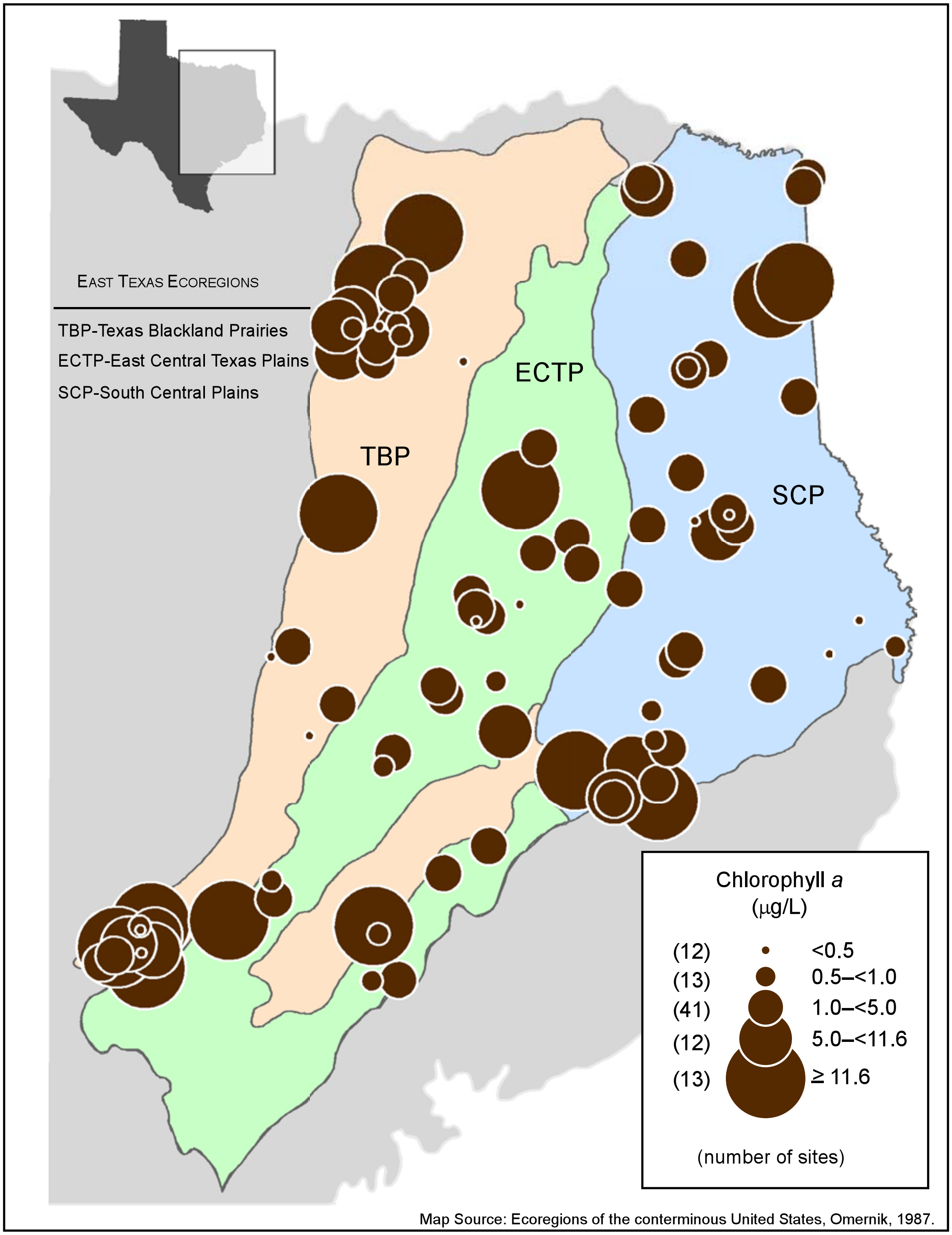

FIGURE 9.—Chlorophyll *a* concentration from analysis of single water samples from each site. The state 85th percentile screen level was 11.6 µg/L.

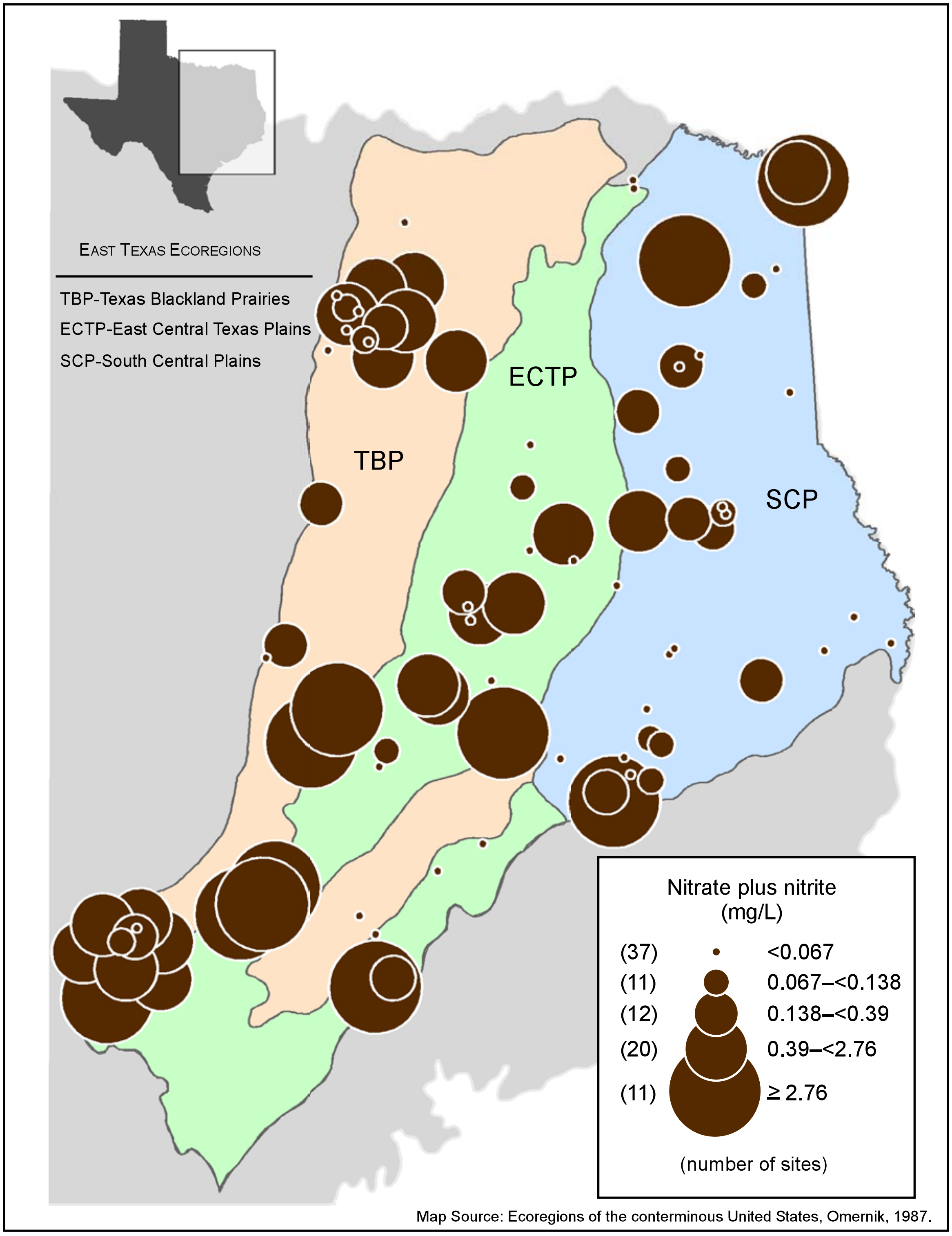

FIGURE 10.—Nitrate plus nitrite concentration from analysis of single water samples from each site. The state 85th percentile screening level was 2.76 mg\L; USEPA 25th percentile for TBP was 0.39 mg\L; USEPA 25th percentile for ECTP was 0.138 mg\L; and USEPA 25th percentile for SCP was 0.067 mg\L.

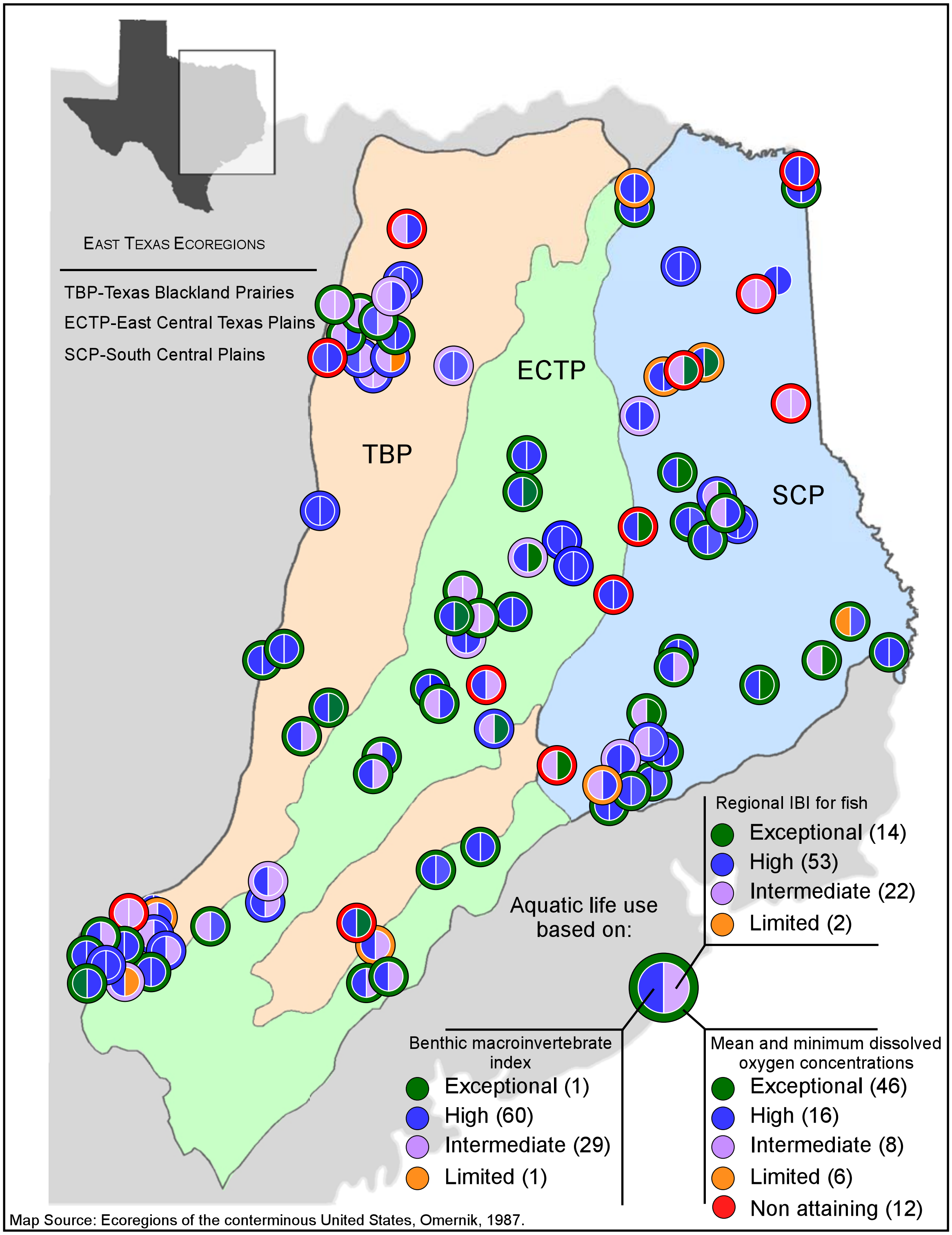

FIGURE 11.—Comparison of IBI, benthic invertebrate index, and combined dissolved oxygen concentration derived aquatic life uses (ALU) from all sites.

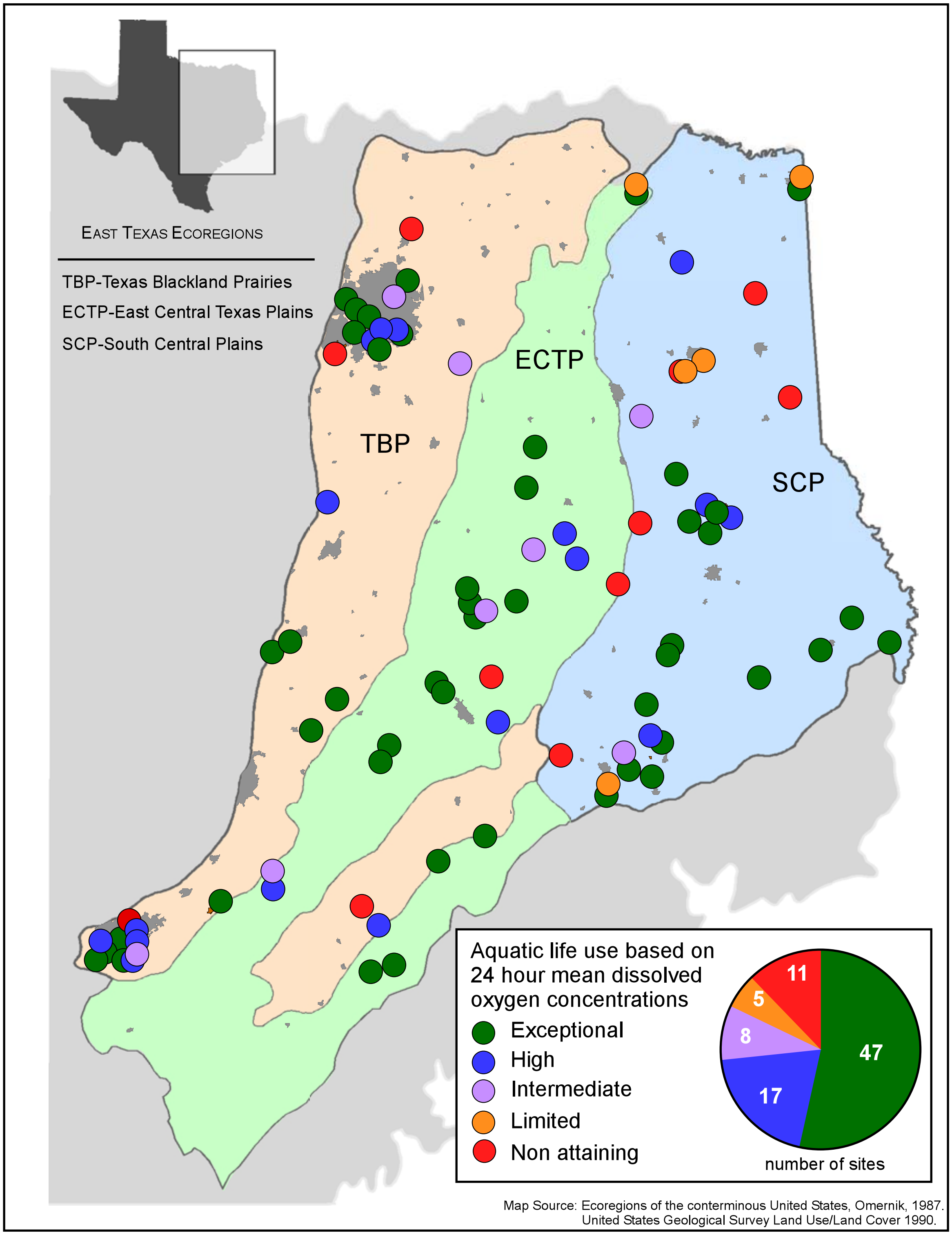

Figure 12.—Aquatic life use based on means from 24-hour dissolved oxygen concentrations.

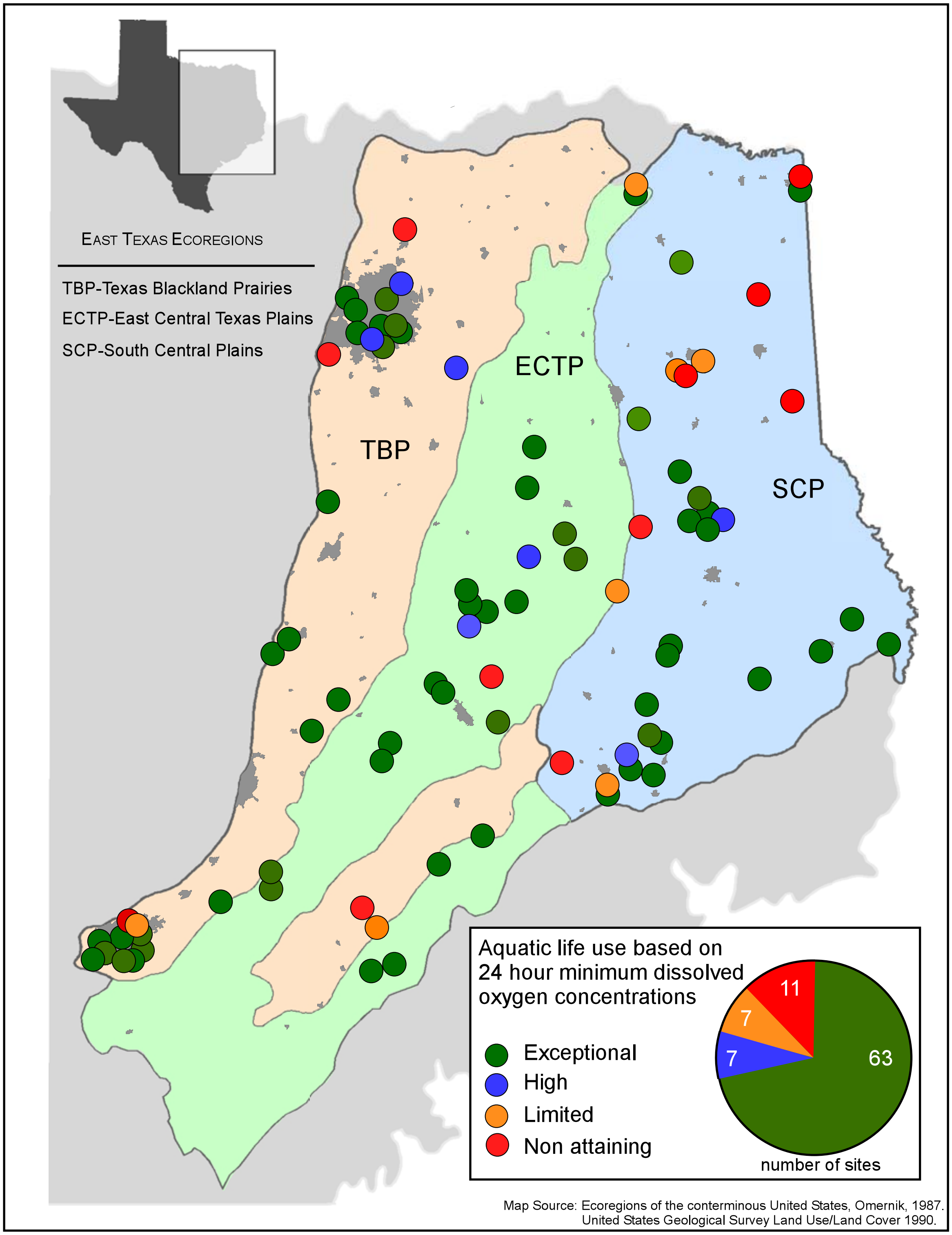

Figure 13.—Aquatic life use (ALU) based on absolute minima from 24-hour dissolved oxygen concentrations.

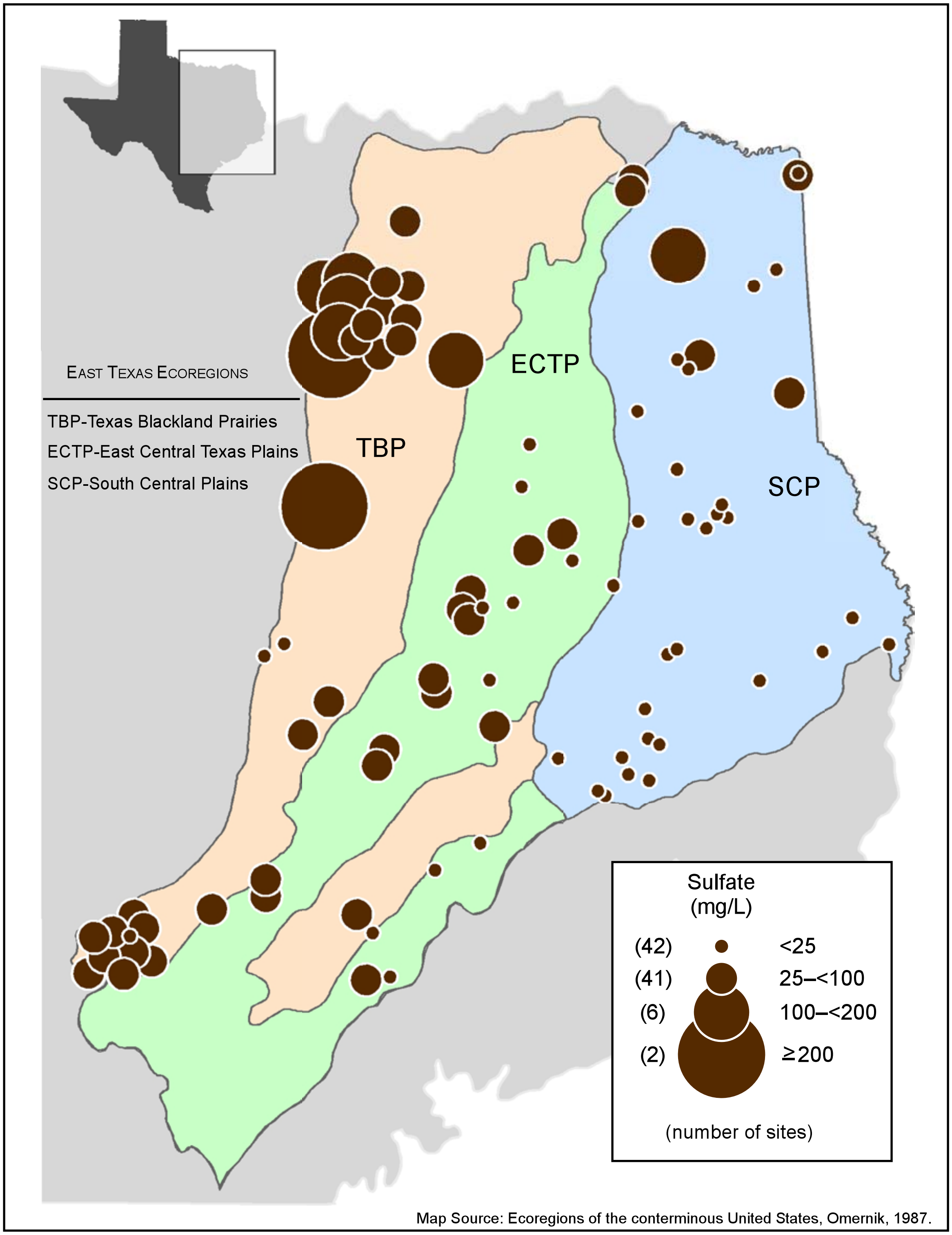

Figure 14.—Sulfate concentration from analysis of single water samples from each site.

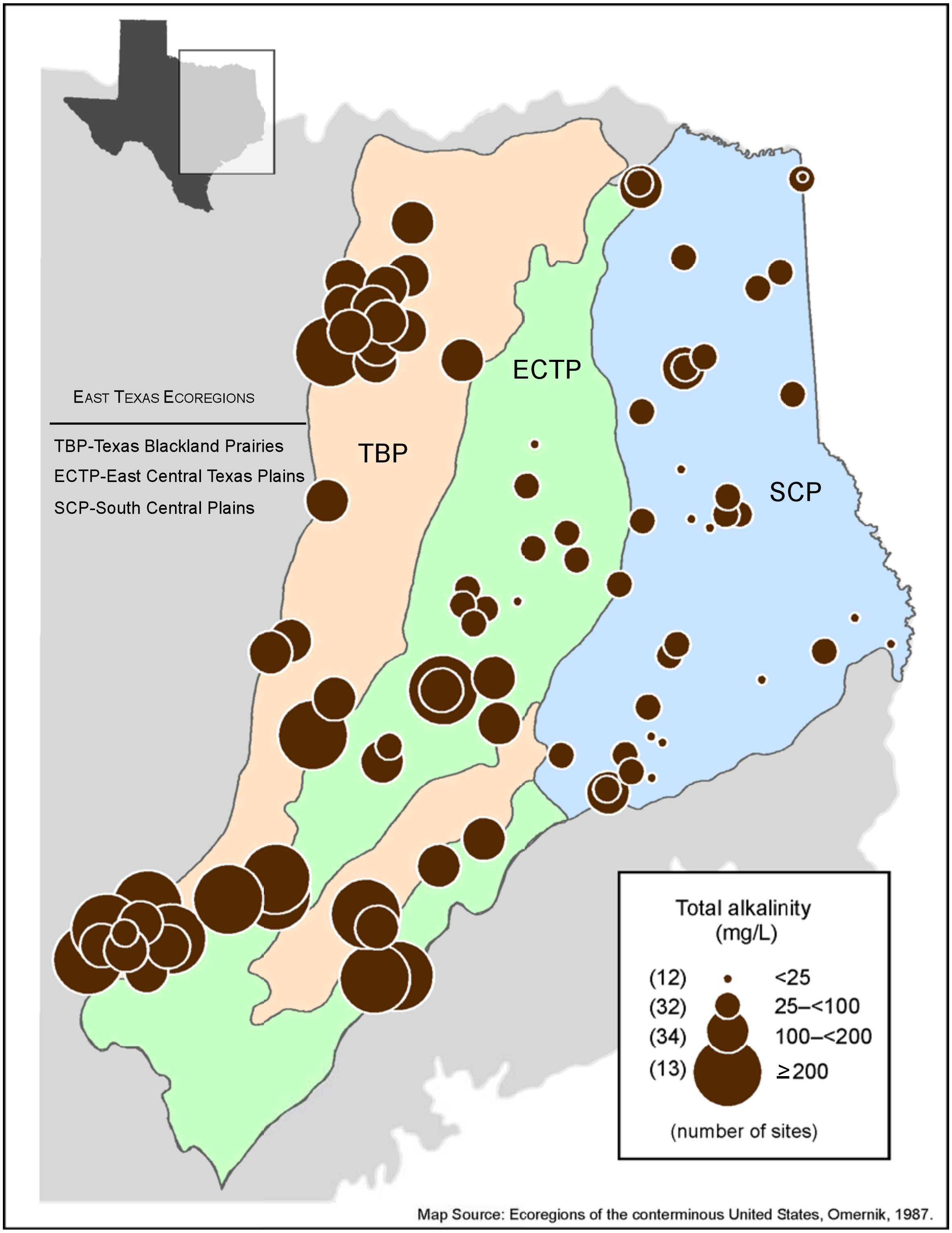

FIGURE 15.—Total alkalinity concentration from analysis of single water samples form each site.

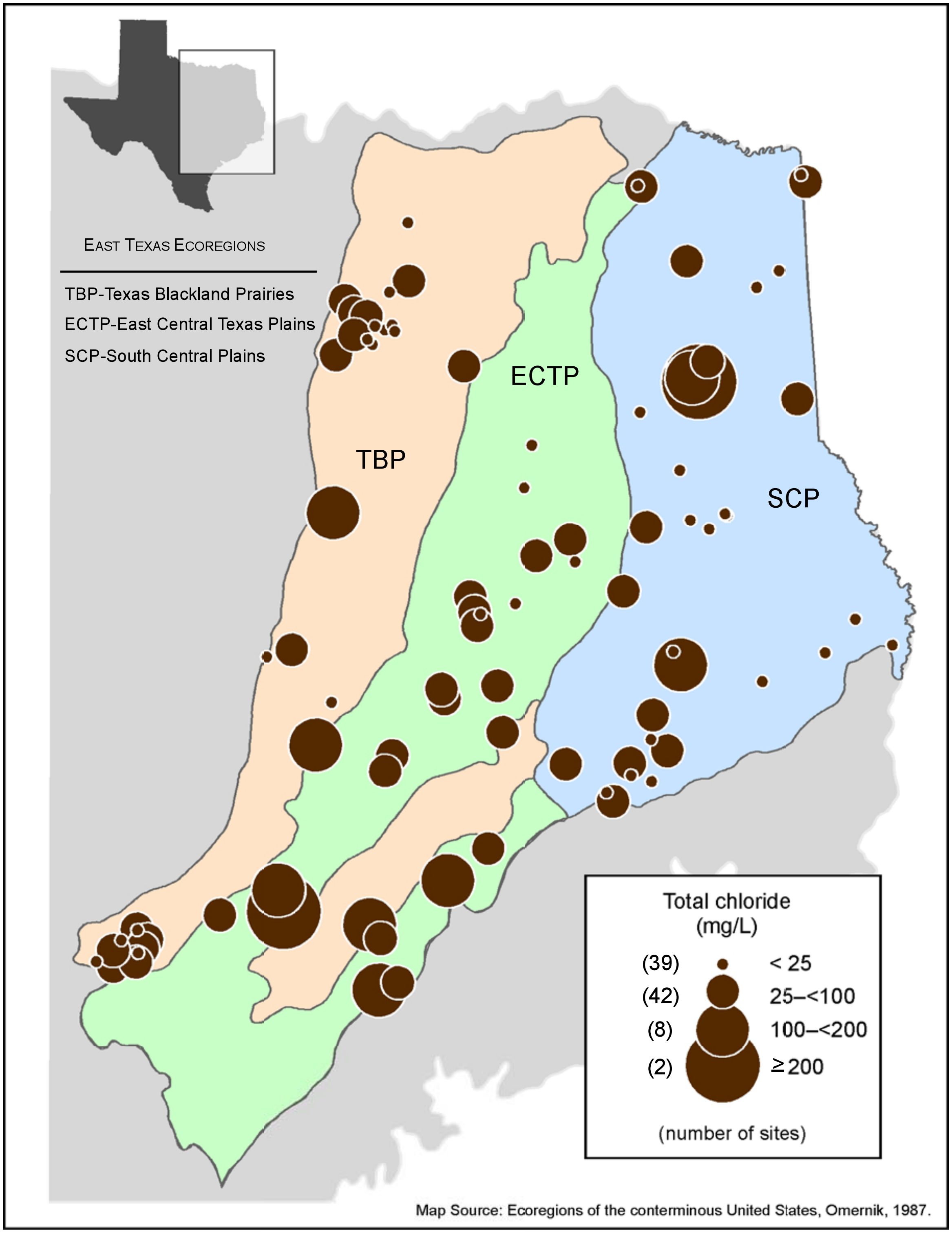

Figure 16.—Total chloride concentration from analysis of single water samples from each site.

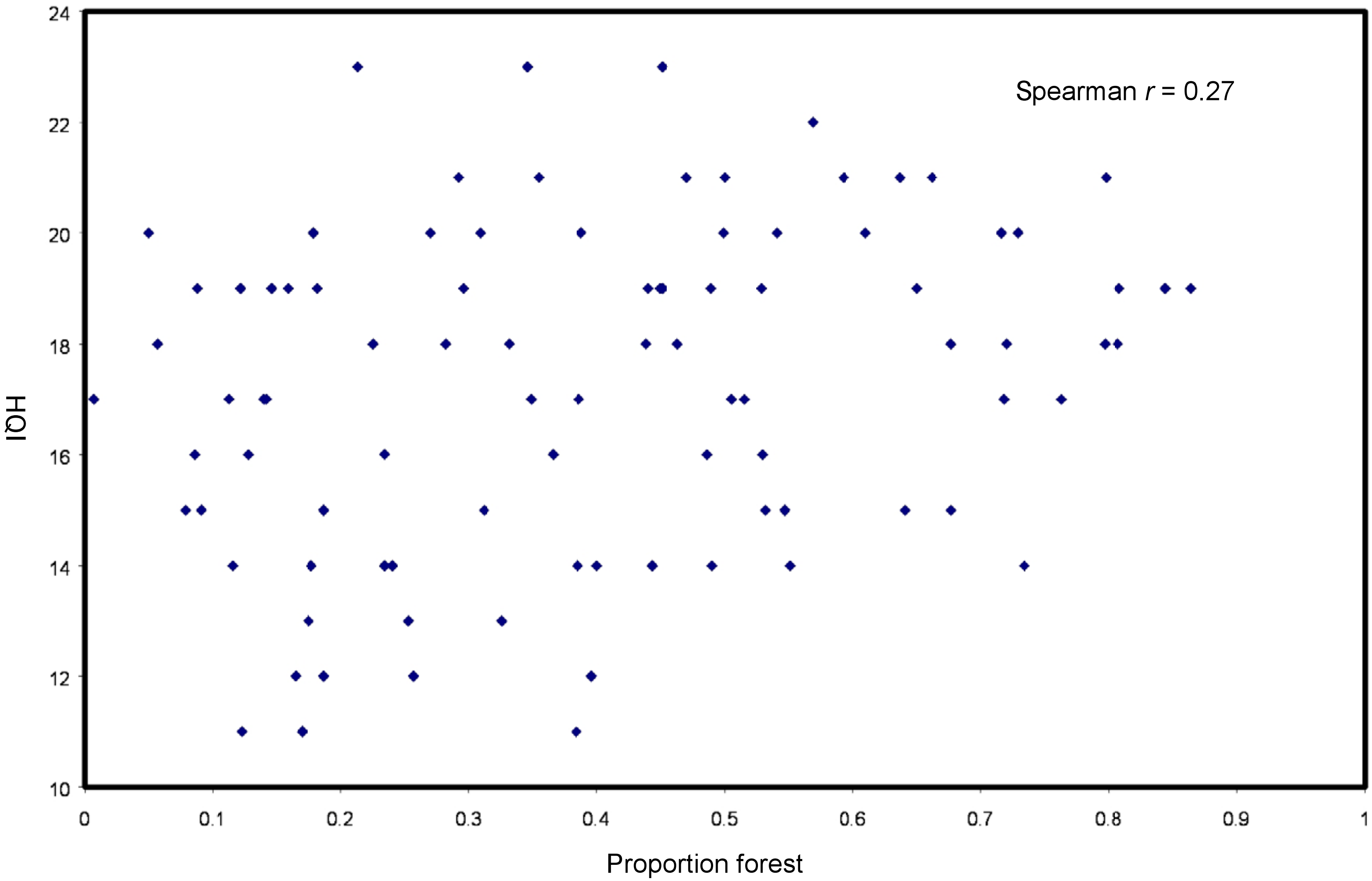

FIGURE 17.—HQI scores versus proportion of total forested area in the basin upstream of each site. The *r* value is from a Spearman correlation (P<0.05).

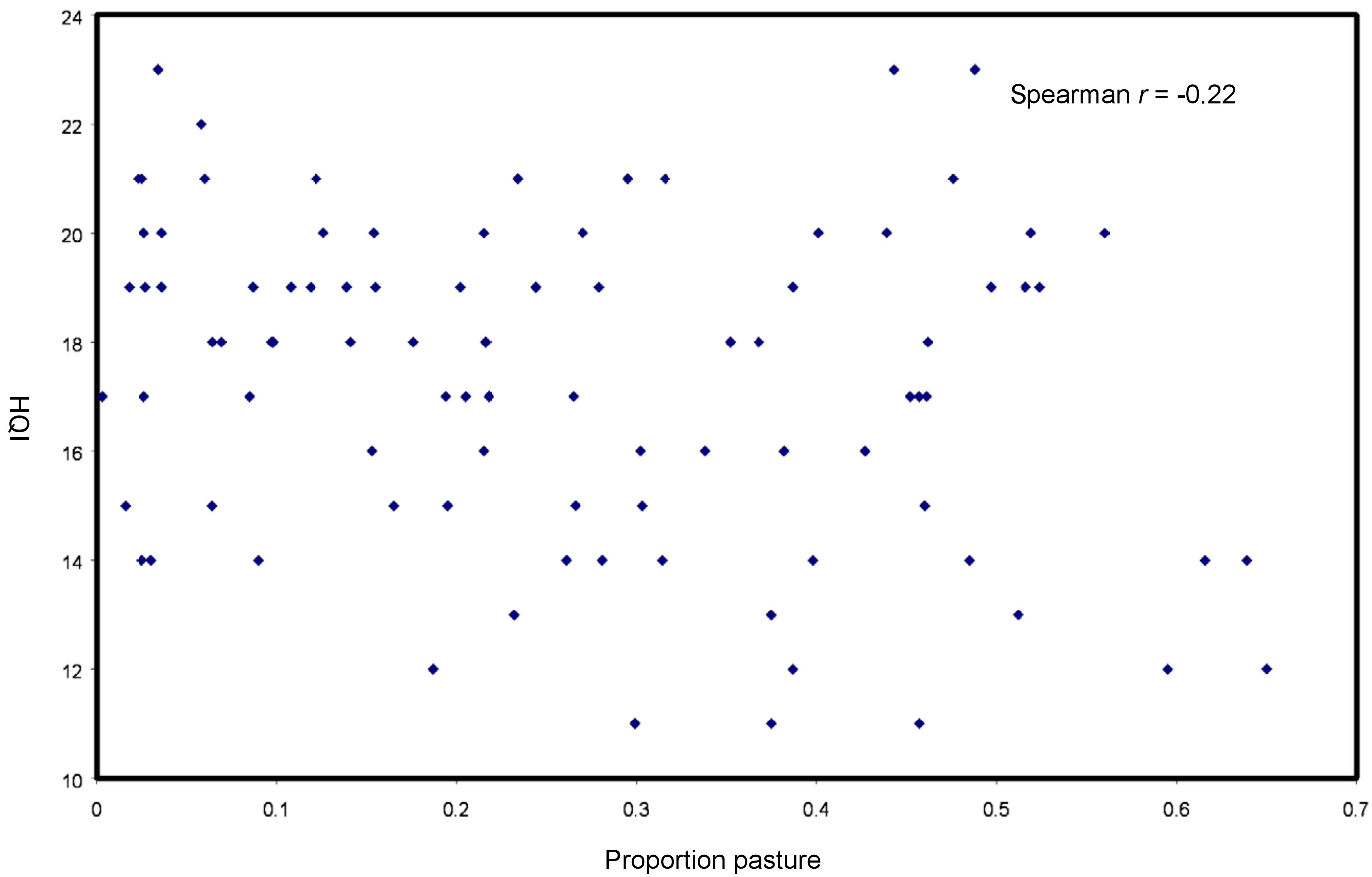

FIGURE 18.—HQI scores versus proportion of pasture in the basin upstream of each site. The *r* value is from a Spearman correlation (P<0.05).

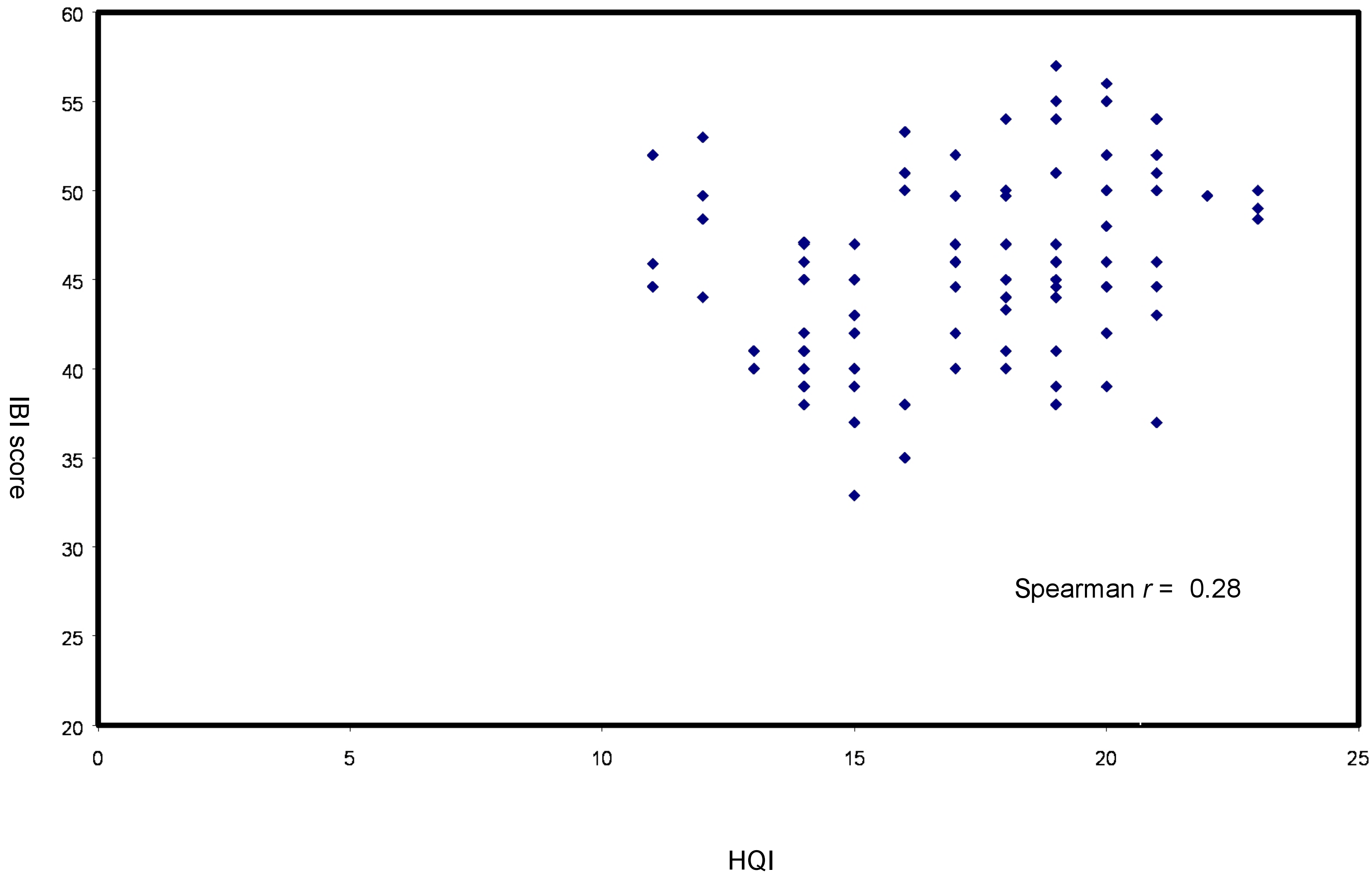

FIGURE 19.—Standardized IBI scores versus HQI scores at each site. The *r* value is from a Spearman correlation ($P<0.05$).

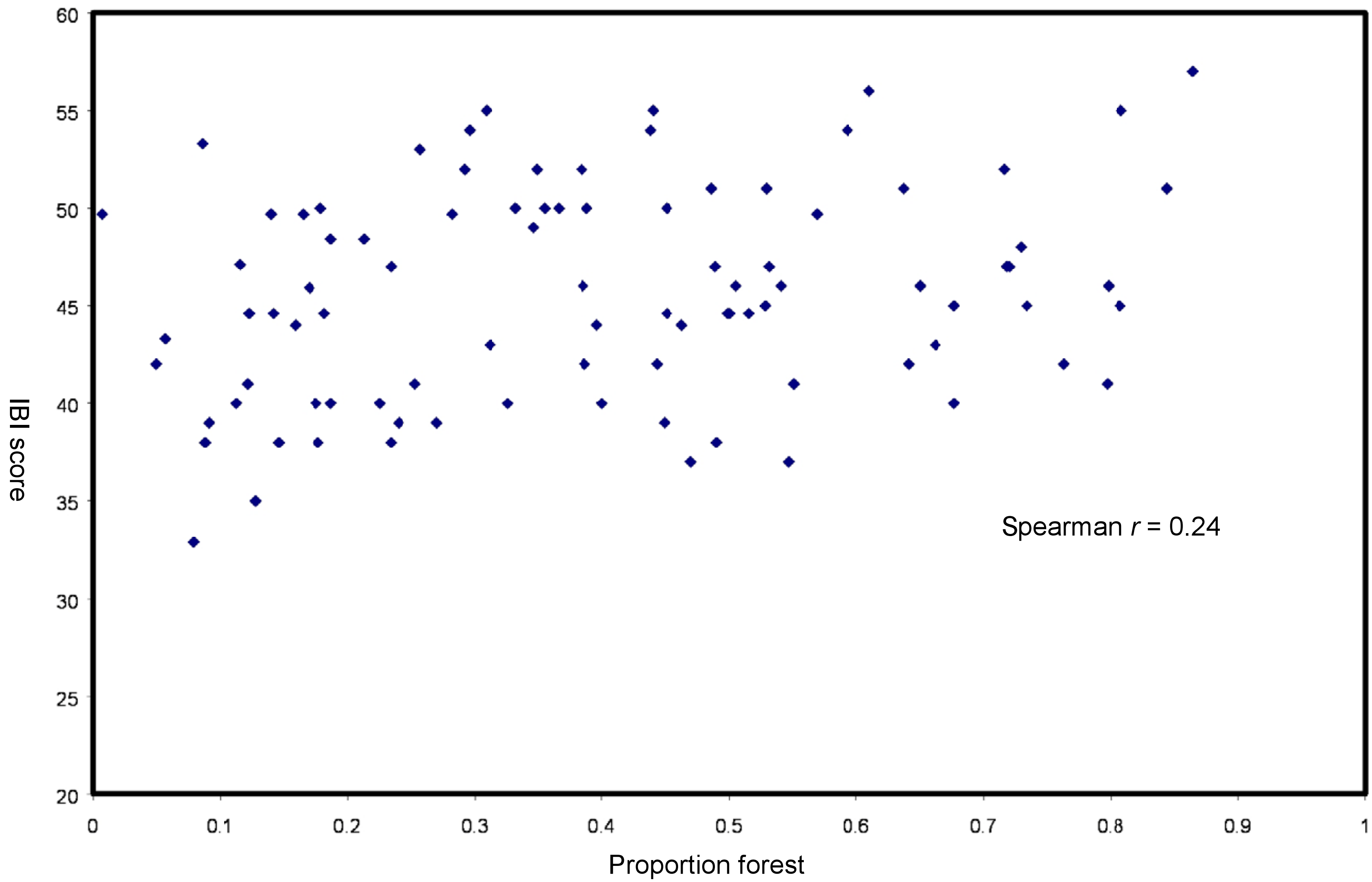

FIGURE 20.—Standardized IBI scores versus proportion of total forested area in the basin upstream of each site. The *r* value is from a Spearman correlation (*P*<0.05).

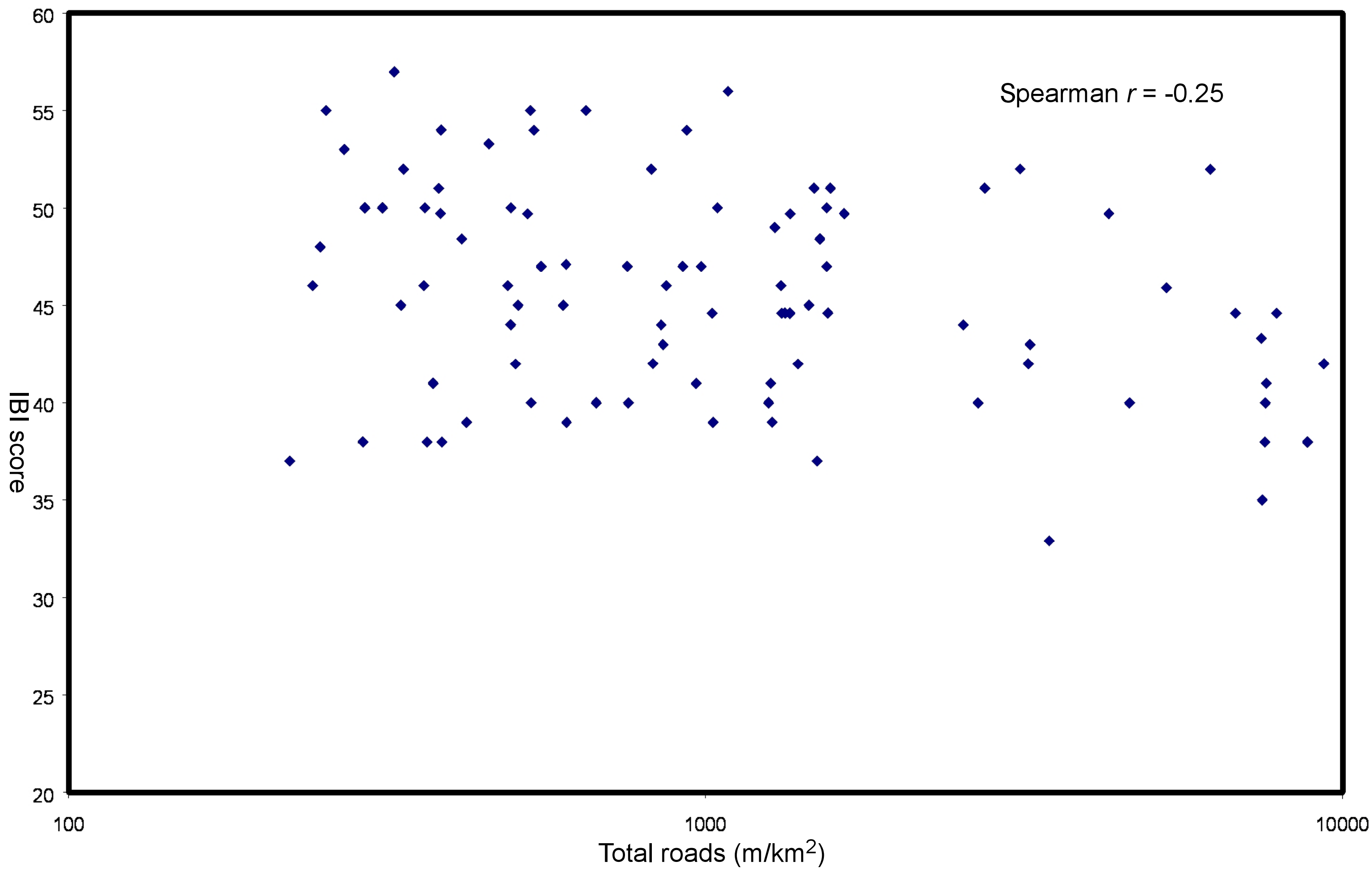

FIGURE 21.—Standardized IBI scores versus the density of all roads in the basin upstream of each site. The *r* value is from a Spearman correlation (P<0.05).

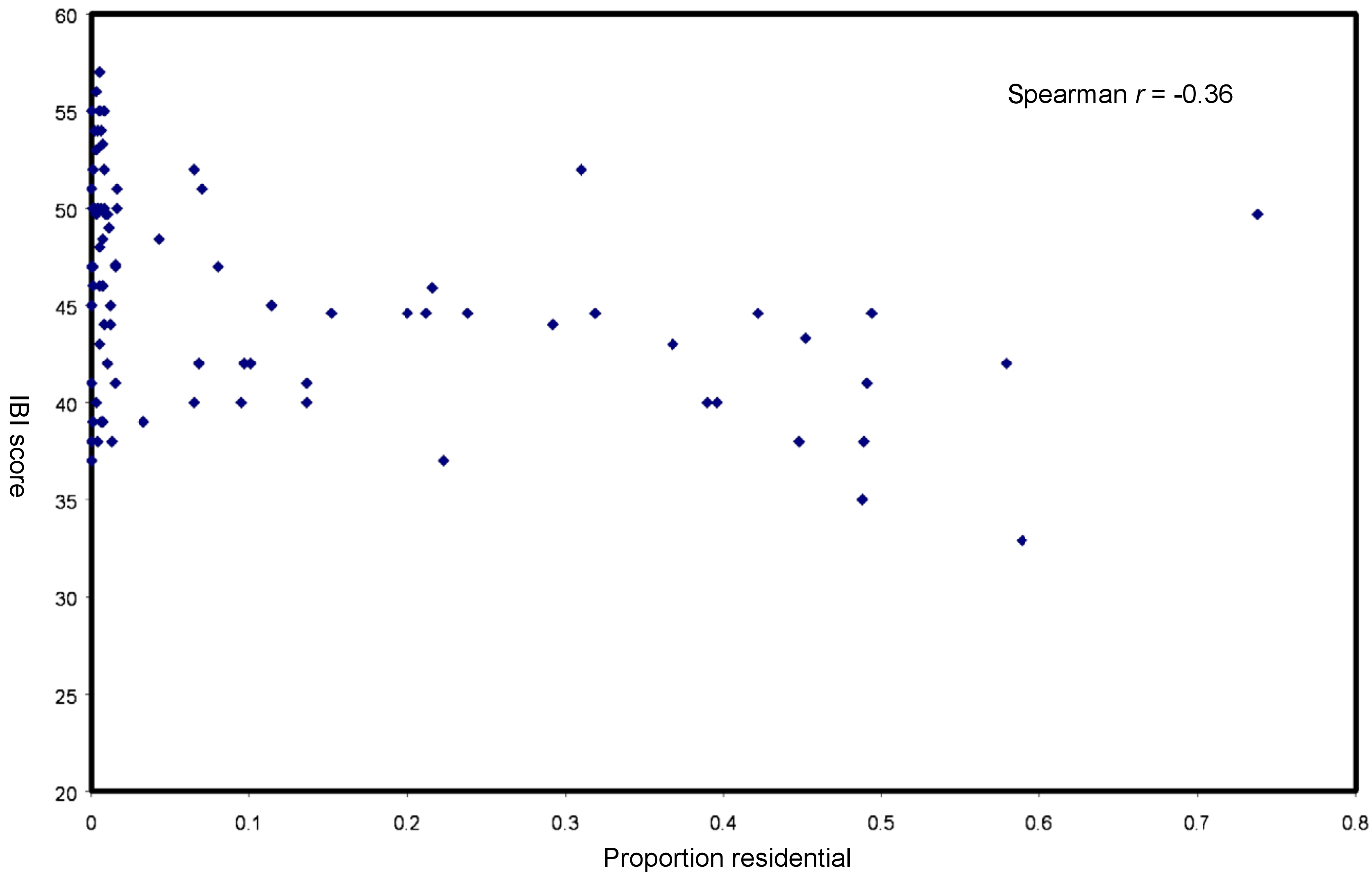

FIGURE 22.—Standardized IBI scores versus the proportion of total residential area in the basin upstream of each site. The *r* value is from a Spearman correlation (*P*<0.05).

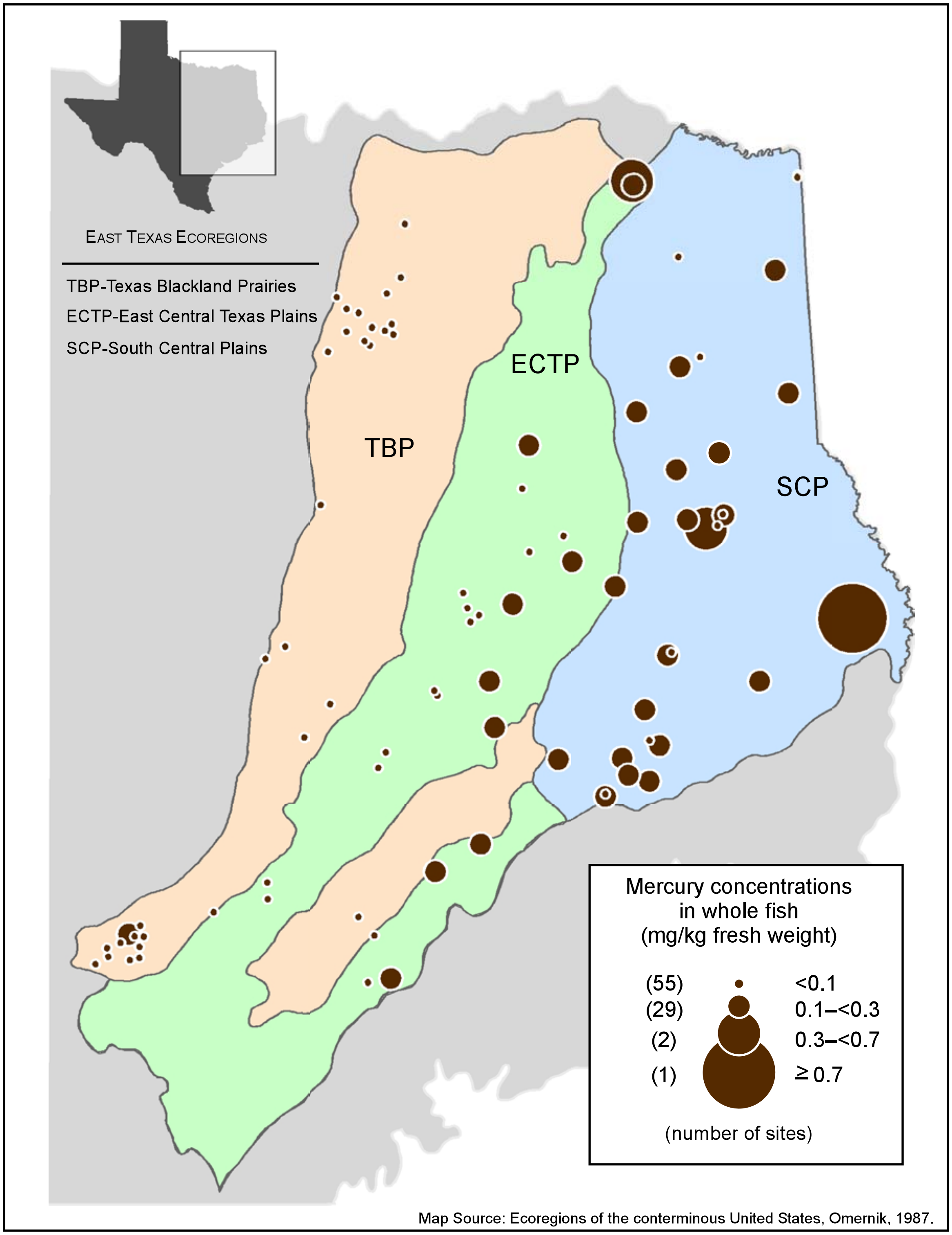

Figure 23.—Mercury concentration ranges in whole fish from each site.

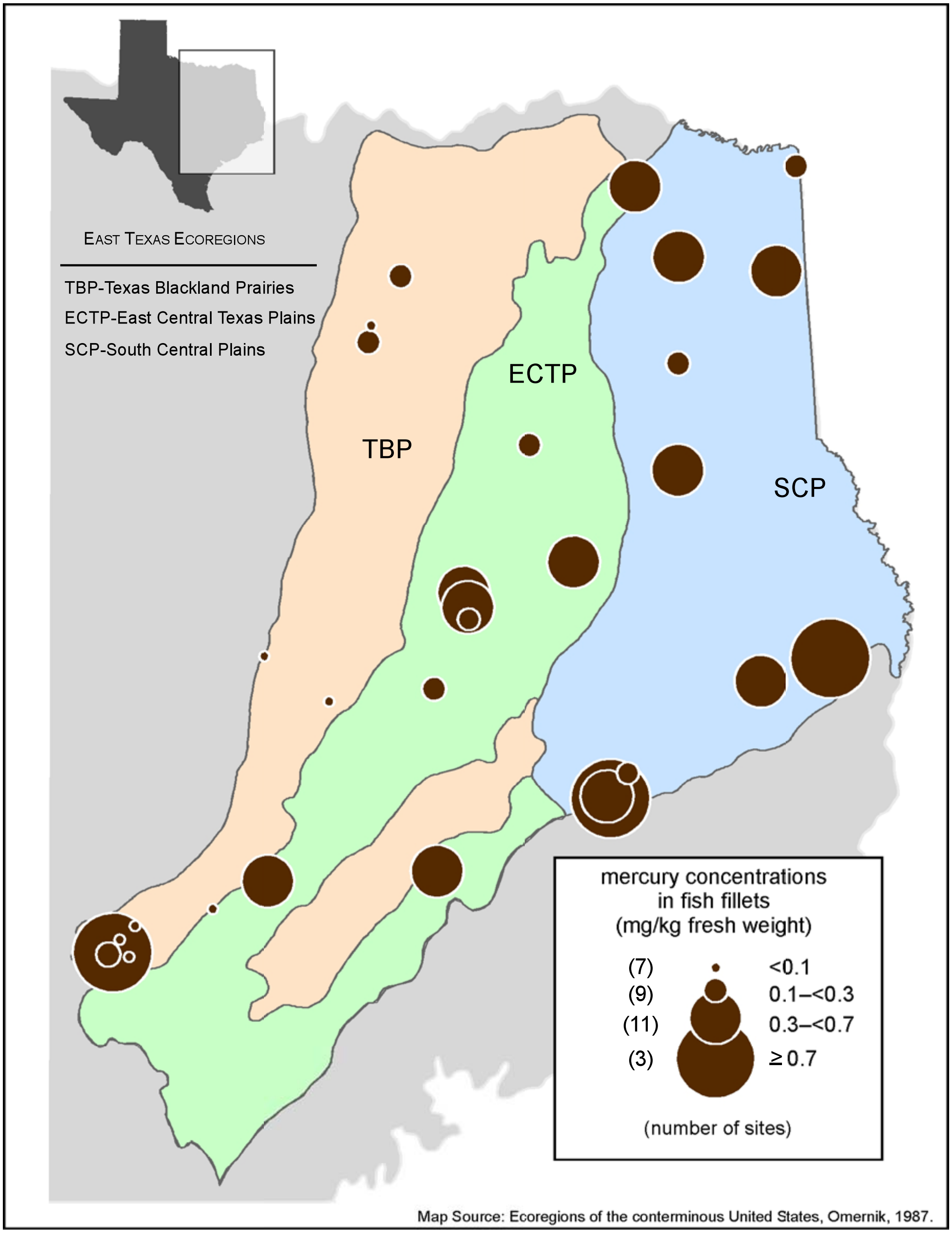

FIGURE 24.—Mercury concentration ranges in fish fillets from each site.