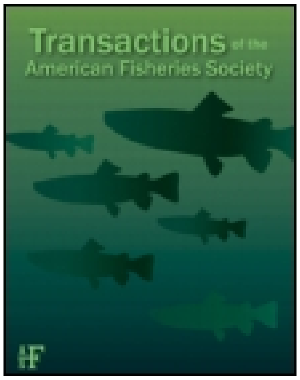

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Fish Assemblages and Environmental Variables Associated with Hard-Rock Mining in the Coeur d'Alene River Basin, Idaho

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Abstract.—As part of the U.S. Geological Survey's National Water Quality Assessment Program, fish assemblages, environmental variables, and associated mine densities were evaluated at 18 test and reference sites during the summer of 2000 in the Coeur d'Alene and St. Regis river basins in Idaho and Montana. Multimetric and multivariate analyses were used to examine patterns in fish assemblages and the associated environmental variables representing a gradient of mining intensity. The concentrations of cadmium (Cd), lead (Pb), and zinc (Zn) in water and streambed sediment found at test sites in watersheds where production mine densities were at least 0.2 mines/km^2 (in a 500-m stream buffer) were significantly higher than the concentrations found at reference sites. Many of these metal concentrations exceeded Ambient Water Quality Criteria (AWQC) and the Canadian Probable Effect Level guidelines for streambed sediment. Regression analysis identified significant relationships between the production mine densities and the sum of Cd, Pb, and Zn concentrations in water and streambed sediment (r^2 = 0.69 and 0.66, respectively; $P < 0.01$). Zinc was identified as the primary metal contaminant in both water and streambed sediment. Eighteen fish species in the families Salmonidae, Cottidae, Cyprinidae, Catostomidae, Centrarchidae, and Ictaluridae were collected. Principal components analysis of 11 fish metrics identified two distinct groups of sites corresponding to the reference and test sites, predominantly on the basis of the inverse relationship between percent cottids and percent salmonids ($r = -0.64$; $P < 0.05$). Streams located downstream from the areas of intensive hard-rock mining in the Coeur d'Alene River basin contained fewer native fish and lower abundances as a result of metal enrichment, not physical habitat degradation. Typically, salmonids were the predominant species at test sites where Zn concentrations exceeded the acute AWQC. Cottids were absent at these sites, which suggests that they are more severely affected by elevated metals than are salmonids.

Hard-rock mining (excluding sand and gravel mining) activities over the last century have profoundly altered water quality, aquatic biological, and hydrologic conditions in the Coeur d'Alene River basin (Ellis 1940; Hoiland et al. 1994; Horowitz et al. 1995; Woods and Beckwith 1997; Maret and Dutton 1999). From the late 1800s to early 1980s, the Coeur d'Alene Mining District, located in the Coeur d'Alene valley of north Idaho, was among the nation's leading producers of lead (Pb), silver, and zinc (Zn). These past mining activities have resulted in Superfund investigations and remediations in the South Fork Coeur d'Alene River. In addition, the U.S. Department of the Interior is currently conducting Natural Resource Damage Assessments (NRDA) and Abandoned Mine Lands (AML) monitoring and remediation in the Coeur d'Alene basin. Despite treatment and cleanup of mine wastes, cadmium (Cd), Pb, and Zn concentrations continue to exceed the U.S. Environmental Protection Agency (USEPA) Ambient Water Quality Criteria (AWQC) and streambed sediment guidelines for the protection of aquatic life (Hornig et al. 1988; Farag et al. 1998; Brennan et al. 1999; Maret and Skinner 2000; Stratus Consulting 2000).

Metals can affect aquatic organisms as toxic substances in water and sediment or as a toxicant in the food chain (Sorensen 1991; Rainbow 1996). The metals Cd, Pb, and Zn associated with mining activities in the Coeur d'Alene River basin have been shown to reduce the survival and growth of resident fish species (Farag et al. 1999). The primary mechanisms of acute metal-induced mortality are structural damage to the gill epithelium and respiratory failure (Evans 1987). Chronic metal poisoning in fish can affect growth and survival and may occur through waterborne and dietary exposure (Sorensen 1991; Farag et al. 1994). Recent studies in the Coeur d'Alene River basin (Woodward et al. 1997; Goldstein et al. 1999) indicated that the behavioral avoidance of metals by fish may have contributed to reduced fish populations.

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Moreover, fish assemblages associated with mining-impacted sites in Colorado and Montana consist of fewer native species and lower abundances (Hughes 1985; McCormick et al. 1994).

Few studies relate fish assemblages to mining activities in the Coeur d'Alene River basin. Laumeyer (1976) completed a fish distribution study of the basin but did not evaluate the fish assemblages in relation to stream conditions. Fisher (1989) developed an index of biotic integrity for headwater streams in north and central Idaho but did not specifically evaluate mining impacts. Reiser (1999) characterized fish populations and stream conditions resulting from mining activities as part of the NRDA of the Coeur d'Alene River basin. Assessment by Reiser (1999) conducted during 1994–1996, demonstrated that fishery resources in the Coeur d'Alene River and major tributaries have been changed as a result of mining and mineral processing operations.

As part of the U.S. Geological Survey (USGS) National Water Quality Assessment Program, fish, environmental variables, and associated mine densities were evaluated at 18 stream sites in the Coeur d'Alene and St. Regis river basins during summer 2000. The objectives of this study were to (1) determine the primary associations between fish assemblages and environmental variables in streams affected by hard-rock mining in the Coeur d'Alene River basin, (2) describe the extent and severity of mining impacts on biotic integrity by using selected measures of the fish assemblages, and (3) characterize the biotic and abiotic conditions of reference streams in this area to help formulate recovery goals and implement effective remediation actions. In the present study we test the hypothesis that streams affected by mining and associated metals consist of fewer native fish species and lower abundances. These findings will provide resource managers with a better understanding of how coldwater fish assemblages change as a result of metal contamination associated with hard-rock mining in the northwestern United States.

Study Area

The study area—consisting of the Coeur d'Alene and St. Regis river basins in northern Idaho and western Montana (Figure 1A)—is located entirely in the Northern Rockies ecoregion (Omernik and Gallant 1986), which is composed primarily of coniferous forests. Land use for the 18 watersheds ranges from 85% to 99% forested land, and the remainder consists primarily of rangeland. Agricultural and urban land uses constitute less than 1% of all watersheds. The study area has a complex geologic history of sedimentation, compressional deformation, igneous activity, and, most recently, extensional block faulting (Kendy and Tresch 1996). Rock types in the study area are almost entirely metasedimentary. Because mineralized areas in the Coeur d'Alene and St. Regis basins are extensive, metal concentrations can be relatively high in undisturbed streams (Maret and Skinner 2000).

Streams in the study area are coldwater, with maximum daily temperatures generally less than 22°C (Brennan et al. 1999). The main source of surface water and groundwater is snowmelt runoff from April to July. Streamflow conditions for the study period (summer 2000) were similar to or below the long-term average. Mean August flows for 2000 were about 84% of the long-term average streamflow for the North Fork Coeur d'Alene River near Enaville and the South Fork Coeur d'Alene River near Pinehurst. Streams in the study area typically have high water clarity, coarse-grained substrates (cobble and boulders), high stream gradients (>1%), well-defined riffles and pools, and very sparse macrophyte growth. Mining and logging practices, channelization, and roads have affected riparian areas, reduced instream habitat, and degraded stream- and groundwater quality (Woods and Beckwith 1997; Beckwith 1998).

At least 269 active and abandoned hard-rock mines of various sizes are located in the study area (Figure 1A). Production mine densities (active and past) in the 18 watersheds range from 0.001 to 0.42/km^2 (Table 1). Historically, metal extraction and processing were relatively inefficient, yielding large volumes of metal-rich tailings that were deposited in and around nearby streams. Many of these mines were located in close proximity to streams. Mine tailings in the study area typically contain elevated levels of trace metals such as arsenic, Cd, copper, Pb, mercury, and Zn (Woods and Beckwith 1997). These tailings and mines continue to provide a source of trace metals to streams, lakes, and reservoirs as streams meander through and erode tailing deposits and transport them downstream. Mine tailings entering the South Fork Coeur d'Alene River have been transported and deposited along the river channel and flood plain into Coeur d'Alene Lake and downstream into the Spokane River. Horowitz et al. (1995) estimated that 85% of the Coeur d'Alene Lake bed is enriched with trace metals. The USEPA (1997) National Sediment In-

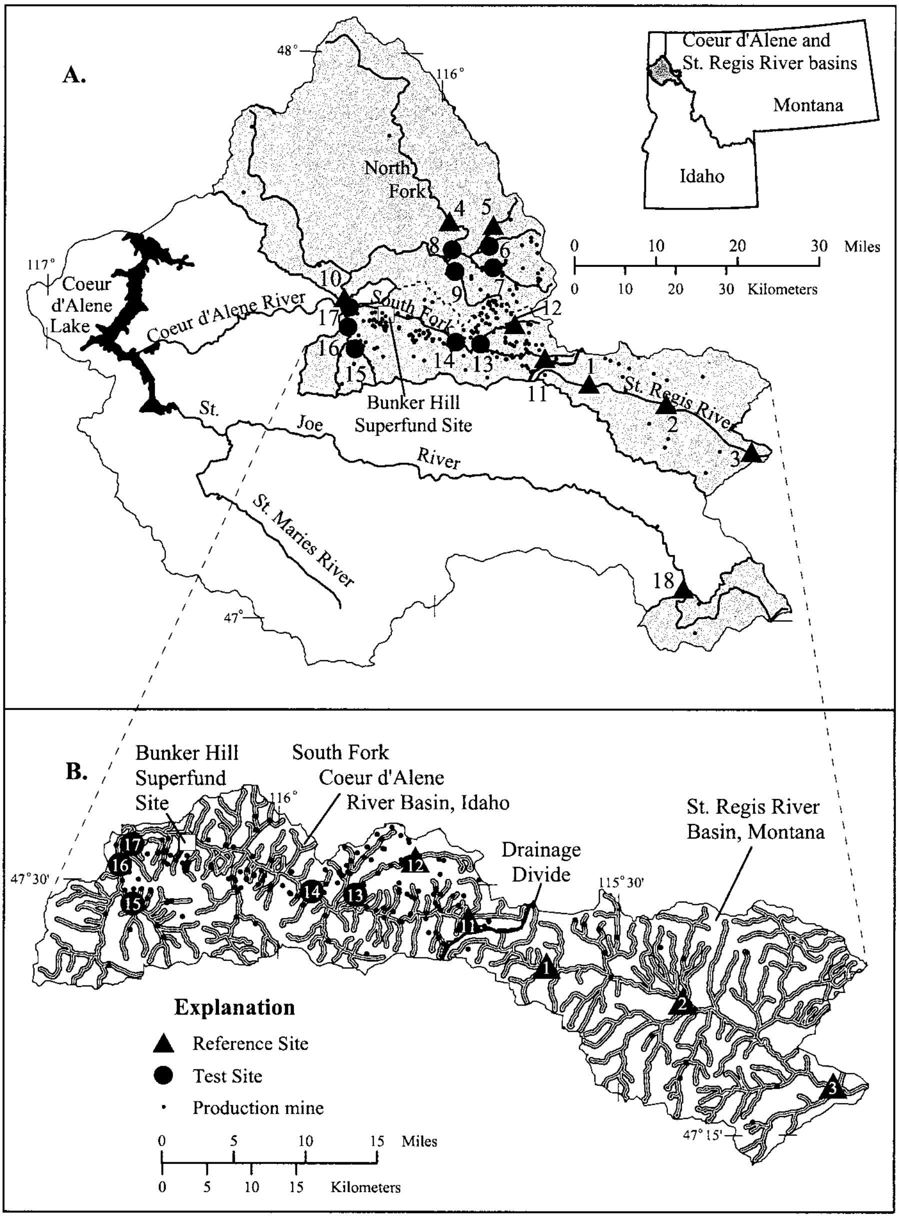

FIGURE 1.—**(A)** Map of the study area (gray shading) showing sampling sites and production mines (active and past) in the Coeur d'Alene and St. Regis river basins. **(B)** Map of the South Fork Coeur d'Alene and St. Regis river basins showing the 500-m stream buffer used to quantify production mine densities in the study basins.

ventory identified the Coeur d'Alene River and Lake as ''areas of probable concern for sediment contamination,'' the most severe contamination category in their assessment.

The composition of the native salmonid population has been altered as a result of stocking programs in the study area (Reiser 1999). Rainbow trout *Oncorhynchus mykiss* have been the most commonly stocked fish in the Coeur d'Alene River basin. Because rainbow trout compete and hybridize with wild cutthroat trout *O. clarki* (Leary et al. 1995), the Idaho Department of Fish and Game no longer stocks these fish in rivers with wild cutthroat trout populations (N. Horner, Idaho Department of Fish and Game, personal communication).

TABLE 1.—Sampling sites, production mine densities in the watersheds and 500-m buffers (250 m from each bank) upstream from each site, and site types (R = reference, T = test). Data on production mines (active and past) are from U.S. Bureau of Mines (1995).

Site	Site number[a]	Production mine density (km^2)		Site type
		Watershed	Buffer	
Montana				
St. Regis River above Rainy Creek	1	0.040	0.000	R
St. Regis River near Haugan	2	0.039	0.075	R
St. Regis River near St. Regis	3	0.025	0.058	R
Idaho				
North Fork Coeur d'Alene River near Prichard	4	0.001	0.003	R
West Fork Eagle Creek below Settlers Grove	5	0.068	0.162	R
East Fork Eagle Creek near Murray	6	0.076	0.200	T
Upper Prichard Creek near Murray	7	0.218	0.354	T
Prichard Creek at Prichard	8	0.154	0.315	T
Beaver Creek near Murray	9	0.180	0.200	T
North Fork Coeur d'Alene River near Enaville	10	0.027	0.034	R
South Fork Coeur d'Alene River above Mullan	11	0.176	0.000	R
Canyon Creek near Burke	12	0.145	0.197	R
Canyon Creek at Woodland Park	13	0.420	1.038	T
South Fork Coeur d'Alene River at Silverton	14	0.314	0.498	T
East Fork Pine Creek above Nabob Creek near Pinehurst	15	0.162	0.396	T
Pine Creek below Amy Gulch near Pinehurst	16	0.218	0.483	T
South Fork Coeur d'Alene River near Pinehurst	17	0.233	0.412	T
St. Joe River at Red Ives Ranger Station	18	0.011	0.032	R

[a] See Figure 1.

Methods

Site selection.—Eighteen sampling sites in the study area were selected to represent streams influenced by a range of mining activities, from minimally disturbed reference sites with little or no prior mining activity to those affected by cumulative mining impacts in the watershed (Table 1). Criteria suggested by Hughes et al. (1986) were used to select reference sites for this study. The selection process included the examination of existing data, consultation with local land management agencies familiar with streams in the area, and reconnaissance of candidate sites prior to sampling to aid in site selection. Reference sites represented regional conditions (site 18), conditions upstream from major mining activities (sites 4, 5, 10, 11, and 12), and conditions in similar (paired) watersheds (sites 1, 2, and 3 in the St. Regis River basin, and sites 11, 14, and 17 in the South Fork Coeur d'Alene River basin). To reduce the effects of stream size, each pair of sites was approximately the same distance from the mouth of the St. Regis or the South Fork Coeur d'Alene River, in river kilometers. The St. Regis and South Fork Coeur d'Alene river basins are similar in size, and site pairs have similar elevations and stream geomorphology. The location of hard-rock production mines provided by the U.S. Bureau of Mines (1995) Minerals Availability System spatial data also helped in the selection of sites affected by varying degrees of mining.

A site consisted of a representative reach generally containing repeating geomorphic channel units (riffles, runs, and pools). Because the length of stream sampled was a function of stream width, reach lengths ranged from 150 to 500 m (Fitzpatrick et al. 1998). The stream sites represented second- through fifth-order streams (Strahler 1957); all were wadeable except for sites 10 and 17.

Environmental variables.—Thirty-seven environmental variables were measured for each site (Table 2), including basin and segment characteristics, physical habitat, and water and streambed sediment physicochemistry. Watershed characteristics (land use, rock type, drainage area, stream order, and production mine densities) were determined by using the geographic information system Arc/Info. Several sources were used to construct the geographic data layers. Watershed boundaries were constructed using hydrography and hydrologic unit boundary data layers (USGS 1975). Stream order was determined from the 1:100,000-scale hydrography data layer (USGS 1985). Percentage land use (agricultural, forested, range, and urban) and percentage rock type were determined for each watershed. Land use was modified from

TABLE 2.—Statistical summary of environmental variables measured at all study sites in 2000 and Mann–Whitney *U*-test results. Variables with asterisks are significantly different ($P < 0.05$) between site types. Measurements of water velocity, dominant substrate, and depth were made at three points (the thalweg and two points equidistant between the thalweg and each bank) at each of 6–11 transects per site. Values represent means for all transect measurements. All measurements of water and streambed sediment physicochemistry were made during low flow. The following abbreviations are used: GIS = Geographic Information System; CV (= 100·SD/mean) = coefficient of variation.

		Test sites (N = 9)			Reference sites (N = 9)		
Variable	Data source	Mean (SD)	Median	Range	Mean (SD)	Median	Range
Watershed and segment characteristics							
Basin area (km^2)	GIS	194 (220)	100	52–738	517 (756)	233	17–2,325
Site elevation (m)*	GIS at reach boundary	783 (77)	798	668–908	954 (187)	1,039	664–1,155
Order	GIS analysis of 1:100,000 hydrography	3.2 (1.0)	3	2–5	3.1 (1.4)	4	2–5
Sinuosity	GIS analysis of stream segment	1.16 (0.09)	1.12	1.07–1.34	1.15 (0.13)	1.15	1.01–1.44
Forest land (%)	GIS analysis of entire basin	93 (5)	92	85–99	94 (4)	95	88–98
Rangeland (%)	GIS analysis of entire basin	5 (4)	6	1–11	4 (3)	3	0–9
Metasedimentary rock (%)	GIS analysis of entire basin	99 (1)	99	97–100	100 (0)	100	100
Physical habitat							
Discharge (m^3/s)	Marsh–McBirney flow meter	0.82 (1.06)	0.45	0.04–3.34	2.05 (2.54)	1.30	0.13–8.02
Gradient (%)	Determined with rod and level for entire reach	1.00 (0.49)	1.20	1.08–1.34	0.89 (0.75)	0.52	0.06–2.30
Riffle (%)	Surface area of reach	59 (24)	65	6–81	55 (23)	50	20–86
Instream cover (%)[a]	Visually determined from at least 30 observations; expressed as a percent of total possible cover types	6 (3)	6	1–11	9 (4)	9	3–14
Open canopy (%)	Determined using a clinometer to measure angles	56 (8)	56	46–72	42 (20)	41	19–79
Width (m)	Determined with rangefinder	10.3 (4.4)	8.1	6.1–19.2	18.6 (16.7)	10.3	4.8–53.6
Width (CV)*	Calculated	30 (11)	30	9–44	18 (7)	16	8–31
Depth (m)	Determined with measuring rod from at least 18 measurements	0.34 (0.17)	0.31	0.15–0.73	0.52 (0.56)	0.40	0.17–1.96
Depth (CV)	Calculated	82 (85)	51	17–299	58 (36)	45	28–150
Velocity (m/s)	Determined with Marsh–McBirney flowmeter from at least 18 observations	0.41 (0.12)	0.44	0.23–0.53	0.42 (0.10)	0.39	0.31–0.61
Velocity (CV)	Calculated	61 (19)	68	35–88	58 (11)	56	43–82
Width:depth ratio	Calculated	32 (10)	31	18–49	37 (17)	34	18–69
Dominant substrate (mm)	Median substrate size determined from at least 18 observations	143 (108)	96	24–384	119 (76)	96	80–320
Embeddedness (%)	Visually determined from at least 18 observations	12 (18)	3	0–46	5 (6)	3	0–17

TABLE 2.—Continued.

Variable	Data source	Test sites (N = 9) Mean (SD)	Median	Range	Reference sites (N = 9) Mean (SD)	Median	Range
Flow stability index[b]	Calculated	0.29 (0.10)	0.26	0.17–0.50	0.39 (0.19)	0.27	0.23–0.76
Bank stability index (score)[c]	Calculated from at least 12 observations of bank characteristics	10 (3)	9	8–17	12 (3)	14	8–16
Water and streambed sediment physicochemistry							
Specific conductance (μS/cm)	Instantaneous measurement at low flow	89 (80)	50	30–272	51 (22)	52	15–93
Dissolved oxygen (mg/L)	Instantaneous measurement at low flow	9.2 (0.5)	9.0	8.5–10.0	9.7 (0.5)	9.8	8.9–10.3
Water temperature (°C)*	Instantaneous measurement at low flow	15.1 (2.2)	15.0	12.6–18.9	11.5 (2.6)	11.3	8.0–15.0
pH	Instantaneous measurement at low flow	7.1 (0.4)	7.0	6.6–7.8	7.4 (0.3)	7.4	6.8–7.7
Hardness (mg/L)	Instantaneous measurement at low flow	38 (37)	19	10–120	21 (9)	22	6–33
Cadmium, dissolved (μg/L)*	Instantaneous measurement at low flow	3 (5)	0	0–13	0	0	0
Cadmium, streambed sediment (μg/g)*	Determined on <63-μm size fraction	30 (50)	3	2–150	0.7 (0.6)	0.6	0–1.7
Lead, dissolved (μg/L)	Instantaneous measurement at low flow	5 (10)	0	0–30	0	0	0
Lead, streambed sediment (μg/g)*	Determined on <63-μg size fraction	2,858 (3,723)	1,000	200–11,000	83 (70)	69	9–200
Zinc, dissolved (μg/L)*	Instantaneous measurement at low flow	503 (630)	110	16–1,510	4 (4)	3	0–12
Zinc, streambed sediment (μg/g)*	Determined on <63-μm size fraction	3,130 (3,740)	840	480–10,000	197 (139)	180	39–430
Total nitrate (mg/L)	Includes nitrite and nitrate concentrations	0.043 (0.067)	0.013	0–0.208	0.015 (0.007)	0.009	0–0.021
Total phosphorus (mg/L)	Instantaneous measurement at low flow	0.008 (0.024)	0	0–0.071	0	0	0
Ammonia, dissolved (mg/L)	Instantaneous measurement at low flow	0.030 (0.084)	0.002	0.002–0.254	0.003 (0.001)	0.002	0.002–0.004

[a] Determined from the presence/absence of all types of fish habitat cover at five locations (within a 1-m zone) along each transect, namely, at the three points where other instream measurements were made and at the left and right edges of water. Habitat types included boulders, wood, undercut banks, aquatic macrophytes, and artificial structures such as tires and riprap (Fitzpatrick et al. 1998).
[b] Ratio of thalweg depth at low flow to bank-full thalweg depth (Rosgen 1994).
[c] Scores of 4–7 indicate stable banks, scores of 8–10 banks at risk, and scores of 16–22 unstable banks (Fitzpatrick et al. 1998).

1:250,000-scale digital data (USGS 1986) consisting of a classification scheme at a 16-ha mapping resolution (Anderson et al. 1976). Geologic information was obtained from Tuck et al. (1996). Production mines (past and active) for each watershed were located by using the U.S. Bureau of Mines (1995) Minerals Availability System. Because most of the mining activity in the study area occurs near streams, a proximity analysis also was conducted on production mine locations relative to streams in each watershed. After extensive analysis and comparison of different buffer widths (including none), a 500-m buffer width (250 m from each bank) was selected for the entire drainage network upstream from a site (Figure 1B). This buffer best differentiates the mining activity that most directly influences the stream sites from land use farther removed from the stream. For all watersheds, the buffer represented only about 35% of the total drainage area but captured 59% (158 of 269) of all production mines.

Stream sinuosity and elevation were derived from USGS 1:24,000-scale Digital Raster Graphic maps. The length of each reach was measured, as well as the length of geomorphic channel units. Six to 11 transects were placed perpendicular to streamflow, equidistant throughout the reach (Fitzpatrick et al. 1998). Along each transect, percentage instream cover, percentage open canopy, wetted stream width, percentage embeddedness, and bank characteristics (height, angle, percentage vegetative cover, and substrate type) were measured. Depth, velocity, and cover types were measured at the thalweg and two intermediate locations along each transect. Velocities were measured at 0.6 of stream depth. Substrate (i.e., silt, sand, gravel, cobble, and boulder), percentage embeddedness, cover types, and bank features were visually categorized. Instream fish cover was visually determined within a 1-m strip along each transect at the three instream locations and the left and right banks. Instream cover categories were boulders, woody debris, undercut banks, aquatic macrophytes, and artificial structures (such as tires and riprap). The frequency of occurrence was determined for each cover type and then expressed as a percentage of all possible cover types for the reach. For data analysis, a mean value was calculated to represent a reach for those variables with multiple measures. Discharge at the time of sampling was measured on-site or taken from streamgaging records. The stream gradient for each site was determined on-site for each reach by using a rod and level. The width:depth ratio was determined as a relative measure of channel cross-sectional shape. Three measures of habitat complexity were calculated, namely, the coefficients of variation (CV = 100 × SD/mean) for width, depth, and velocity (Gorman and Karr 1978; Rahel and Hubert 1991). A flow stability index was calculated as the ratio between low-flow depth and estimated bank-full depth (Rosgen 1994; Lammert and Allan 1999). A flow stability index approaching 1.0 represents less flow fluctuation between low- and bank-full flow events and, therefore, higher stability. A bank stability index was also calculated on the basis of measures of bank angle, height, substrate type, and vegetative cover (Fitzpatrick et al. 1998). Scores were assigned to each measure and summed for a total bank stability score. The mean of all bank scores was used to represent the bank stability index score for the reach. Scores of 4–7 represent stable banks; scores of 8–10 represent banks at risk of erosion; and scores of 16–22 represent unstable banks.

Water quality and streambed sediment samples were collected at each site during low-flow conditions in August and September 2000 using methods described by Shelton and Capel (1994). Water quality samples were collected for analysis of selected nutrients and trace metals. A portion of each sample was filtered through a 0.45-μm filter; analytes in this portion are hereafter referred to as dissolved. Samples were shipped on ice to the USGS National Water Quality Laboratory in Arvada, Colorado, for analysis (Fishman 1993). Field measurements of water temperature, dissolved oxygen, pH, and specific conductance also were recorded (Wilde and Radtke 1998).

Streambed sediment was sampled at 5–10 locations within a reach, usually along channel margins. The upper 2 cm of bed sediment was collected from undisturbed, continuously wetted, depositional zones by using a plastic scoop. The subsamples were composited and wet-sieved through a 63-μm nylon mesh, placed on ice, and submitted to the laboratory for analysis. Because trace metals are not expected to significantly sorb to larger sediment particles, a 63-μm sieve size was chosen to normalize the size fraction between sites and to maximize the concentrations of trace metals in the streambed sediment. The trace metals in the bed sediment were analyzed by the USGS Geologic Division Mineral Resources Laboratory in Lakewood, Colorado, by using inductively coupled plasma-mass spectrometry (Briggs and Meier 1999).

Field quality assurance (QA) consisted of water quality blanks and duplicate sediment samples. Laboratory QA consisted of routine blank analyses and replicates for water samples (Fishman 1993), and analyses of reference material for sediment samples (Arbogast 1990). There were no metals detected in blank water samples above the minimum reporting limit. Concentrations of Cd, Pb, and Zn in duplicate sediment samples had relative differences of 0–15%, and values for metals in reference material were within quality assurance guidelines (±3 standard deviations). All other analysis results for QA laboratory samples were within acceptable USGS method standards.

Fish collections.—Fish sampling was conducted at all sites (except site 10) during low streamflow conditions in August and September 2000. Because of equipment failure at site 10, a more representative fish assemblage collected in 1998 was used in the analysis. Fish sampling consisted of single-pass electrofishing. Fish were collected from most sites with backpack electrofishers (Smith-Root model 12) using pulsed direct current. Sampling was conducted in an upstream direction and included all available habitats within the reach. In larger, wadeable streams (>20 m wide), two backpack electrofishers were used simultaneously. Two unwadeable reaches (sites 10 and 17) were sampled by using both a backpack electrofisher and an electrofishing boat. The boat was equipped with a Smith-Root model VI-A direct current pulsator and a 5,000-W, 240-V generator with bow-mounted electrodes and motor. Sampling effort was measured as the total time that power was applied to the electrodes, typically from 1,000–4,000 s.

Fish were anesthetized with a dilute solution of clove oil and ethanol, identified to species, counted, measured for total length and weight, examined for external anomalies following methods described by Meador et al. (1993), and returned to the stream. Specimens of selected species were retained for reference and verification of field identifications. Age-class determinations for salmonids and cottids were based on length-frequency distributions and descriptions by Wydoski and Whitney (1979) and Scott and Crossman (1973). A voucher collection from this study is located in the Orma J. Smith Museum of Natural History, Albertson College of Idaho, Caldwell, Idaho. Because tailed frogs *Ascaphus truei* have been shown to be effective indicators of stream conditions (Bennett and Fisher 1989; Corkran and Thoms 1996), the number collected by electrofishing at each site was noted.

Data analysis.—To facilitate data analysis, sampling sites were categorized as either reference or test sites on the basis of production mine densities upstream and the concentrations of metals in the water and streambed sediment. The median mine density of 0.2 mines/km^2 in the 500-m buffer was used to divide sites into two equal groups of test and reference sites. Test sites were those with production mine densities of 0.2 mines/km^2 or more in the stream buffer (Table 1) and elevated metal

TABLE 3.—Summary of cadmium, lead, and zinc concentrations in water (dissolved; filtered through a 0.45-μm filter) and streambed sediment (<63 μm), August–September 2000. Cadmium concentrations in water noted as less than 1.0 μg/L may have exceeded water quality criteria at low hardness (<30 mg/L), but this could not be determined because of the 1.0-μg/L detection level. Asterisks denote reference sites.

Site number	Water (μg/L)			Streambed sediment (μg/g dry weight)		
	Cadmium	Lead	Zinc	Cadmium	Lead	Zinc
1*	<1.0	<1.0	12	1.1	81	430[c]
2*	<1.0	<1.0	5.3	0.5	41	180
3*	<1.0	<1.0	4.5	<0.1	20	74
4*	<1.0	<1.0	<1.0	<0.1	20	62
5*	<1.0	<1.0	3.1	0.6	69	100
6	<1.0	<1.0	67[a]	3.0	1,000[c]	790[c]
7	<1.0	<1.0	49[a]	3.1	710[c]	840[c]
8	<1.0	<1.0	24	2.1	460[c]	480[c]
9	<1.0	<1.0	16	2.2	200[c]	620[c]
10*	<1.0	<1.0	1.6	1.1	160[c]	290
11*	<1.0	<1.0	3.4	1.4	150[c]	340[c]
12*	<1.0	<1.0	2.8	1.7	200[c]	260
13	13[a]	30[a]	1,510[a]	62[c]	11,000[c]	8,200[c]
14	7.0[a]	10[b]	990[a]	43[c]	4,400[c]	5,400[c]
15	1.2[a]	<1.0	320[a]	3.6[c]	1,000[c]	1,100[c]
16	<1.0	<1.0	110[a]	3.2	550[c]	740[c]
17	8.7[a]	5.6[b]	1,440[a]	150[c]	6,400[c]	10,000[c]
18*	<1.0	<1.0	<1.0	<0.1	9	39

[a] Concentration exceeds acute and chronic criteria for the protection of aquatic life (USEPA 2000).

[b] Concentration exceeds chronic criteria for the protection of aquatic life (USEPA 2000).

[c] Concentration exceeds Probable Effect Level for aquatic life (Cd = 3.53, Pb = 91.3, and Zn = 315 μg/g [dry weight]; Canadian Council of Ministers of the Environment 1995).

concentrations in water or streambed sediment that exceeded one or more guidelines for the protection of aquatic life (Table 3). Only Cd, Pb, and Zn concentrations are reported in this study because they have been identified as the metals most elevated in the Coeur d'Alene River basin streams and of greatest ecological concern (Brennan et al. 1999; Stratus Consulting 2000).

The Canadian Interim Sediment Quality Guidelines (ISQG) were used to evaluate the severity of metal contamination in streambed sediment. More specifically, the ISQG Probable Effect Level (PEL) for each metal, defined as the concentration above which adverse effects to aquatic life are predicted to occur frequently, was used (Canadian Council of Ministers of the Environment 1995). Because the less than 63-μm size fraction, rather than bulk sediment, was analyzed, the likelihood of exceeding a guideline may be increased. Dissolved metals in water were compared with USEPA (2000) acute and chronic AWQC for the protection of aquatic

TABLE 4.—Origins (N = native, I = introduced), temperature preferences, relative abundances, and collection sites for fish collected in the Coeur d'Alene and St. Regis river basins in 2000 (1998 for site 10). Origin and temperature preferences are from Zaroban et al. (1999).

Species	Origin	Water temperature preference	Relative abundance (%)	Collection
Brook trout *Salvelinus fontinalis*	I	Cold	7.8	1, 2, 3, 5, 6, 7, 8, 9, 10, 11, 14, 15, 16, 17
Brown trout *Salmo trutta*	I	Cold	0.7	2, 3
Bull trout *Salvelinus confluentus*	N	Cold	0.1	18
Cutthroat trout	N	Cold	15.5	1, 2, 4, 5, 6, 7, 8, 9, 10, 11, 12, 14, 15, 16, 17, 18
Mountain whitefish *Prosopium williamsoni*	N	Cold	1.0	2, 3, 10, 17, 18
Rainbow trout	I	Cold	1.9	2, 3, 10, 14, 17
Shorthead sculpin *Cottus confusus*	N	Cold	65.0	1, 2, 3, 4, 5, 8, 9, 10, 11, 12, 18
Torrent sculpin *Cottus rhotheus*	N	Cold	1.6	4, 8, 10
Bridgelip sucker *Catostomus columbianus*	N	Cool	0.2	10
Longnose sucker *Catostomus catostomus*	N	Cold	0.1	17
Largescale sucker *Catostomus macrocheilus*	N	Cool	0.2	4, 10, 17
Longnose dace *Rhinichthys cataractae*	N	Cool	1.9	3, 4, 8, 9, 10, 18
Northern pikeminnow *Ptychocheilus oregonensis*	N	Cool	1.3	10
Redside shiner *Richardsonius balteatus*	N	Cool	0.8	10
Speckled dace *Rhinichthys osculus*	N	Cool	0.8	4
Tench *Tinca tinca*	I	Warm	0.2	17
Pumpkinseed *Lepomis gibbosus*	I	Cool	0.1	10
Brown bullhead *Ameiurus nebulosus*	I	Warm	0.8	10, 17

life. Dissolved, rather than total recoverable, metals were evaluated in this study because the dissolved fraction is the most bioavailable and potentially toxic. Because water hardness affects bioavailability and the toxicity of most metals, criterion values were modified to account for variation in the water hardness among streams (USEPA 2000). This study may not accurately describe all occasions where Cd criteria were exceeded because of the low water hardness and laboratory detection level of 1.0 μg/L. On occasions where hardness is below 30 mg/L, acute and chronic criteria of less than 1.0 μg/L for Cd are possible.

Because criteria are available only for individual metals in water and streambed sediment, Clements et al. (2000) developed a simple index to account for the additive effects of metal toxicity on aquatic life. Cumulative toxic units (CTU) for Cd, Pb, and Zn in water and streambed sediment were calculated for this study in a similar manner. Following recommendations by Dyer et al. (2000), acute AWQC and the ISQG PEL values for these metals were used as toxicity reference values (TRV). Each CTU value represents the sum of Cd, Pb, and Zn concentrations in water and streambed sediment at each site divided by the appropriate TRV. A CTU value above 1.0 represents potentially toxic levels of cumulative Cd, Pb, and Zn in water or streambed sediment at a site. Our goal was to provide resource managers with a measure of the severity of metal contamination in the streams sampled.

All statistical analyses were performed using SYSTAT (Wilkinson 1999). All possible combinations of responses and explanatory variables were examined using Spearman's rank correlation matrices. Relations between mine densities and the sum of Cd, Pb, and Zn concentrations in water and streambed sediment were described using regression analyses. Significant differences between reference and test sites were determined with the nonparametric Mann–Whitney *U*-test. Nondetections of chemical constituents (below the minimum reporting level) were set to zero prior to statistical analysis.

Fish assemblages were analyzed as relative abundance data and 11 fish metrics for evaluating stream conditions and human disturbances (Bennett and Fisher 1989; Maret et al. 1997; Grafe 2000; Mebane 2001). These metrics were: number of coldwater native species, number of cottid age classes, number of salmonid age classes, percent coldwater individuals, percent introduced individuals, percent salmonids, percent cottids, percent external anomalies, catch per unit effort (CPUE, fish per minute of electrofishing), cottid biomass, and salmonid biomass. Geographic origin (native or introduced) and temperature preferences were assigned to each species (Table 4) following methods described by Zaroban et al. (1999).

Principal components analysis (PCA) was performed on fish metrics to summarize the relationships among metrics. Prior to this analysis, fish metrics were examined graphically for normality. Percent introduced individuals, percent salmonids, percent cottids, and salmonid biomass were $\log_{10}(x + 1)$ transformed to improve normality. Principal components having eigenvalues greater than 1 were retained and rotated by use of the varimax procedure in SYSTAT (Wilkinson 1999). SYSTAT standardizes variables to a mean of 0 and standard deviation of 1.

Results

Environmental Variables

The statistical summaries of watershed and segment characteristics, physical habitat, and water and streambed sediment physicochemistry describe expected conditions for the reference and test sites (Table 2). Eight environmental variables were significantly different ($P < 0.05$) between reference and test sites: site elevation, CV of width, water temperature, Cd concentrations in water and streambed sediment, Pb concentrations in streambed sediment, and Zn concentrations in water and streambed sediment. Site elevations were slightly higher for reference sites because some reference sites were necessarily located upstream from mining areas. Although median water temperatures were significantly higher for test (15.1°C) than for reference (11.5°C) sites, temperatures at all sites were below Idaho's Water Quality Standards criterion of 19°C for the protection of coldwater aquatic life (Idaho Department of Health and Welfare 1999). The CV of width was higher for test sites for reasons that are unclear. The instream cover for fish was limited, ranging from 1% to 11% for all sites (Table 2), and comprised mostly boulders. Large, woody debris was scarce or absent at many sampling sites.

The concentrations of dissolved Cd, Pb, and Zn in water and streambed sediment were significantly higher at test sites than at reference sites. The only exception was dissolved Pb in water, which was not significantly different. It is not surprising that there is no difference in dissolved Pb between site types because Pb is associated predominantly with sediment (Hem 1985). More than 75% of the dissolved Cd and Pb concentrations in water were less than the reporting limit; only at test sites 13, 14, 15, and 17 were concentrations of one or both of these metals measurable (Table 3).

Many of the metal concentrations in water and sediment exceeded the acute and chronic AWQC and PEL (Table 3). Some of the highest metal concentrations in water and streambed sediment were associated with the test sites Canyon Creek (site 13), East Fork Pine Creek (site 15), and South Fork Coeur d'Alene River (sites 14 and 17). However, concentrations of Pb and/or Zn at reference sites 1, 10, 11, and 12 also exceeded the PEL for streambed sediment. Sites 1, 11, and 12 are all headwater sites upstream from most mining activities. In fact, site 1 on the St. Regis River above Rainy Creek is in the Clark Fork basin on the other side of the mountain divide from the Coeur d'Alene River basin. These elevated metal concentrations in streambed sediment at reference sites may be the result of the limited mining upstream from these sites, natural mineral deposits, or prevailing winds that historically transported airborne metal particulates from the nearby smelter operations at the Bunker Hill Superfund site near Kellogg, Idaho. The dissolved metals in water at the reference sites did not exceed acute or chronic AWQC.

Regression analysis (Figure 2) identified significant relationships between the production mine densities (500-m stream buffer) in the different basins and the sum of Cd, Pb, and Zn concentrations in water and streambed sediment ($r^2 = 0.69$ and 0.66, respectively; $P < 0.01$),. Thus, concentrations of Cd, Pb, and Zn in water and sediment are directly related to the number of mines in close proximity to streams upstream from sampling sites. Combined metal concentrations in both water and streambed sediment at test sites generally exceeded reference site concentrations, some by as much as 1–2 orders of magnitude. Zinc was the major component of metal concentrations in sediment, averaging about 63% of the sum of concentrations at all sites. Cadmium represented a very small component at less than 1%, and Pb was intermediate at 36%. Nearly the entire sum of metal concentrations in water at all sites, ranging from approximately 97 to 100% when metals were detected, was composed of dissolved Zn. Lead and Cd concentrations were detected at test sites only in water and composed less than 2% of the sum of concentrations in water.

The relative severity of the Cd, Pb, and Zn contamination in streambed sediment and water is shown by ranking all sites using the CTU scores above 1.0 (Figure 3). CTUs for streambed sediment and water for all sites were significantly correlated ($r = 0.93$; $P < 0.05$) and varied greatly among sampling sites, ranging from 1.4 to 164.1 for streambed sediment and from 1.4 to 34.9 for water. The CTUs for both water and streambed sediment were highest at Canyon Creek at Wood-

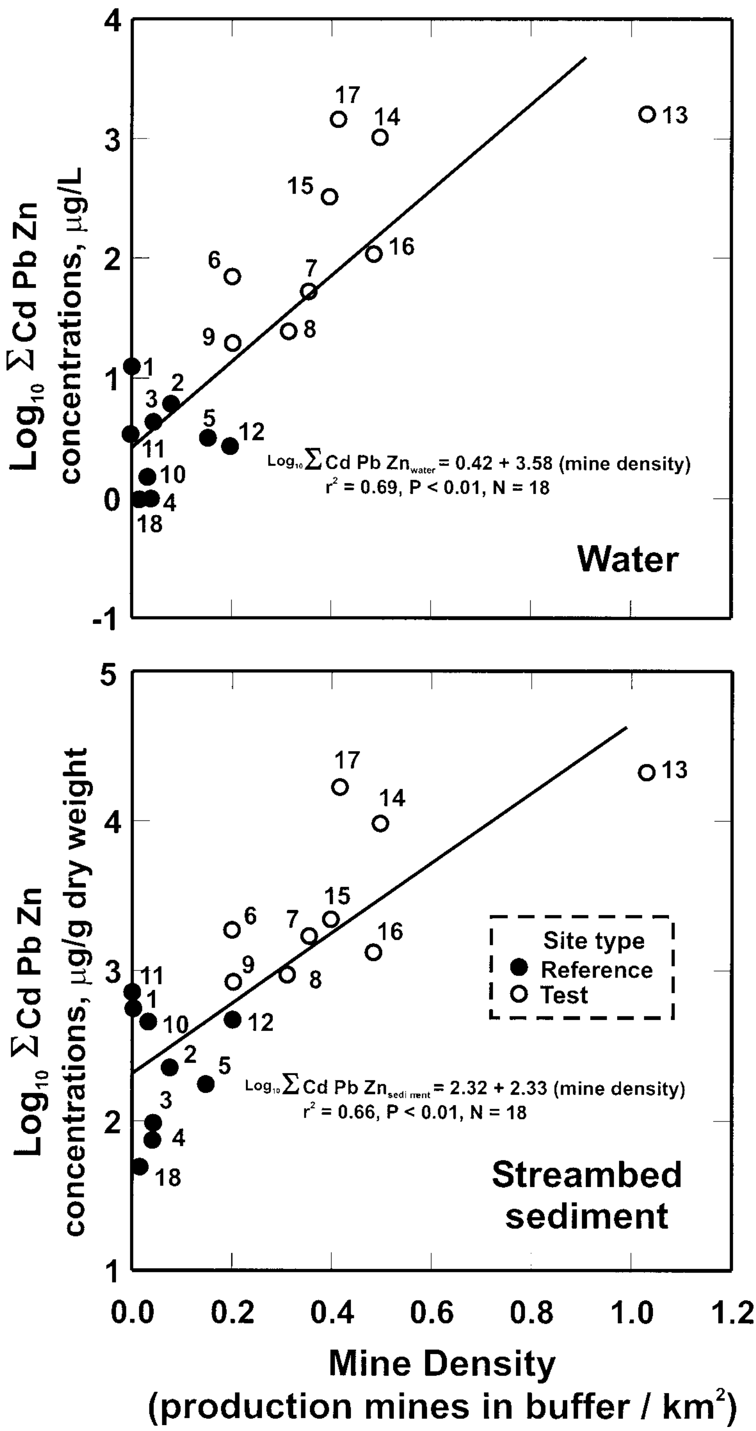

FIGURE 2.—Relationships between production mine density in 500-m stream buffer and the sum of Cd, Pb, and Zn concentrations in water and streambed sediment. Sampling sites are identified as reference and test sites (Table 1; Figure 1).

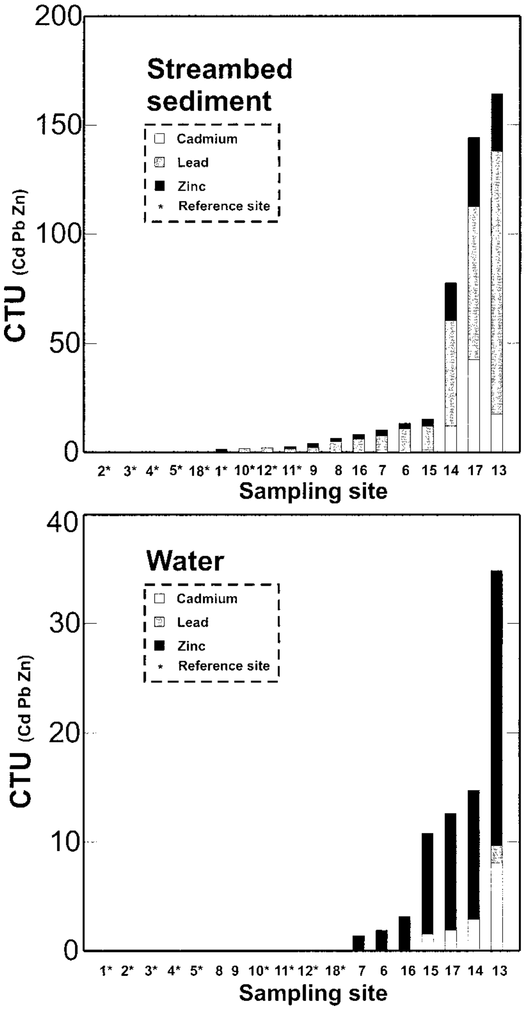

FIGURE 3.—Ranking of sampling sites on the basis of cumulative toxic unit (CTU) scores for Cd, Pb, and Zn concentrations in streambed sediment and water. Index values for streambed sediment are based on Canadian Interim Sediment Quality Guidelines Probable Effect Level (Canadian Council of Ministers of the Environment 1995). Index values for water (dissolved concentrations) are based on Ambient Water Quality Criteria (USEPA 2000). Only CTUs greater than 1.0 are shown; each such CTU is an estimate of the cumulative toxicity of metals at a site.

land Park (site 13), indicating that this was the most contaminated site sampled. Since the CTUs for most reference sites were not above 1.0, cumulative effects from Cd, Pb, and Zn contamination are unlikely. However, the CTUs for streambed sediment and water at test sites 6, 7, 13, 14, 15, 16, and 17 exceeded 1.0, indicating that cumulative metal concentrations are potentially toxic to aquatic life (especially benthic invertebrates living on or in the streambed sediment).

Fish Species and Metrics

Eighteen species of fish in the families Salmonidae, Cottidae, Cyprinidae, Catostomidae, Centrarchidae, and Ictaluridae were collected (Table 4). A total of 3,535 fish were collected from the 18 sites sampled during this study. Fish collected at all sites appeared in good health with few external anomalies (<0.1%). The number of species collected at a site ranged from 0 at Canyon Creek at Woodland Park (site 13) to 13 at North Fork Coeur d'Alene River near Enaville (site 10). Shorthead sculpin were collected at 61% of all sites and was the most abundant species, comprising 65% of all fish collected. Salmonids constituted 27% of all fish collected; cutthroat trout and brook trout were collected at 89% and 78% of the sampling sites, respectively. Introduced species collected during this study were brook trout, brown trout, rainbow trout, tench, pumpkinseed, and brown bullhead. Bull trout, a federally protected species, was collected only at site 18 (the St. Joe River). Tailed frogs were collected only at reference sites 5, 11, and 18.

The statistical summaries of the 11 fish metrics describe expected conditions for the reference and test sites affected by mining activities in the study area (Table 5). Seven fish metrics were significantly different ($P < 0.05$) between reference and test sites: the number of coldwater native species, the number of cottid age-classes, percent introduced individuals, percent salmonids, percent cottids, CPUE, and cottid biomass. The number of coldwater native species, number of cottid age classes, percent cottids, CPUE, and cottid biomass were significantly higher for reference sites, and percent introduced individuals and percent salmonids were significantly higher for test sites. Brook trout were the most common introduced species at test sites. The number of salmonid age-classes, percent coldwater individuals, percent external anomalies, and salmonid biomass were not significantly different between site types.

The results from PCA, based on 11 fish metrics, showed separation between most reference and test sites (Figure 4). Test sites 8 and 9 grouped with the reference sites, indicating that they contained similar fish assemblages. Two reference sites, 4 and 10 (both fifth-order streams), did not group with reference or test sites. Fewer coldwater individuals, lower salmonid biomass, and lower CPUE were recorded for these sites. These sites

TABLE 5.—Statistical summary of 11 fish metrics calculated for all study sites in 2000 and Mann–Whitney *U*-test results. Variables with asterisks are significantly different ($P < 0.05$) between site types.

Metric	Test sites ($N = 9$) Mean (SD)	Test sites Median	Test sites Range	Reference sites ($N = 9$) Mean (SD)	Reference sites Median	Reference sites Range
Number of coldwater native species*	1.3 (0.9)	1.0	0–3	2.6 (0.7)	2.0	2–4
Number of cottid age-classes*	1.1 (2.2)	0.0	0–5	4.9 (0.3)	5.0	4–5
Number of salmonid age-classes	4.1 (1.8)	5.0	0–5	4.4 (1.0)	5.0	2–5
Percent coldwater individuals	87.1 (32.9)	100.0	0–100	86.0 (27.2)	100.0	31–100
Percent introduced individuals*	32.9 (31.1)	32.2	0–84	5.9 (9.5)	1.6	0–29
Percent salmonids*	72.3 (41.3)	100.0	0–100	12.6 (9.2)	8.7	2–30
Percent cottids*	14.8 (30.8)	32.2	0–84	72.5 (26.6)	87	15–92
Percent external anomalies[a]	0.2 (1.9)	0.0	0–2	0.2 (0.5)	0.0	0–2
Catch per unit effort[b]*	2.85 (2.59)	2.59	0.00–8.11	5.34 (2.75)	6.14	0.85–8.15
Cottid biomass (g/m^2)*	0.06 (0.12)	0.00	0.00–0.33	0.51 (0.45)	0.47	0.01–1.25
Salmonid biomass (g/m^2)	0.82 (0.48)	0.70	0.00–1.58	0.93 (0.89)	0.73	0.08–3.05

[a] Includes deformities, eroded fins, lesions, and tumors.
[b] Fish captured per minute of electrofishing.

also contained more catostomid and cyprinid species, which are generally coolwater species (Table 4). Torrent sculpin, a cottid species characteristic of larger streams (Laumeyer 1976), was also found at these two sites.

Axes 1 and 2 of the PCA accounted for 44% and 29% of the variance among sites, respectively. The fish metrics with high factor loadings on axis 1 ($> \pm 0.70$) were: percent salmonids, percent introduced individuals, the number of coldwater native species, percent cottids, and the number of cottid age-classes. The metrics with high factor loadings on axis 2 were: percent coldwater individuals, salmonid biomass, and CPUE. The sites with low scores on PCA axis 1 (Figure 4) contained higher percent cottids and more age-classes of cottids, whereas the sites with high scores contained higher percent salmonids and more introduced individuals. This PCA clearly shows that the separation between test and reference sites is predominantly the result of the inverse relation between percent cottids and percent salmonids ($r = -0.64$; $P < 0.05$). Cottids were collected only at two of nine test sites. Test sites typically contained mostly salmonids; introduced brook trout constituted 44% to 84% of all fish collected at sites 7, 15, 16, and 17.

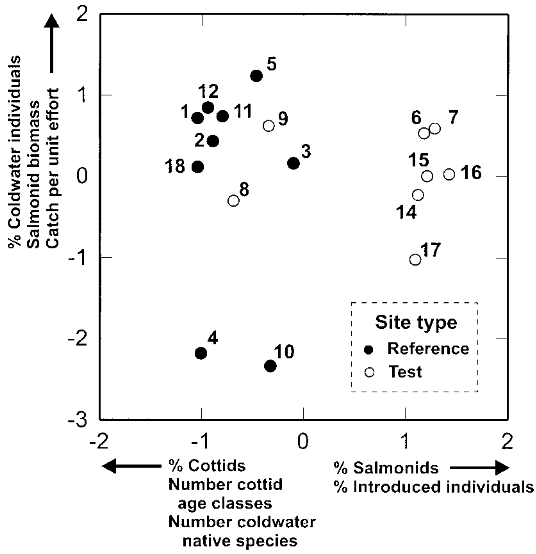

FIGURE 4.—Principal components ordination of the 17 sampling sites based on the 11 fish metrics shown in Table 5. Site 13 was not included because no fish were found there. Fish metrics with high factor loadings ($> \pm 0.70$) are listed along the axes; arrows indicate their direction of increase. Axes 1 (*x*-axis) and 2 (*y*-axis) accounted for 44% and 29% of the variance among sites, respectively.

The differences in fish assemblages between reference and test sites can be illustrated by comparing three sites in the South Fork Coeur d'Alene River basin, Idaho, with three sites in the St. Regis River basin, Montana (Figure 1B). Test sites 14 and 17 are affected by mining along the tributaries of Canyon and Nine Mile Creeks that flow into the South Fork Coeur d'Alene River upstream from site 14 (Figure 5). These tributaries carry high concentrations of Cd, Pb, and Zn, which increases metal loads into the South Fork Coeur d'Alene River by an order of magnitude or more (Woods 1999). Cottids composed about 70% to 90% of total individuals for reference sites 1, 2, and 3 in the St. Regis River basin, and site 11 upstream from major mining impacts in the South Fork Coeur d'Alene River. Because of their small size relative to salmonids, cottids composed 10% to

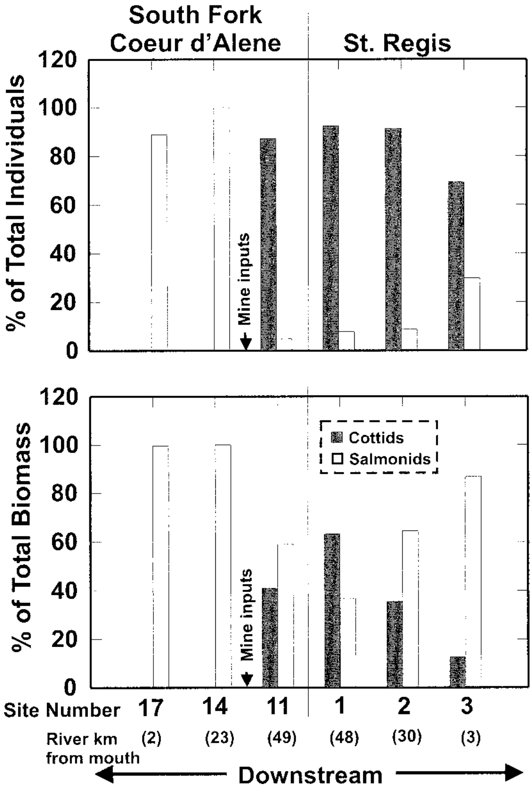

FIGURE 5.—Pairwise comparisons of percent total individuals (upper panel) and total biomass (lower panel) of cottids (shaded bars) and salmonids (clear bars) among three sites on the South Fork Coeur d'Alene River, Idaho, and three reference sites on the St. Regis River, Montana (see Figure 1B for site locations).

60% of the total biomass for these same sites. Cottids were absent at sites 14 and 17 downstream from the tributaries delivering high concentrations and loads of metals into the South Fork Coeur d'Alene River. In contrast, cottids were collected throughout the entire St. Regis River, from its headwaters to its confluence with the Clark Fork.

Fish Metrics and Environmental Relations

Seven fish metrics that were significantly different between reference and test sites were correlated with measures of mine densities and associated Cd, Pb, and Zn concentrations in water and streambed sediment (Table 6). Each metric was correlated with four or more of the explanatory variables associated with mining. The number of coldwater native species, the number of cottid age-classes, percent cottids, CPUE, and cottid biomass were negatively correlated with mine density and metal concentrations in water and streambed sediment. Some of the highest negative correlation coefficients were associated with the cottid metrics and the explanatory variables related to mining. Percent introduced individuals and percent salmonids were positively correlated with mine density and metal concentrations in water and streambed sediment.

Percent cottids and percent salmonids were plotted against dissolved Zn in water for all sites (Figure 6). The results illustrate the response of these metrics over a gradient of concentrations from less than the reporting limit (1.0 μg/L) to levels exceeding the acute AWQC by just over 1 (site 7) to 25 times (site 13). A strong inverse relation was evident between percent cottids and percent salmonids for all sites. These metrics were not significantly related only for those sites where Zn concentrations were below the AWQC. These results indicate that these two metrics have a predictable relation only when Zn toxicity becomes an overwhelming limiting factor. Generally, cottids composed most of the assemblage at the reference sites (about 70 to 90% of all fish), with the exception of sites 4 and 10 on the North Fork

TABLE 6.—Spearman rank correlation coefficients between selected fish metrics and (1) production mine densities in 500-m buffers/km^2 and (2) concentrations of Cd, Pb, and Zn in water and streambed sediment for all 18 sampling sites in the Coeur d'Alene and St. Regis river basins in 2000; $P < 0.05$*, $P < 0.01$**.

Metric	Production mines	Water (dissolved; μg/L)			Stream sediment (μg/g dry weight)		
		Cd	Pb	Zn	Cd	Pb	Zn
Number of coldwater native species	−0.68**	−0.51*	−0.43	−0.76**	−0.73**	−0.72**	−0.72**
Number of cottid age-classes	−0.77**	−0.64**	−0.54*	−0.78**	−0.82**	−0.84**	−0.83**
Percent introduced individuals	0.51*	0.23	0.07	0.55*	0.49*	0.45	0.51*
Percent salmonids	0.56*	0.14	0.01	0.48*	0.50*	0.48*	0.48*
Percent cottids	−0.77**	−0.58*	−0.49*	−0.68**	−0.74**	−0.77**	−0.73**
Catch per unit effort[a]	−0.54*	−0.69**	−0.65**	−0.50*	−0.59**	−0.67**	−0.59**
Cottid biomass (g/m^2)	−0.72*	−0.58*	−0.49*	−0.67**	−0.66**	−0.69**	−0.68**

[a] Fish captured per minute of electrofishing.

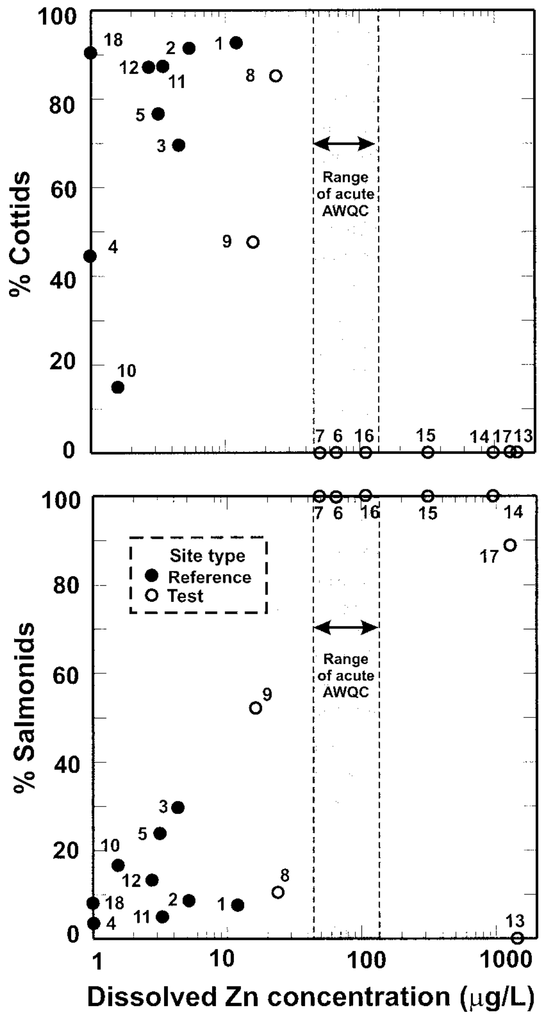

FIGURE 6.—Comparison of the concentration of dissolved Zn in the water for the 18 sampling sites and percent cottids (upper panel) and salmonids (lower panel).. Also shown is the range of the acute Ambient Water Quality Criteria (AWQC; USEPA 2000), which are based on dissolved Zn and the associated hardness concentration.

Coeur d'Alene River. Both of these sites are larger in size than most other sampling sites, and the assemblage included cyprinid and catostomid species that were not present at most other sites. Although sites 8 and 9 were categorized as test sites, their fish assemblages resembled those found at the reference sites.

Discussion

The production mine densities upstream and within a 500-m buffer proved to be useful measures of mining effects on fish assemblages. Maret and Skinner (2000) noted significant relations between production mine densities and metal concentrations in streambed sediment, but mine density has not previously been used to describe watershed disturbance or metal concentrations that may be detrimental to aquatic life. Mine densities in the 500-m buffer area that were 0.2 mines/km^2 or more appeared to relate to elevated metal concentrations and associated differences in fish assemblages. The exceptions were the test sites Prichard Creek at Prichard (site 8) and Beaver Creek near Murray (site 9), which contained fish assemblages more similar to those at reference sites. Even though streambed sediment at these two test sites was enriched in Pb and Zn, concentrations did not exceed AWQC. These results indicate that AML remediation efforts might not affect fish assemblages at these two sites.

Although water temperatures were higher at the test sites, temperatures at all sites were below Idaho's coldwater temperature criterion. Furthermore, Reiser (1999) measured continuous summer water temperatures at numerous sites in the Coeur d'Alene River basin and also concluded that water temperatures were generally not limiting to the trout fishery. Wydoski and Whitney (1979) reported finding shorthead sculpins in waters with temperatures as high as 24°C, which is much higher than temperatures measured during this study. Recent summer temperature studies of minimally disturbed wilderness streams of the Salmon River basin in Idaho have found both shorthead sculpins and cutthroat trout associated with maximum daily water temperatures ranging from 12°C to 23.5°C (D. Ott, USGS, personal communication).

The traditional measures of watershed land use (e.g., percent forest or percent range) were not useful explanatory variables for describing the effects of human activities on fish assemblages. Maret and Skinner (2000) also found no relation between traditional measures of basin land use (e.g., percent forest, percent rangeland, percent agriculture, and percent urban) and metal concentrations in streambed sediment in the Coeur d'Alene River basin.

Instream cover was found to be limited at all sites, with woody debris especially scarce. According to Woods and Beckwith (1997), more than 100 years of logging has resulted in a loss of riparian areas and forest cover along streams in the Coeur d'Alene River basin, and many drainages have not yet recovered. Habitat improvements designed to increase instream cover would benefit the fish assemblages in many of these streams where metal toxicity is not limiting.

Concentrations of Cd, Pb, and Zn at reference sites in the study area were typically below AWQC. However, some enrichment of Pb and Zn in streambed sediment is likely as a result of limited mining and/or natural deposits. Atmospheric transport and deposition may be another potential source of contamination, especially in those areas where prevailing winds historically carried metals from the smelter operation at Bunker Hill.

Comparisons of data between reference and test sites using both upstream-versus-downstream and paired-watershed approaches provided evidence that metals associated with mining have affected the fish assemblages. Zinc is the predominant metal contaminant in both water and streambed sediment. Relations between Zn concentration in water and fish metrics suggest cottids are more sensitive to elevated Zn concentrations in the streams of the study area than are salmonids.

Fish assemblages at sites upstream from intensive mining activities, which include the South Fork near Mullan (site 11) and Canyon Creek near Burke (site 12), comprised predominantly coldwater native cutthroat trout and shorthead sculpin. In contrast, fish assemblages at sites downstream from mining areas comprised fewer fish (with lower CPUE and biomass), no cottids, and, in the most severe case, no fish (Canyon Creek at Woodland Park, site 13).

The introduced brook trout was common at most test and reference sites in the study area. Therefore, metrics that do not include this introduced species, such as percent coldwater native individuals or percent native salmonids, would be more effective for evaluating metal contamination. A measure of the expected fish density or biomass from reference streams, normalized for area sampled, also may be useful, particularly if only native salmonids or cottids are used. Because cottids would be expected to be present in most reference streams of the study area, their presence alone would make a useful screening tool to evaluate where metal contamination might be impairing biotic integrity. Waite and Carpenter (2000) noted the importance of cottids for assessing the biotic integrity of Oregon streams.

Even though shorthead sculpin was the most abundant species collected during this study, it was absent from streams where Zn concentrations exceeded the AWQC. Laumeyer (1976) noted that this species was the most widely distributed species in the Coeur d'Alene River basin except in areas influenced by mining. McCormick et al. (1994) noted the absence of Paiute sculpin *Cottus beldingi* near the mining areas of the Eagle River, Colorado, but identified no toxicological reason for their absence. Mebane (2001) determined that cottid age-classes declined with increasing fine sediment in Idaho streams but did not evaluate mining variables. Benthic invertivores such as cottids (which live on or near the stream bottom and feed predominantly on benthic invertebrates) are more likely to come in contact with contaminated bed sediment and to receive a more sustained dietary exposure to metals than would the more mobile salmonids. Alternatively, behavioral avoidance by emigration during periods of high metal concentration, combined with recolonization rates, may account for the presence of salmonids and the absence of cottids in water where metal concentrations exceed the AWQC. The greater mobility of cutthroat trout relative to that of shorthead sculpin could help explain this difference. According to Hilderbrand and Kershner (2000), cutthroat trout often move seasonally 1–8 km, whereas shorthead sculpin are more sedentary and display movement patterns that are generally more restricted (Gasser et al. 1981). Also, the ability of salmonids to migrate into uncontaminated tributaries to spawn protects the early life stages by reducing or eliminating the exposure to elevated metals in water and streambed sediment. In contrast, the nonmigratory cottids could not protect their eggs or larvae with this strategy. Gagen et al. (1993) found that even though the mottled sculpin *C. bairdi* was as tolerant as brook trout of low pH and high aluminum concentrations in laboratory tests, it was more restricted in its distribution than brook trout. These investigators suggested that this difference may be due to the inability of mottled sculpin to avoid toxic acidic episodes or that the early life stages of cottids may be more sensitive than those of brook trout.

Windward Environmental (2001), representing the Idaho Department of Environmental Quality, has proposed site-specific water quality criteria for the South Fork Coeur d'Alene River on the basis of laboratory toxicity tests. Laboratory toxicity testing of numerous native fish and macroinvertebrate species indicated that cutthroat trout (<30-d-old) fry were most sensitive to Zn. Windward Environmental also determined that cutthroat trout was about four to five times more sensitive to Zn ($LC50$ of 196–277 μg/L) than shorthead sculpin ($LC50$ of $>$ 1,068 μg/L) at water hardness concentrations similar to those found in the South

Fork Coeur d'Alene River. These toxicity tests used 1- to 2-year-old shorthead sculpin (30–60 mm total length), which may not be the most sensitive life stage. Early life stages of salmonids are often more sensitive to Zn than adults (USEPA 1987). It is likely that if young-of-the-year shorthead sculpin were tested, they would be more sensitive than older fish to Zn. Because these laboratory tests were also performed in an aquarium without contaminated bed sediment, they failed to simulate the potentially harmful exposure of benthic fish (such as shorthead sculpin) to Zn. The CTUs in sediments indicated that at seven test sites potentially toxic conditions exist for aquatic life, especially for sculpins and benthic invertebrates living in close proximity to these streambed sediments.

Our findings indicate that conclusions derived primarily from laboratory tests should be viewed with caution until they can be verified and supported with field data. Contrary to site-specific laboratory toxicity test results, our field studies suggest cutthroat trout are not affected as strongly as shorthead sculpin. Cutthroat trout were present at test sites where concentrations of Zn were as much as 12 times higher than the acute AWQC. Shorthead sculpins were absent at these same test sites as a result of metal enrichment from hard-rock mining, not physical habitat degradation such as sedimentation. Reiser (1999) noted an upstream-versus-downstream change in fish assemblages in the South Fork Coeur d'Alene River; stream reaches affected by mining downstream from the Canyon Creek confluence contained depressed populations of cottids and mountain whitefish.

The increased resistance to metals shown by salmonids at test sites may be the result of acclimation to acutely toxic metal concentrations (Stubblefield et al. 1999). However, this acclimation may have more long-term consequences on the fish that are able to survive in an environment where metal concentrations are above the AWQC. According to Farag et al. (1999), a metabolic cost may exist for these acclimated survivors because biochemical changes occur in their bodies (e.g., increased metallothionein production), which help to minimize the effects of metals on their systems and could cause reduced fitness, less reproductive success, and reduced immune defenses.

Our study results provide evidence that additional research is warranted on shorthead sculpin life history and toxicity testing by using the early life stages of shorthead sculpin. The findings of this field study suggest that the shorthead sculpin is more responsive to metals in field conditions than are cutthroat or brook trout individuals and may prove to be useful indicators for evaluating hard-rock mining remediation and restoration efforts in other areas of the West. The additional benefits of using cottids as water quality indicators are that they are not as mobile as salmonids, are not stocked, and are seldom harvested.

Biomonitoring is being used more frequently as a method to assess stream health, and fish assemblage information is a key component of these assessments. The three cottid metrics used in this evaluation discriminated between test and reference sites. Mebane (2001) also found cottid abundance and age-classes to be useful metrics for evaluating stream conditions in Idaho. The number of coldwater native species and abundance of all fish species (CPUE) were also useful metrics for evaluating stream conditions because both metrics declined as metal concentrations increased. By contrast, percent introduced individuals and percent salmonids increased as a result of increased metal concentrations associated with mining. Even if salmonids are present at sites affected by mining, the food supply of aquatic macroinvertebrates may not be adequate to support a viable population. A reduction in the abundance and diversity of macroinvertebrates has been shown to be related to elevated metal concentrations associated with mining (Clements et al. 2000; Mebane 2001). In addition, Farag et al. (1999) reported that fish growth and survival were reduced when the fish were fed macroinvertebrates from the South Fork Coeur d'Alene River. They attributed these adverse effects to elevated metal concentrations in the tissue of the consumed macroinvertebrates. Future biomonitoring would benefit from an evaluation of the availability and quality of macroinvertebrates in riffle habitats to ensure that they are adequate to support a sustainable fishery.

In summary, this study provides a better understanding of the environmental factors affecting fish assemblages in the Coeur d'Alene River basin. The weight of evidence provided in this study demonstrate that streams downstream from areas of intensive hard-rock mining contained fewer native fish and lower total fish abundances as a result of metal enrichment in both water and streambed sediment. Although physical habitat degradation did not appear to be a significant factor (on the basis of the relative absence of fish cover at most sites), habitat enhancement likely would benefit most stream fisheries throughout the Coeur d'Alene River basin if metal concentrations were not limiting. Production mine densities near streams were a useful measure of mining intensity. Zinc was the contaminant most clearly associated with affected fish assemblages; concentrations exceeded AWQC and the PEL most often in streambed sediment. Contrary to findings from recent toxicity tests used to develop site-specific Coeur d'Alene metals criteria for the South Fork River, our field studies suggest that shorthead sculpin are more affected by elevated Zn concentrations than are cutthroat or brook trout. Further field studies and toxicity testing on the early life stages of shorthead sculpins are needed to determine the causes for their disappearance in streams affected by elevated concentrations of metals associated with hard-rock mining. If such studies were designed to more closely mimic natural conditions, the exposure risk of aquatic organisms to metals toxicity could be more accurately evaluated.

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