Factors Explaining the Distribution and Site Densities of the Neosho Madtom (*Noturus placidus*) in the Spring River, Missouri

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ABSTRACT

The Neosho madtom, a Federally-listed threatened species endemic to the Arkansas River system, is presently restricted to selected mainstem reaches of the Neosho, Cottonwood, and Spring rivers in Missouri, Kansas, and Oklahoma. These rivers are affected by anthropogenic factors such as municipal waste discharges and agricultural runoff. The Spring River also drains the Tri-State Mining District, where zinc-lead mining occurred in the past. Our objective was to assess effects of water quality degradation, due mainly to mining-related contaminants, on aquatic communities of the Spring River by comparison with those of the Neosho-Cottonwood system. We found higher densities of *N. placidus*, finer-textured riffle substrate, and lower concentrations of cadmium and lead in benthic macro-invertebrates in the Neosho-Cottonwood system than in the Spring River. In the Spring River, we found no substrate differences between sites with and without *N. placidus*; however, taxonomic richness of the benthic macro-invertebrate and fish communities were greater, densities of *N. placidus* and other fishes were higher, and concentrations of zinc and cadmium in benthic macro-invertebrates were lower at sites with *N. placidus*. Pore waters from three sites in the Spring River system were toxic to *Ceriodaphnia dubia*; mortality was greater than 80%, and there was no reproduction. Concentrations of zinc and cadmium in pore waters and sediment were high at these sites relative to non-toxic sites, and had SEM/AVS ratios considered potentially toxic. Toxicity tests, concentrations of metals in benthic macro-invertebrates, toxic unit (U_T) modeling of pore waters, and an empirical habitat model support a hypothesis of contaminant involvement in the distribution of *N. placidus*.

INTRODUCTION

The Federally-listed (threatened) Neosho madtom (*Noturus placidus*) is a small ictalurid (generally <75 mm total length) endemic to parts of the Arkansas River system. *N. placidus* is currently found in the mainstems of the Neosho, Cottonwood, and Spring rivers in Kansas, Missouri, and Oklahoma, where it inhabits reaches with slow to moderate flow, moderate depths, and unconsolidated pebble and gravel substrate (Moss 1983). The Spring River and its tributaries drain the Tri-State Mining District, where abandoned zinc (Zn)-lead (Pb) mines and the weathering of tailings has caused elevated concentrations of cadmium (Cd), Pb, and Zn in water, sediment, and biota; some tributaries (Short, Turkey, Center, Willow, and parts of Shoal Creeks) are heavily contaminated (Barks 1977; Schmitt et al. 1993; Wildhaber et al. in press a). *N. placidus* population densities are much greater in the Neosho-Cottonwood system than in the Spring River (Wilkinson et al. 1996; Wildhaber et al. in press b). The reaches inhabited by most Spring River Neosho madtoms are upstream of most current sources of mining-derived pollution in the watershed (Barks 1977; Wilkinson et al. 1996).

METHODS OF STUDY

Our overall objective was to assess the natural and anthropogenic factors that may be limiting populations of riffle-dwelling benthic fishes in the Spring River, especially the Neosho madtom. Our basic approach was to quantitatively characterize the riffle environment inhabited by and aquatic communities found with *N. placidus* in the Neosho-Cottonwood system, where no mining has occurred; and to use this information as a baseline against which to compare the mining-affected Spring River system.

We developed an empirical model based on 1991 physical habitat, water chemistry, and nutrient measurements from the Neosho-Cottonwood system to predict 1994 *N. placidus* densities in the Neosho, Cottonwood, and Spring rivers (Wildhaber et al. in press b). By comparing 1991 and 1994 data; 1994 predicted vs. observed densities; and the results of toxicity tests and other measurements, we assessed the extent to which basic environmental quality, metals contamination, or both limited *N. placidus* in the Spring River. Along with *N. placidus* densities, our measurements included: aquatic community (i.e., riffle-dwelling fishes and benthic macro-invertebrates); physical habitat; water chemistry (including nutrients); metals in surface waters, pore waters, sediments, and invertebrates; and porewater toxicity tests conducted with *Ceriodaphnia dubia*. Here we present an overview of our investigations; details of methods and results are presented elsewhere (Wilkinson et al. 1996; May et al. 1997; Schmitt et al. 1997; Allert et al. 1997; Wildhaber et al. 1997, in press a; in press b).

Field and Laboratory Procedures

Study sites were located on the Neosho and Cottonwood rivers, and on the Spring River and several of its tributaries. Measurements of aquatic community, water quality (including contaminants and nutrients), and habitat were made at 33 sites spanning both river systems during late summer and early fall, 1994. In 1995, 12 sites in the Spring River system were re-sampled, and porewater sampling and toxicity testing were conducted (Figure 1). Sediment and pore water from a reference site in another watershed (Tavern Creek, Site 13) were also evaluated.

At each fish collection site, 3-5 transects were established to sample all potential *N. placidus* habitat (i.e., the total length of gravel bars to a maximum water depth of 1.25 m). On each transect, 3-5 stations spaced equally across the stream were sampled for fish, benthic macro-invertebrates, and substrate. Fish collected by kick-seining an area of 3.0-m $\times$ 1.5-m using a 3-mm (square) mesh. Fishes were identified in the field and released. Benthic macro-invertebrates were collected with a Hess sampler (0.1-m^2 area, 0.3-mm mesh collection bag). Benthic macro-invertebrate samples were preserved and returned to the laboratory for sorting and identification to the lowest taxonomic level possible without mounting individual specimens (generally genus, except for the Chironomidae). Substrate was collected with a cylindrical grab sampler and sieved and weighed in the field; fines (<2 mm) were returned to the laboratory for further textural analysis. Current velocity and water depth were also measured at each station.

Dissolved oxygen, pH, conductivity, and temperature were measured and a surface grab sample for nutrient and elemental analyses was collected at the center transect of each site. Additional non-quantitative samples of benthic macro-invertebrates were collected for analyses of elemental contaminants. Sediment was collected from depositional areas for chemical analysis and porewater extraction. Most metals were analyzed by inductively coupled argon plasma (ICAP) emission spectroscopy; Pb and Cd in some samples were analyzed by atomic absorption spectroscopy (AAS).

Pore water was extracted under pressure (N_2) by the method of Carr and Chapman (1995). Porewater samples were serially diluted and tested for toxicity with *Ceriodaphnia dubia*; tests ran for 7 d (U.S. EPA 1989). Aliquots of the composited sediments were collected for elemental analyses by ICAP and for acid volatile sulfide (AVS) and simultaneously extracted metals (SEM). Pore water was also analyzed by ICAP-ES, AAS (Pb and Cd), and ICAP-mass spectrometry.

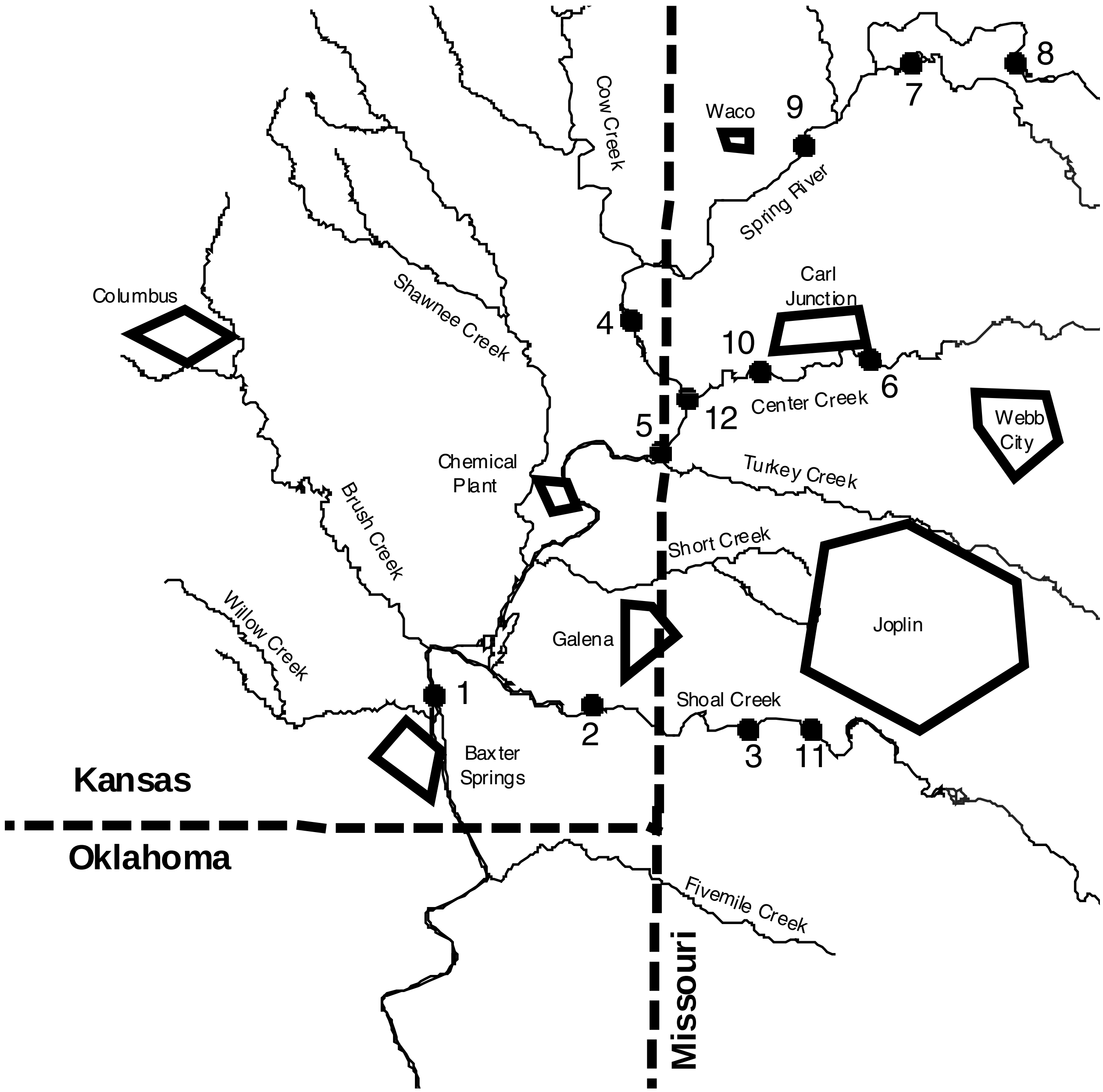

Figure 1. Spring River sites sampled in 1995.

Toxic Units and AVS Modeling

Relative ecological risks from sediment metals were evaluated using the toxic unit model of Wildhaber and Schmitt (1996). A toxic unit (U_T) is defined as the ratio of the estimated concentration of a contaminant in the pore water of a test sediment (C_{wp}) to the estimated chronic aquatic toxicity of that contaminant (C_{wps}):

$$U_T = C_{wp} / C_{wps}$$

The U_T for all measured contaminants (here, just metals) are summed to obtain a total toxicity estimate for that sediment.

The AVS model is based on the assumption that under the reducing conditions present in sediment pore waters, sulfides control the concentrations and hence bioavailability of divalent metals. Because the sulfide salts of metals are extremely insoluble; their formation renders the metals biologically unavailable. AVS modeling of porewater concentrations therefore adjusts the maximum potential porewater concentrations of metals downward based on the amount of AVS present (DiToro et al. 1990). We allocated AVS to metals in weak acid extracts based on the solubility product constants, K_{sp}, of their sulfide salts (Weast et al. 1988). Accordingly, AVS was allocated to metals in the following order: Copper (Cu), Cd, Pb, Zn, nickel (Ni), and iron (Fe); i.e., FeS is the most soluble sulfide and CuS the least. An equimolar amount of sulfide was allotted to each metal, in the order of their K_{sp}, until either all AVS was accounted for or all SEM were considered sulfide-bound.

The following relative potential toxicity values (i.e., one toxic unit) were obtained or computed from chronic toxicity information for each element, as follows: Cu (5.6 •g/L, U.S. EPA 1980), Cd (computed, U.S. EPA 1984a), Pb (computed, U.S. EPA 1984b), Zn (computed, U.S. EPA 1987), Ni (computed, U.S. EPA 1986), and Fe (1000 •g/L, U.S. EPA 1976). Because the computed values are hardness-dependent, we used hardness values typical of the Neosho-Cottonwood and Spring River systems (i.e., 150 mg/L as $CaCO_3$) in our computations.

Statistical Analysis

We analyzed the data at the level of site averages to assess differences between the Neosho-Cottonwood and Spring River systems, and among Spring River sites with and without Neosho madtoms. We calculated site densities of *N. placidus* and, as a group, the riffle-dwelling fishes that could be considered benthic competitors of *N. placidus*, by dividing the total number of Neosho madtoms or benthic fish competitors collected at a site by the total area sampled with the kick seine. Determination of benthic fish competitors was based on known habitat preferences and feeding habitats of each species as described by Pflieger (1975). We also computed fish species rarefaction (Hurlbert 1971) for each site. Statistical methods included analysis of variance, correlation analysis, multivariate analysis of variance, principal componants analysis, and discriminant analysis.

RESULTS

Physical and Chemical Variables

Substrate, water quality, and contaminant concentrations differed between the Neosho-Cottonwood system and the Spring River and between Spring River sites with and without Neosho madtoms (Allert et al. 1997; Schmitt et al. 1997; Wildhaber in press b). Mean substrate particle diameter was smaller in the Neosho-Cottonwood system than in the Spring River (Figure 2). Among the mining-derived metals of concern in the Spring River system, only Zn occurred at ICAP-detectable concentrations in surface waters and Zn concentrations was generally higher in the tributaries than in the mainstem. Detectable levels of Zn, Cd, Pb, Cu, Fe, and Ni were found in pore waters at all sites in 1995 (Table 1). Concentrations of all elements were higher in the tributaries except for Ni, which was highest at Sites 1 and 4.

Concentrations of mining-derived elements in sediments paralleled those found in surface and pore waters. Concentrations of Pb (2120 •g/L), Zn (13800 •g/L), Cd (84.1 •g/L), Cu (51.2 •g/L), and Al (16000 •g/L) were 3- to 10-fold higher at Site 12 than any other site by ICAP-ES. Sediments collected at sites above Center Creek had the lowest concentrations of Cd (0 - 2.88 •g/L), Cu (5.74 - 16.7 •g/L), Pb (6.77 - 34.4 •g/L), and Zn (103 - 615 •g/L). Concentrations of Ni in sediments from Sites 1, 4, 5, 7, and 12 were similar (21.8 - 29.1 •g/L). AVS and SEM results for sediments were consistent with ICAP results (Table 2). Total toxic units (mostly attributable to Pb and Zn) were greatest at Sites 5 and 12 (Table 3), at the mouths of Turkey and Center creeks, respectively (Figure 1).

Concentrations of Cd, Mn, Ni, and Pb were higher in benthic macro-invertebrates from the Spring River than the Neosho-Cottonwood system; however, Mg and Sr concentrations were higher in the Neosho-Cottonwood system (Figure 3). Concentrations of Ba in benthic macro-invertebrates were significantly lower and concentrations of Cd and Zn were significantly higher at sites without *N. placidus* than at sites with.

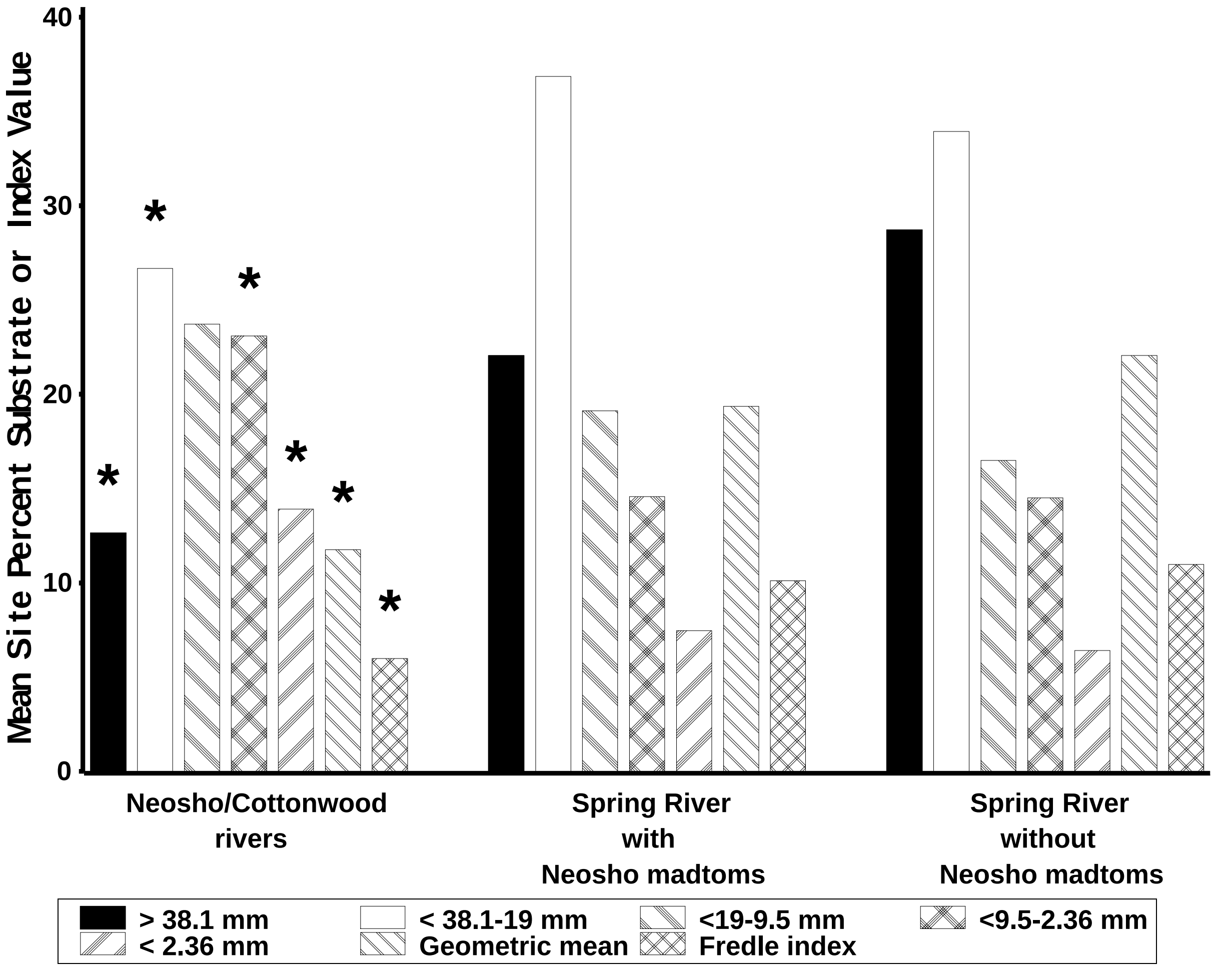

Figure 2. Mean proportion by weight of substrate in each of five size categories, and geometric mean, and fredle index (McMahon et al. 1996) for combined size categories. None of the substrate measures showed a significant (P > 0.05) difference between Spring River sites with and without Neosho madtoms based on analysis of variance (ANOVA). '*' represents significant ANOVA differences in substrate measures between the Neosho-Cottonwood system and the Spring River.

Biological Variables

Neosho madtoms were found at many Neosho-Cottonwood sites and at 8 of the 10 uppermost Spring River sites sampled in 1994 (Wildhaber et al. in press a, in press b). No Neosho madtoms were found in any Spring River tributaries. Results of 1995 fish sampling in the Spring River were similar--Neosho madtoms were present at the 4 mainstem sites sampled (all four had yielded Neosho madtoms in 1994), and none were found in tributaries (Allert et al. 1997). In benthic macro-invertebrate samples, the uppermost Spring River sites also supported the greatest numbers of Ephemeroptera, Plecoptera, and Trichoptera (the EPT taxa) (Wildhaber et al. in press b). The lowest numbers of EPT taxa were found at contaminated sites on Spring River tributaries, the benthic macro-invertebrate faunas of which were dominated by chironomids and oligochaetes. The reduced representation of EPT taxa and the dominance of Chironomidae and Oligochaeta at these sites suggest that water or habitat quality is degraded.

There were both similarities and differences in fish species densities and fish community composition between the Neosho-Cottonwood and Spring River systems. Fish rarefaction was significantly lower in the Neosho-Cottonwood system than in the Spring River system, and

Table 1. Dissolved concentrations (μg/L) of 8 elements in porewaters as measured by ICAP-MS ([a]) and Zeeman atomic absorption ([b]) Porewaters were collected from depositional areas at each site. TC represents Tavern Creek.

Site	Ca	Cd[a]	Cd[b]	Cu[a]	Cu[b]	Fe	Mg	Ni	Pb[a]	Pb[b]	Zn
1	171500	3.2	2.9	1.8	2.1	7760	24500	15.0	2.0	1.8	467.0
2	44500	1.4	1.2	2.5	2.6	75.0	3370	5.6	0.69	0.70	87.0
3	43700	1.0	0.97	13.0	9.5	15.0	3290	7.2	1.3	1.2	77.0
4	60300	0.44	0.38	1.6	1.9	38.0	8250	9.0	0.48	0.61	19.0
5	57500	0.70	0.71	1.6	1.3	32.0	7650	6.3	0.90	1.3	28.0
6	43100	0.16	0.24	0.8	0.62	2240	2660	4.9	0.75	0.69	197.0
7	43000	0.39	0.33	1.2	1.6	77.0	3050	2.4	0.62	1.0	20.0
8	104700	0.35	0.27	2.6	2.2	14600	7840	6.8	0.70	0.58	19.0
9	39000	0.32	0.23	1.2	1.3	73.0	3070	3.0	0.51	0.43	8.1
10	49500	0.13	0.12	0.88	0.77	49.0	3120	3.6	0.75	0.90	68.0
11	40400	0.27	0.24	1.2	1.1	22.0	2850	2.2	0.9	1.0	36.0
12	46600	2.6	2.3	2.0	2.0	1170	3010	5.7	3.1	2.4	1681
TC	34200	0.68	0.61	1.6	1.4	2780	24200	2.7	0.98	1.0	18.0

Table 2. Percent moisture, loss on ignition (LOI), concentrations of acid volatile sulfide (AVS) and simultaneoulsy extracted metals (SEM) in sediment. AVS are μmol/g dry weight and SEM concentrations are expressed in μg/g dry weight.

Site	% Moisture	LOI	AVS	Cd	Cu	Fe	Ni	Pb	Zn
1	28.06	1.5	2.10	3.60	2.30	1490	9.0	78.0	573
2	30.72	1.2	0.85	6.20	5.10	2640	4.9	94.0	756
3	36.05	0.8	0.06	2.50	1.40	797	4.2	71.0	777
4	21.89	2.3	0.16	0.39	1.40	1570	4.4	5.6	75.0
5	19.75	1.2	0.53	2.00	1.60	2430	4.8	42.0	355
6	41.04	1.5	6.00	21.0	13.0	3170	6.8	821	3570
7	26.94	1.8	0.50	0.57	1.80	1960	2.4	10.0	60.0
8	24.97	0.7	0.68	0.65	0.75	2940	1.1	4.4	46.0
9	23.48	1.2	0.31	0.18	0.79	1050	1.3	3.3	22.0
10	26.57	0.9	4.80	4.60	1.80	2130	3.0	127	997
11	23.34	0.9	0.04	1.40	1.10	777	4.1	44.0	270
12	40.13	3.2	1.20	108	45.0	13300	20.4	3410	14500
Tavern Creek	21.26	0.8	1.00	0.03	1.10	684	1.0	2.7	4.8

Table 3. Toxic units (U_T).attributable to the indicated element and total toxic units (Σ U_T) at each site in 1995 (see Figure 1).

Site	Cd	Pb	Zn	Ni	Fe	Σ U
1	<0.1	<0.1	7.20	0.94	3.7	11.0
2	<0.1	<0.1	11.3	0.05	6.0	17.3
3	<0.1	23.7	9.50	0.03	1.4	34.7
4	<0.1	<0.1	1.40	0.06	5.6	7.02
5	<0.1	<0.1	104	0.09	9.9	114
6	<0.1	<0.1	37.4	0.05	4.6	42.0
7	<0.1	<0.1	0.53	0.03	5.3	5.90
8	<0.1	<0.1	0.08	0.01	8.8	8.90
9	<0.1	<0.1	0.25	0.02	3.4	3.70
10	<0.1	<0.1	12.0	0.03	5.9	17.9
11	<0.1	25.2	5.90	0.06	2.5	33.7
12	43.9	773	584	5.50	19.8	1426
Tavern Creek	<0.1	<0.1	<0.1	<0.1	2.4	2.40

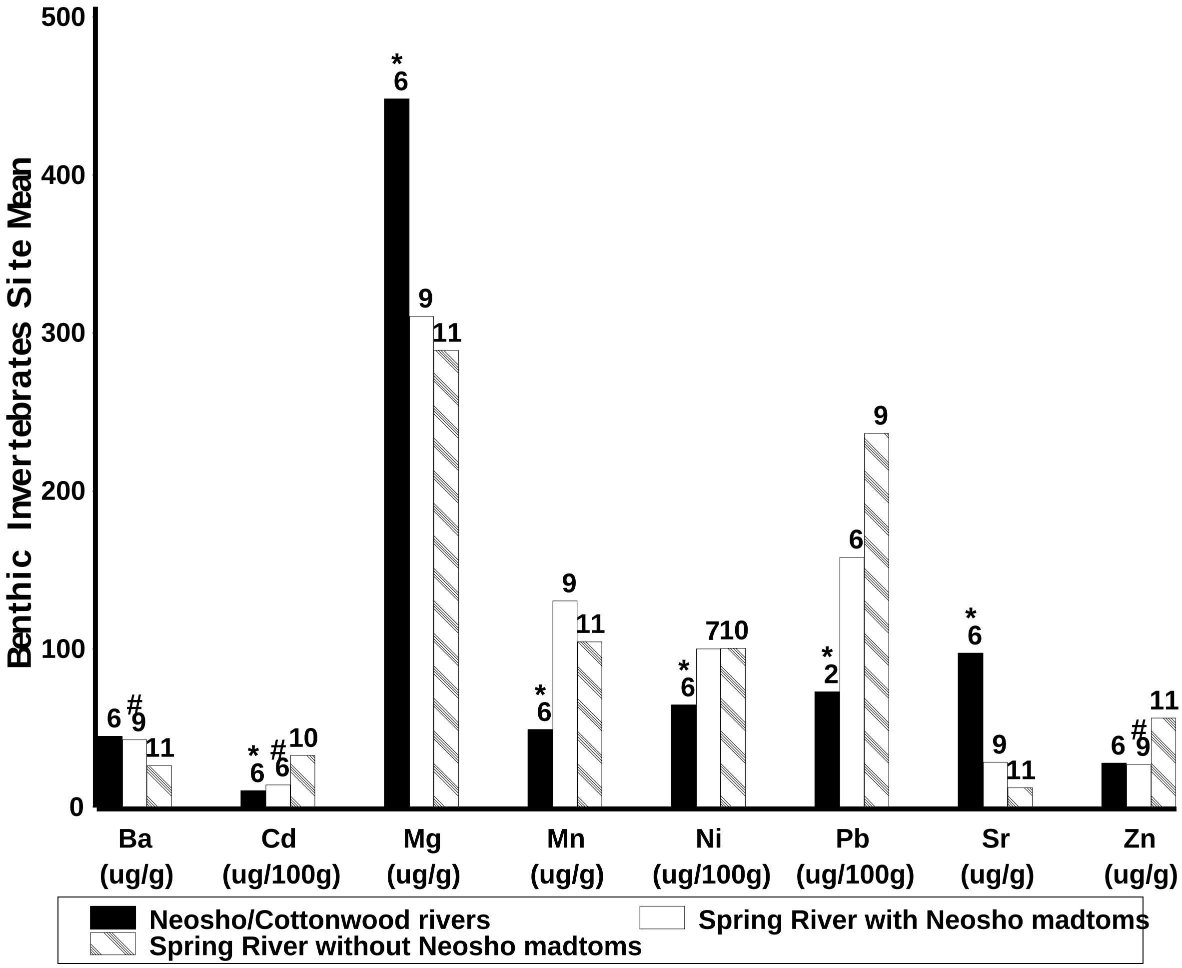

Figure 3. Mean benthic invertebrate tissue elemental concentrations that showed a significant difference (P < 0.05 analysis of variance--ANOVA) between the Neosho-Cottonwood system and the Spring River and/or Spring River sites with and without Neosho madtoms. Numbers over bars represent number of sites where the element was found at detectable levels (number of values used to calculate the bar value). Significant differences between Spring River sites with and without Neosho madtoms are represented by '#'. Significant differences between the Neosho-Cottonwood system and the Spring River are represented by '*'.

richness indices for fish species, macroinvertebrates, and the EPT index were significantly greater at Spring River sites with *N. placidus* than at sites without (Wildhaber et al. in press a, b). Pore water from three contaminated Spring River sites (Sites 1, 6, and 12) were toxic to *C. dubia*; survival of adults was >80% at all other sites (Table 4).

Relationships Among Variables

Principal component analysis (PCA) using all variables from both watersheds resulted in the selection of percentages of substrate fines, turbidity, alkalinity, hardness, NH_3, SO_4, Ba, Cu, Mn, Ti, NO_2/NO_3, porewater oxidation-reduction potential, and concentrations of Mg, Sr, Cd, Mn, and invertebrate Ni, which collectively accounted for 59% of the variability in data separating sites in the Neosho-Cottonwood system from those in the Spring River system (Figure 4). Restricting PCA to Spring River sites resulted in the selection of porewater alkalinity, NH_3, turbidity; EPT, and invertebrate Ba and Zn concentrations, which accounted for 81% of the variability in the data

Table 4. Summary of acute toxicity tests (*Ceriodaphnia dubia*) using site waters. Reproduction was calculated according to EPA guidelines. Site 1 - Site 7 and Tavern Creek (TC) were included in Test 1. Site 8 - Site 12 were included in Test 2. W1 and W2 represent well water for Tests 1 and 2, respectively. [a] = 50% dilution water.

Site	Number of replicates	% Survival	Average reproduction
1[a]	10	0	0
2[a]	10	90	29.0
3	10	90	25.6
4	10	100	28.4
5	10	90	22.9
6	9	22.2	0
7	9	100	27.1
8	10	90	19.5
9	10	80	20.4
10	9	100	28.1
11	10	100	25.0
12	10	0	0
TC	10	100	23.9
W1	10	100	24.6
W2	10	100	24.6

separating sites with and without *N. placidus* (Figure 4).

In the Neosho-Cottonwood system and in the Spring River above Center Creek, *N. placidus* densities predicted by the empirical model (see Wildhaber et al. in press b) on the basis of physical habitat attributes, water quality, and elemental analyses did not differ significantly from observed densities. At Spring River sites below Center Creek, however, observed densities were significantly lower than predicted values (Wildhaber et al. in press b; Figure 5).

CONCLUSIONS

Concentrations of mining-derived metals (i.e., Pb, Cd, Zn) were elevated in sediments and pore waters at sites on Shoal and Center Creeks, and at Spring River sites below the confluences of these tributaries. Toxic unit values indicated that some metals were bioavailable in Center and Shoal Creeks, and at sites in the Spring River below Center Creek. Toxicity test results indicated that populations of *N. placidus* could be affected in Center Creek and in the Spring River below Center Creek. Results obtained in 1995 (Allert et al. 1997) corroborated those of the 1994 studies (Schmitt et al. 1997; Wildhaber et al. 1997), which found lower than expected densities of *N. placidus* in the Spring River where contaminants (i.e., Pb, Cd, Zn) were elevated in sediments, pore waters, and benthic invertebrates.

Collectively, our findings of higher *N. placidus* densities, smaller substrate, and lower concentrations of Cd and Pb in benthic invertebrates in the Neosho-Cottonwood system than in the Spring River system suggest that differences in Neosho madtom densities are due both to differences in habitat and to contaminants. In the Spring River, both anthropogenic and natural factors may limit populations of *N. placidus*. Where levels of mining-derived contamination are low, *N. placidus* densities seem to be limited primarily by physical habitat and water quality. Where significant contamination has occurred, metals may limit *N. placidus* either directly (i.e., through food or waterborne toxicity) or indirectly (by limiting the benthic invertebrate food base). Our studies demonstrate that an integrated approach, which includes the assessment of natural and anthropogenic factors, is necessary to determine the factors that may limit fish populations.

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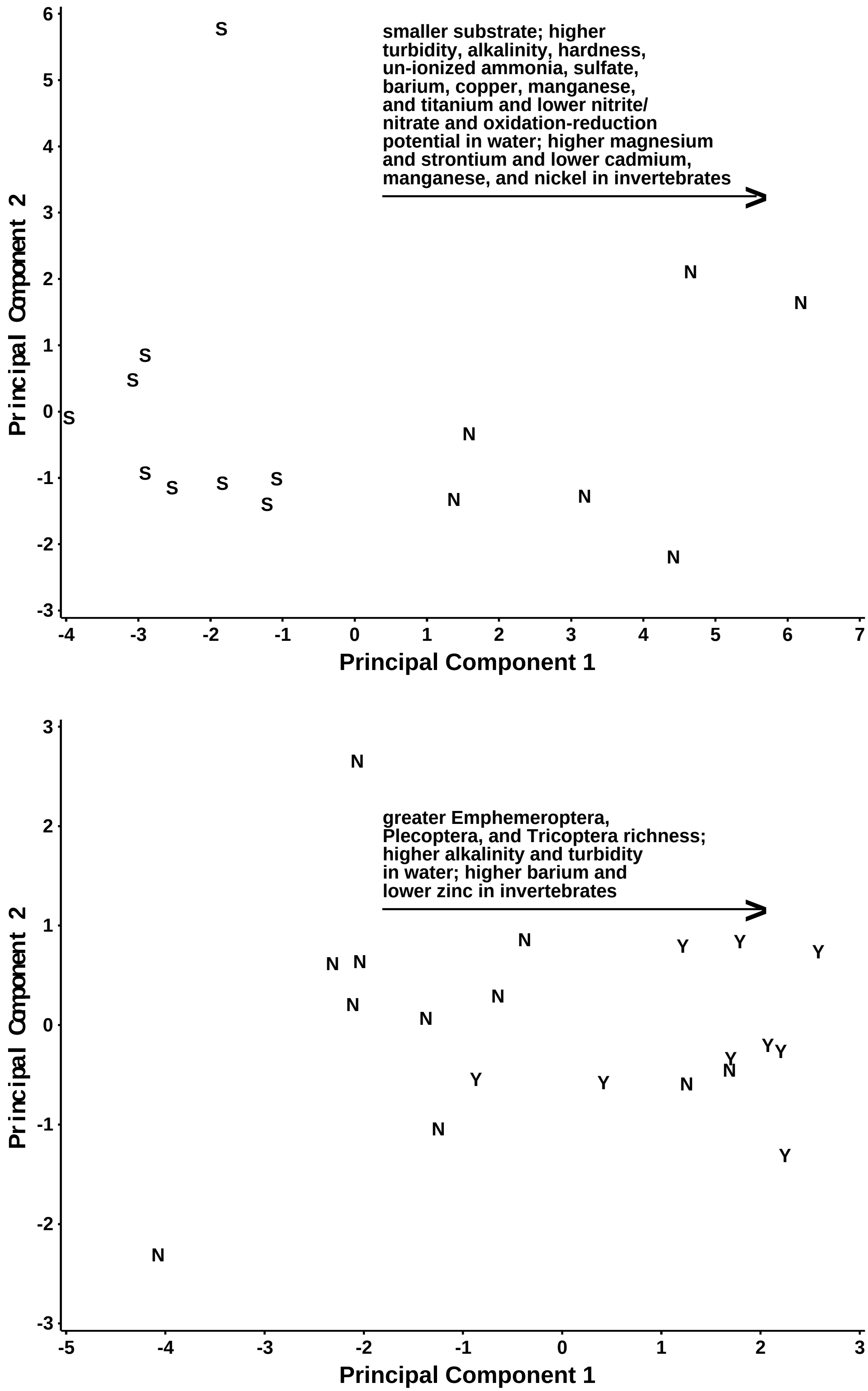

Figure 4. Principle components analyses differentiating Neosho-Cottonwood (N) from Spring River (S) sites (upper panel); and Spring River sites with (Y) and without (N) Neosho madtoms (lower panel).

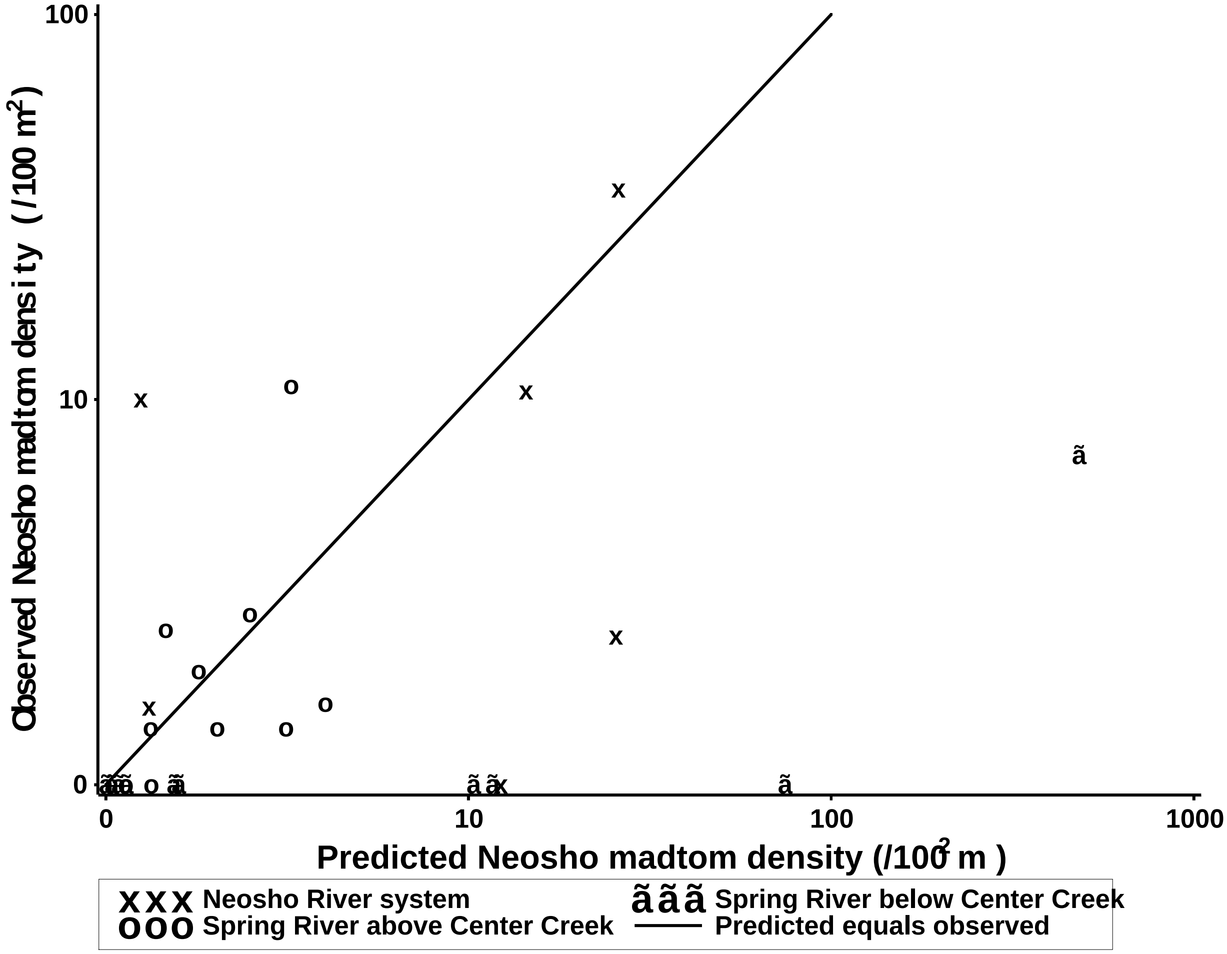

Figure 5. Predicted vs. observed Neosho madtom densites (Wildhaber et al. in press b).

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