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Chapter 18

SELENIUM AND OTHER TRACE ELEMENTS IN WATER, SEDIMENT, AQUATIC PLANTS, AQUATIC INVERTEBRATES, AND FISH FROM STREAMS IN SE IDAHO NEAR PHOSPHATE MINING

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ABSTRACT

Nine stream sites in the Blackfoot River watershed in SE Idaho were sampled in June 2000 for water, surficial sediment, aquatic plants, aquatic invertebrates, and fish. Selenium (Se) and other elements were measured in these aquatic ecosystem components and a hazard assessment was performed on the data. Water quality characteristics were relatively uniform among the nine sites examined. Of the aquatic components assessed, water was the least informative, especially its analysis for Se contamination because measured concentrations were substantially below the national water quality criterion of 5 $\mu g L^{-1}$. In contrast, Se and several other elements were elevated in sediment, aquatic plants, and aquatic invertebrates from several sites, indicating accumulation in sediments and cycling through plants and invertebrates. Only Se in fish was elevated to concentrations of potential concern. A hazard assessment of Se in the aquatic environment suggests low hazard at Trail Creek and Sheep Creek, moderate hazard at upper Slug Creek and lower Slug Creek, and high hazard at Angus Creek near the mouth, upper East Mill Creek, lower East Mill Creek, Dry Valley Creek, and lower Blackfoot River.

INTRODUCTION

Phosphorus is present in economically mineable quantities in organic-rich black shales of the Permian Phosphoria Formation, which constitutes the Western Phosphate Field. There are four active open pit mines in the SE Idaho Phosphate District that produce phosphate from the Meade Peak Phosphatic Shale Member, and 10 inactive mines (Montgomery Watson, 1999; Jasinski et al., Chapter 3). Mining is by open-pit surface mining and waste materials are generally disposed of on the surface in tailing piles, ponds, landfills, and dumps. Many of the waste-rock piles have drainage systems designed to move surface water and groundwater away from waste piles where it reaches other surface

waters eventually draining into tributaries and finally the Blackfoot River and Blackfoot Reservoir. Thus, water movement distributes potentially toxic trace elements to aquatic and terrestrial ecosystems.

The Blackfoot River watershed has several active and inactive phosphate mines that potentially adversely affect aquatic resources in several tributaries of the Blackfoot River (see Fig. 16-2, Chapter 16). As early as 1970–1976, concerns were expressed about contamination of the Blackfoot River and its tributaries by elements released from phosphate mining (Platts and Martin, 1978). Recent concerns have been the subject of several reports (Montgomery Watson, 1999, 2000, 2001a,b). Several investigations by the US Geological Survey (USGS) reported the chemical composition of weathered and less weathered strata of the Meade Peak Phosphatic Shale (Desborough et al., 1999; Herring et al., 2000a,b). Other USGS investigations reported trace element concentrations in aquatic bryophytes and terrestrial plants that were influenced by mining activities (Herring and Amacher, 2001; Herring et al., 2001).

Release of toxic trace elements from waste-rock piles and accumulation in the food chain has resulted in adverse biological effects. In recent years, five of six horses in a pasture in Dry Valley were euthanized due to Se toxicosis; two horses in the Woddall area were put down due to Se poisoning, and 60–80 sheep were found dead on the old Stauffer Mine site (within the Caribou National Forest) due to Se poisoning (Caribou County Sun, 1999). Twenty-six of the sheep were found at the south end of Rasmussen Ridge Mine near a spring or seep at a waste-rock pile. Elevated concentrations of Se and other trace elements have been reported in limited samples of fish fillets and aquatic invertebrates (Montgomery Watson, 1999). A recent report suggested that Se concentrations in fish and wildlife were sufficiently elevated to cause adverse effects in sensitive fish species (Piper et al., 2000).

The purpose of this study was to determine the concentrations of Se and other trace elements in water, surficial sediment, aquatic plants, aquatic invertebrates, and fish from streams in SE Idaho near phosphate mining. This information was used in a hazard assessment of the potential effects of Se and other elements on aquatic resources in areas of the Blackfoot River watershed that are potentially impacted by phosphate mining.

METHODS AND MATERIALS

Samples of water, surficial sediment, aquatic plants, aquatic invertebrates, and fish were collected in June 2000 from nine sites in the Blackfoot River watershed located in SE Idaho.

Collection site description

1. The Angus Creek (ACM) site was located at the crossing of the creek by Forest Route 095, approximately 0.5 km above the confluence with the Blackfoot River (see Table 18-I for coordinates of all sampling sites). The land on either side of the road is

TABLE 18-I

Latitude and longitude of nine sites sampled in southeast Idaho near Soda Springs keyed to Fig.18-1

Site name and ID	Location[1]	
Angus Creek near mouth (ACM)	N42°49′42.24″ W111°20′15.04″	±11 m
Upper East Mill Creek (UEMC)	N42°48′26.74″ W111°18′38.82″	±14 m
Lower East Mill Creek (LEMC)	N42°48′53.56″ W111°18′25.98″	±11 m
Trail Creek (TC)	N42°45′29.82″ W111°26′47.53″	±8 m
Upper Slug Creek (USC)	N42°37′50.85″ W111°18′20.51″	±14 m
Lower Slug Creek (LSC)	N42°42′23.79″ W111°22′03.89″	±9 m
Sheep Creek (ShpC)	N42°51′46.92″ W111°20′00.53″	±18 m
Dry Valley Creek (DVC)	N42°46′58.71″ W111°22′26.47″	±14 m
Lower Blackfoot River (LBR)	N42°49′12.49″ W111°33′09.33″	±10 m

[1]Global positioning system (courtesy of Phil Moyle, US Geological Survey).

managed by the Idaho Department of Fish and Game and primarily was grassland habitat with sparse forbs and very limited grazing activity (see Fig. 18-1).

2. The upper East Mill Creek (UEMC) site was located near an unmaintained dirt road (Forest Route 309) about 3.2 km from the intersection of Forest Route 102 and Forest Route 309, about 8 km above the confluence with the Blackfoot River. The land is managed by the US Forest Service and was pine forest with some sagebrush in open areas. Sample collection was in a generally open area of forbs, grass, and sparse pine trees with very limited grazing activity.
3. The lower East Mill Creek (LEMC) site was located at the crossing of the creek by Forest Route 102, approximately 4 km below the upper East Mill Creek site and about 4 km above the confluence with the Blackfoot River. The site is on private land (Bear Lake Grazing Company) accessed by landowner permission (Ms. Joan Bunderson). The land on either side of the road supported grass and sagebrush with moderate grazing. Sample collection was about equally distributed upstream and downstream of the road crossing.

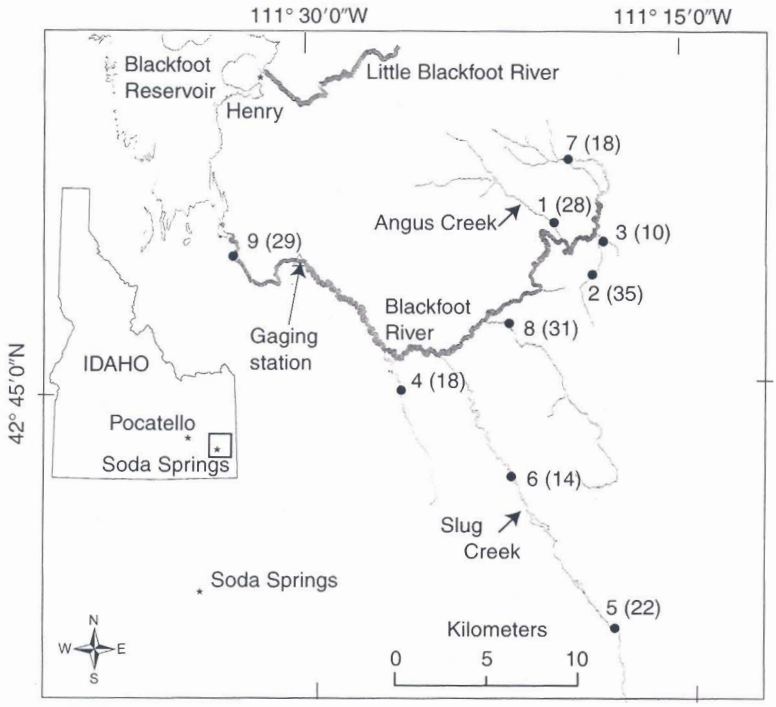

Fig. 18-1. Map of sample sites: 1, Angus Creek; 2, upper East Mill Creek; 3, lower East Mill Creek; 4, Trail Creek; 5, upper Slug Creek; 6, lower Slug Creek; 7, Sheep Creek; 8, Dry Valley Creek; 9, lower Blackfoot River. Numbers in parenthesis are total hazard assessment scores from Table 18-XV; modified from Presser et al., Chapter 16.

4. The Trail Creek (TC) site was located at the crossing of the creek by Trail Creek Road (Forest Route 124), about 8 km above the confluence with the Blackfoot River. The site is on private land accessed by landowner permission (Mr. Val Bloxham). The land on either side of the road was grazed grassland. Sample collection was primarily downstream of the crossing in an area with light grazing, whereas the upstream side of the crossing had moderate to heavy grazing.
5. The upper Slug Creek (USC) site was located about 2 km inside the US Forest Service boundary on Slug Creek Road (Forest Route 095), about 0.4 km off the gravel road that parallels the stream. The site was located above the influence of mining activities and is thus the reference site. The US Forest Service manages the land. The vegetation around the stream was primarily willow-type shrubs with sparse grass, with a substantial amount of sagebrush and quaking aspen nearby. The site had light grazing.
6. The lower Slug Creek (LSC) site was located at the intersection of Slug Creek Road (Forest Route 095) and Old Mill Road (Forest Route 124). The site is in the road right-of-way and about 10 km above the confluence with the Blackfoot River. The land

around the stream is primarily grassland and was heavily grazed in one area downstream and lightly grazed in two other areas upstream. Sample collection was upstream of the heavily grazed area in a stream section with little grazing.

7. The Sheep Creek (ShpC) site was located on private land (Sheep Creek Guest Ranch) about 3 km upstream of the crossing of the creek by Forest Route 095. The site was accessed with landowner permission (Mr. Phil Baker). The land along the stream was primarily pine forest with some forbs, shrubs, and grass. Sample collection for water, sediment, and fish was upstream of most animal and human activity at the end of a private road and about 2 km above Lanes Creek, which flows into the Blackfoot River about 4 km downstream. Aquatic plants and one fish species were collected about 1.5 km downstream of the primary sample collection site and near the stream crossing with Forest Route 095 in an area of sagebrush and grass with moderate grazing.
8. The Dry Valley Creek (DVC) site was located on private land (Hunsacker Ranch) 0.5 km from Forest Route 122 and accessed along a railroad track that parallels the creek. The site is about 0.75 km above the confluence with the Blackfoot River. The land was accessed with landowner permission (Mr. and Mrs. Keith Hunsacker). The land along the stream was primarily grassland with some shrubs nearby and moderate grazing.
9. The lower Blackfoot River (LBR) site was located on county parkland on the upstream side of a large steel and concrete bridge located on Blackfoot River Road about 0.5 km east of State Highway 34. The site was located about 1 km above the confluence with Blackfoot Reservoir. The land was accessed by landowner permission (Caribou County Commissioner Carol Davids-Moore). The land along the river was primarily riparian with some grass and light camping activity.

Sample collection

Samples of water, surficial sediment, aquatic plants, aquatic invertebrates, and fish were collected at each of the nine stream sites. Water sample bottles were conditioned by immersion in site water three times. Sample container conditioning and water sample collection followed US Geological Survey procedures. Water samples were collected using depth-integrated sampling techniques. For each sample site, one set of samples was unfiltered and a second set filtered through a 0.45 μm polycarbonate filter. One filtered water sample was collected for measurement of major cations and anions. A 200 mL sample of filtered water was collected in an acid-cleaned polyethylene bottle for analysis of Se concentrations and a second bottle was collected for trace element anaysis concentrations (hereafter referred to as other elements). A reagent deionized-water blank was also collected. Water samples for Se analysis were acidified with ultrapure HCl and those for other elements were acidified with ultrapure HNO_3. All samples were stored frozen.

Sediment was collected using a plastic scoop to acquire surficial sediments including organic detritus (hereafter called detritus), but not pebbles or plant material. The scoop and

sample containers were rinsed in ambient water for sufficient time to condition the equipment to ambient conditions prior to sample collection. After sediments had settled, excess water was removed and the sample stored frozen. One sample was analyzed for Se and Hg and a second sample for other elements.

Submerged aquatic plants (white-water buttercup, *Ranunculus longirostris*) were collected from each site by hand. The sample consisted of leaf whorls removed from stems using plastic or stainless steel forceps. Two sets of plant samples were collected from each site, squeezed to remove excess water, weighed, bagged in Whirl-Pak bags, labeled, and stored frozen. One composite set was analyzed for Se concentrations and the other set analyzed for other element concentrations.

Aquatic invertebrates were sieved from bed substrate materials collected either by D-frame kick nets or by removing large stones with attached invertebrates. Substrate was placed in large polypropylene trays and invertebrates separated from substrate using forceps or glass tubes with suction bulbs. No aquatic invertebrates were collected from lower East Mill Creek (due to their paucity) or lower Slug Creek (not sampled). Two sets of invertebrate samples were collected from each site, separated by taxa group, weighed, bagged in Whirl-Pak bags, labeled, and stored frozen. Each composite invertebrate sample contained about half the available taxa weight. One composite set was analyzed for Se concentrations and the other set analyzed for other element concentrations. An opportunistic sample of crane fly nymphs (Tipulidae) was collected from Angus Creek headwaters at the base of Wooley Valley mine Unit 4 waste pile and analyzed like the other invertebrates.

Fish were collected by electrofishing using a Coffet Mark-10 electroshocker. The equipment was rinsed in ambient water for sufficient time to condition the equipment to ambient conditions. Two sets of fish samples were collected from each site, euthanized with MS-222 (tricaine methanesulfonate), identified to species, measured for total length and weight, bagged in Whirl-Pak bags, and stored frozen. No fish were collected from lower East Mill Creek in spite of substantial effort. One set of fish was analyzed for Se concentrations in whole body and the other set analyzed for other element concentrations. The species collected included cutthroat trout (*Oncorhynchus clarki*), brook trout (*Salvelinus fontinalis*), mottled sculpin (*Cottus bairdi*), longnose dace (*Rhinichthys cataractae*), speckled dace (*Rhinichthys osculus*), and redside shiner (*Richardsonius balteatus*).

Water quality analyses and flow measurement

Ambient-water samples collected at each study site were analyzed for general water quality characteristics according to standard methods (American Public Health Association and other associations, 1995). Site water was analyzed *in situ* for conductivity, pH, temperature, and dissolved oxygen concentration.

Immediately after arrival of water samples at the mobile laboratory, conductivity, pH, alkalinity, hardness, Ca, Mg, and temperature were determined. A subsample of 200 mL was collected and stored at 4°C with no preservative, and transported to USGS Laboratory

in Yankton for analysis of sulfate and chloride. A second subsample of 125 mL was collected, acidified with 0.5 mL concentrated H_2SO_4, and transported to Yankton for analysis of ammonia concentrations. Ammonia was measured using ion-selective electrodes following the procedures for low concentration measurements of the electrode manufacturer (Orion Research, 1990, 1991; Analytical Technology Incorporated Orion, 1994). Chloride was measured by the mercuric nitrate titration method (Hach Company, 1997).

Element analysis

Water, surficial sediment, aquatic plants, aquatic invertebrates, and fish were analyzed for Se concentrations by atomic absorption spectroscopy-hydride generation (AA-HG; Environmental Trace Substances Laboratory (ETSL), University of Missouri). Analyses incorporated appropriate quality assurance/quality control (QA/QC) procedures such as standardizing equipment with certified reference material, determination of limit of detection, analysis of reagent blanks, spiked samples (termed % recovery of digested spike), duplicate analysis samples (termed % relative standard deviation, RSD), certified reference materials, reference materials spiked before digestion (termed % recovery of reference material), and reference material spiked after digestion (termed % recovery of reference standard). Analysis for Se concentrations was based on Method 7000 of the US Environmental Protection Agency (1983), and results are reported on a dry weight basis.

A subsample of sediment was digested using a four-acid procedure and analyzed for concentrations of Se by AA-HG and Hg by cold vapor AA (USGS lab, Denver). All analyses incorporated QA/QC including standardizing analytical instruments using certified reference materials, determination of the limit of detection, and analysis of certified reference standards. Results are reported on a dry weight basis.

Water, surficial sediment, aquatic plant, aquatic invertebrate, and fish samples were analyzed for other element concentrations by inductively coupled argon plasma atomic emission spectrophotometry (ICP) for Al, As, Ba, Be, B, Cd, Cr, Cu, Fe, Pb, Mg, Mn, Mo, Ni, Sr, V, and Zn. Analyses were conducted by ETSL with QA/QC as described above. Analysis was based on Method 6010 of the US Environmental Protection Agency (1983) and results are reported on a dry weight basis.

A subsample of sediment was digested using a four-acid procedure and analyzed by ICP by the USGS lab, Denver, for Ca, Mg, P, K, Na, Al, Sb, As, Ba, Be, Bi, Cd, Ce, Cs, Cr, Co, Cu, Ga, Fe, La, Pb, Li, Mn, Mo, Ni, Nb, Rb, Sc, Ag, Sr, Tl, Th, Ti, U, V, Y, and Zn, with QA/QC as listed above.

Statistical analyses

Data were analyzed using computer programs (Statistical Analysis System, Inc., 1985) to determine the relation between various measures made during the study. The Pearson correlation coefficients were determined for the relations between water-quality characteristics

and Se concentrations in water, sediments, aquatic plants, aquatic invertebrates, and fish. For fish residue data for each sample location, the geometric mean was used in the correlation analyses with other variables.

RESULTS AND DISCUSSION

Quality assurance/quality control of chemical analyses

In general, the limit of detection (LOD) was 0.2 $\mu g\,g^{-1}$ for Se in surficial sediment, aquatic plant, aquatic invertebrate, and fish matrices and 0.5 $\mu g\,L^{-1}$ for water (see CD, Appendix 2, Table A). The procedure blank had a concentration less than the LOD. The percent relative standard deviation (duplicate preparation and analysis) ranged from <LOD to 6.2%, indicating consistent sample handling during preparation, digestion, and analysis. Percent recovery of Se from certified material ranged from 81 to 94%, indicating the digestion and analysis procedure accurately measured Se concentrations. Percent recovery of Se from samples spiked before digestion ranged from 85 to 103%, indicating no loss of Se during digestion. Percent recovery of Se spiked in reference material before digestion ranged from 88 to 96%, indicating that the digestion procedure did not alter the amount of spiked Se in the reference material. Percent recovery of Se spiked in reference material after digestion ranged from 94 to 96%, which indicated little interference by other elements during the Se analysis.

For determination of other elements in water, sediment, aquatic plants, aquatic invertebrates, and fish by ICP, the LOD, procedural blanks, relative standard deviation of duplicate preparation and analysis, and spike recoveries were comparable to those for Se (see CD, Appendix 2, Table B).

The QA/QC measures for the determination of Hg by cold vapor AA and Se by AA-HG in sediments at the USGS lab involved analysis of three reference materials (see CD, Appendix 2, Table C). The recovery of Hg from reference materials ranged from 82 to 102% and for Se from 107 to 109%.

For the measurement of elements in sediments at the USGS lab, the LOD, percent relative standard deviation for duplicate preparation and analysis, and percent recovery of elements in standard reference materials were comparable to those from the ETSL (see CD, Appendix 2, Table C).

The measurements for pH, conductivity, and alkalinity for water quality were comparable between the Mobile and Yankton USGS labs (Table 18-II, only Yankton data provided), with correlations for pH of 0.77 ($P = 0.02$), conductivity of 0.99 ($P < 0.0001$), and alkalinity of 0.99 ($P < 0.0001$), respectively. Dry Valley Creek had the highest conductivity, hardness, Ca, Mg, chloride, and sulfate concentrations of the nine sites examined. The other stream sites had similar water-quality characteristics. The nine sites were well oxygenated (Table 18-III) and characterized by near-neutral pH and moderately low hardness and alkalinity. In general, water quality seemed suitable for all types of aquatic life including algae, plants, zooplankton, benthic organisms, and fish.

TABLE 18.II

Water-quality characteristics from nine locations listed in Table 18-I; number of samples in brackets; for $n > 1$, water quality measure is the mean; data from USGS Yankton, SD lab and USGS Mobile, ID lab

Measure	ACM	UEMC	LEMC	TC	USC	LSC	DVC	ShpC	LBR
pH	8.2 [2]	8.1 [1]	8.3 [1]	8.3 [2]	8.1 [1]	8.2 [1]	7.9 [2]	8.2 [1]	7.9 [2]
Conductivity (μmhos cm^{-1})	393 [1]	370 [1]	367 [1]	380 [1]	398 [1]	402 [1]	611 [1]	371 [1]	412 [1]
Hardness (mg L^{-1} as a CO_3)	198 [2]	187 [1]	188 [1]	188 [2]	202 [1]	204 [1]	284 [2]	191 [1]	212 [2]
Calcium (mg L^{-1})	61 [2]	55 [1]	55 [1]	57 [2]	59 [1]	59 [1]	86 [2]	58 [1]	59 [2]
Magnesium (mg L^{-1})	11 [2]	12 [1]	12 [1]	11 [2]	13 [1]	14 [1]	17 [2]	11 [1]	16 [2]
Alkalinity (mg L^{-1} as a CO_3)	170 [2]	178 [1]	179 [1]	183 [2]	206 [1]	198 [1]	186 [2]	182 [1]	204 [2]
Chloride (mg L^{-1})	4.9 [1]	2.0 [1]	1.9 [1]	6.1 [1]	3.5 [1]	5.3 [1]	8.8 [1]	1.5 [1]	2.5 [1]
Sulfate (mg L^{-1})	27.5 [1]	13.2 [1]	13.0 [1]	7.1 [1]	<5.0 [1]	9.9 [1]	96 [1]	12.8 [1]	10.2 [2]
Un-ionized ammonia (mg L^{-1} NH_3-N)	0.001 [1]	<0.001 [1]	<0.001 [1]	<0.001 [1]	<0.001 [1]	0.001 [2]	<0.001 [1]	0.001 [1]	<0.001 [1]
Total ammonia (mg L^{-1} as N)	0.02 [1]	0.02 [1]	<0.01 [1]	0.01 [1]	0.01 [1]	0.02 [2]	<0.01 [1]	0.03 [1]	<0.01 [1]

Water

Selenium

Selenium concentrations in water from seven sites were less than the LOD (< 0.5 μg L^{-1}) (Table 18-IV). However, Se in water at upper East Mill Creek, 30 μg L^{-1}, and lower East Mill Creek, 15 μg L^{-1}, was substantially higher than the national water-quality criterion for the protection of aquatic life of 5 μg L^{-1} (US Environmental Protection Agency, 1987).

Controversy exists as to the best material (water, biota, or sediment) to use as a national criterion for Se (US Environmental Protection Agency, 1998b). A recent peer consultation

TABLE 18-III

Water-quality characteristics measured *in situ* and in the USGS Mobile, ID lab at the nine locations listed in Table 18-I

Measure	ACM	UEMC	LEMC	TC	USC	LSC	DVC	ShpC	LBR
pH	8.4	8.3	8.4	8.3	8.2	8.1	8.1	8.3	7.8
Conductivity (μmhos cm^{-1})	380	357	353	362	388	399	548	359	407
Bicarbonate (mg L^{-1})	198	210	206	215	244	234	227	214	256
Carbonate (mg L^{-1})	4.3	4.3	5.3	4.8	4.3	2.4	0	5.3	0
Dissolved oxygen (mg L^{-1})	8.5	9.1	8.7	10.9	9.3	10.6	10.2	9.6	9.1
% Saturation dissolved oxygen	93	84	85	102	86	111	111	87	98
Water temperature (°C)	20	12	23	12	12	22	19	11	19
Discharge (cfs)	1.63	1.48	0.78	3.08	0.92	2.96	0.80	5.03	87.5

TABLE 18-IV

Se concentrations for water (μg L^{-1}) and for sediment, aquatic plants, and aquatic invertebrates (μg g^{-1} dry weight); collected from nine locations listed in Table 18-I and site ACH[1]

Matrix	ACH	ACM	UEMC	LEMC	TC	USC	LSC	DVC	ShpC	LBR
Water	NS[2]	<0.5	29.9	14.9	<0.5	<0.5	<0.5	<0.5	<0.5	<0.5
Sediment	NS	1.0	32.5	45.8	0.8	0.8	1.7	5.2	0.5	2.1
Aquatic plant	NS	0.9	30.2	74.1	0.8	1.1	1.5	3.8	0.6	4.5
Aquatic invertebrate	186	7.2	35.7	NS	2.9	4.9	NS	19.5	2.6	5.4

($n = 1$). Analyses from ESTL.

[1]ACH: crane fly nymphs collection site at Angus Creek headwaters.

[2]NS: not sampled.

workshop on Se aquatic toxicity and bioaccumulation discussed the technical issues underlying the freshwater aquatic life chronic criterion for Se, and concluded that water was a poor choice (US Environmental Protection Agency, 1998b). The growing body of Se literature, which emphasizes the toxicological importance of the dietary route of Se exposure, requires a reassessment of the continued use of a water-based criterion (Hamilton, 2002).

Other elements

Concentrations of other elements in water were comparable among the nine sites (Table 18-V). Although upper and lower East Mill Creek had elevated Se concentrations, those waters were not among the highest in other element concentrations, except Sr. Dry Valley Creek, relative to the other eight sites, had elevated Al, Fe, Mg, and Mn. However, no element concentrations in water seemed elevated to concentrations of concern.

Concentrations of elements are generally the basis of water quality standards issued by the US Environmental Protection Agency (1998a, 1999). However, investigations indicated that dietary routes of exposures of elements were important in discerning effects on biota

TABLE 18-V

Element concentrations ($\mu g\,L^{-1}$; $n = 1$) in water from nine locations listed in Table 18-I

Element	ACM	UEMC	LEMC	TC	USC	LSC	DVC	ShpC	LBR
Al	46	54	26	42	50	20	89	63	98
As	14	13	11	11	16	12	13	11	11
Ba	37	24	26	56	67	39	42	60	54
Be	<0.8	<0.8	<0.8	<0.8	<0.8	<0.8	<0.8	<0.8	<0.8
B	48	48	37	43	46	40	51	47	48
Cd	<3	<3	<3	<3	<3	<3	<3	<3	<3
Cr	8	7	8	7	8	8	7	8	8
Cu	<2	<2	<2	<2	<2	<2	12	<2	<2
Fe	103	89	90	92	106	94	164	90	106
Pb	<10	<10	<10	<10	<10	<10	<10	<10	<10
Mg	12,900	14,000	13,200	11,200	13,000	13,000	17,300	12,700	16,600
Mn	86	4	5	79	28	31	70	10	42
Mo	<5	<5	<5	<5	<5	<5	<5	<5	<5
Ni	12	9	9	9	9	18	13	9	9
Sr	199	245	234	150	136	230	203	111	208
V	<2	<2	<2	<2	<2	<2	<2	<2	<2
Zn	<5	<5	<5	<5	<5	<5	<5	<5	<5

Analyses from ESTL.

(reviewed in Hamilton and Hoffman, 2003). For example, Kiffney and Clements (1993) reported that monitoring concentrations of Cd, Cu, and Zn in aquatic invertebrates was a better indicator of element bioavailability in the Arkansas River of Colorado, which was impacted by acid-mine drainage, than element concentrations in water.

Comparison to other Idaho data

The Idaho Mining Association Selenium Subcommittee (Selenium Subcommittee) investigated concentrations of Se, Cd, Mn, Ni, V, and Zn in water from numerous sites in the Southeastern Idaho Phosphate Resource Area and concluded that Se was the major contaminant of potential concern (Montgomery Watson, 1999). In May 1998, Se concentrations in water at 12 of 37 stream sites exceeded the US Environmental Protection Agency criterion of 5 $\mu g L^{-1}$, whereas in September 1998 only one stream, East Mill Creek (32 $\mu g L^{-1}$), exceeded the criterion (Montgomery Watson, 1999). In the May 1998 sampling, the stream sites exceeding the criterion included five on the Blackfoot River (5–12 $\mu g L^{-1}$), Trail Creek (8.7 $\mu g L^{-1}$), Dry Valley Creek (5.6 $\mu g L^{-1}$), and two on East Mill Creek (210 and 260 $\mu g L^{-1}$). The values reported by Montgomery Watson (1999) were higher than those measured here.

Montgomery Watson (2000) continued measuring Se concentrations in waters of the Blackfoot River in 1999. Concentrations in the lower Blackfoot River near our sampling site were 6.7 $\mu g L^{-1}$ in May, 2.1 $\mu g L^{-1}$ in June, 2.4 $\mu g L^{-1}$ in July, and 1.5 $\mu g L^{-1}$ in August, which shows the variability over time that can occur in the river (see Presser et al., Chapter 16). Montgomery Watson (2000) reported similar variability in Dry Valley Creek: 49 $\mu g L^{-1}$ in May, 6.8 $\mu g L^{-1}$ in June, 2.7 $\mu g L^{-1}$ in July, and 1 $\mu g L^{-1}$ in August. In May 1999, Dry Valley Creek (49 $\mu g L^{-1}$) and Spring Creek (46 $\mu g L^{-1}$) were major Se contributors to the Blackfoot River. Above Spring Creek the Blackfoot River had < 1 $\mu g L^{-1}$, whereas below Spring Creek concentrations from upstream to downstream were 9.0, 7.4 (above Dry Valley Creek), 7.9 (below Dry Valley Creek), 9.8, 7.2, 8.2, and 6.7 $\mu g L^{-1}$. Thus, substantial contamination of the Blackfoot River occurred during 1999. This contamination was evidenced in Se concentrations in sediment, aquatic plants, aquatic invertebrates, and fish measured here.

Montgomery Watson (2001a,b) reported additional Se concentrations in water sampled in September 1999 and May 2000. Most water samples in September 1999 – April 2000 contained < 5 $\mu g L^{-1}$, except for Dry Valley Creek, which contained 12 $\mu g L^{-1}$ above the Blackfoot River, 270 $\mu g L^{-1}$ downstream of Maybe Creek, and 120 $\mu g L^{-1}$ upstream of Maybe Creek; and East Mill Creek, which contained 19 $\mu g L^{-1}$ (Montgomery Watson, 2001a). In May 2000, Se concentrations of >5 $\mu g L^{-1}$ were reported in the Blackfoot River (5.5–7.1 $\mu g L^{-1}$) and several creeks including State Land Creek (10 $\mu g L^{-1}$) and two tributaries (16 $\mu g L^{-1}$), North Fork Wooley Valley Creek (98 $\mu g L^{-1}$), lower Slug Creek (6.3 $\mu g L^{-1}$), Dry Valley Creek (8–87 $\mu g L^{-1}$), Angus Creek (6.5 $\mu g L^{-1}$), Spring Creek (28 $\mu g L^{-1}$), and East Mill Creek (400 $\mu g L^{-1}$) (Montgomery Watson, 2001b). These data demonstrate continued Se contamination of the Blackfoot River watershed. The data also demonstrate that Se contamination occurs primarily during spring runoff (see Presser et al., Chapter 16).

Sediment

Selenium

Selenium concentrations in surficial sediment were relatively low at five sites (<2 $\mu g\ g^{-1}$), moderately elevated at the Dry Valley Creek and lower Blackfoot River (2.1–5.2 $\mu g\ g^{-1}$), and very elevated at the upper and lower East Mill Creek (33–52 $\mu g\ g^{-1}$) (Tables 18-IV and 18-VI). Sheep Creek and Trail Creek had the lowest Se concentrations in surficial sediment.

Selenium concentrations in surficial sediment from upper Slug Creek, Sheep Creek, and Trail Creek were below the no hazard rating of $<1\ \mu g\ g^{-1}$, and lower Slug Creek and Angus Creek were $<2\ \mu g\ g^{-1}$, the minimal hazard rating of 1–2 $\mu g\ g^{-1}$ proposed by Lemly (1995), and below the no effect concentration of $<2\ \mu g\ g^{-1}$ proposed by Stephens et al. (1997). However, Presser et al. (1994) and Moore et al. (1990) used 0.5 $\mu g\ g^{-1}$ as a reasonable Se concentration in sediment to represent the threshold between uncontaminated background conditions and environments with elevated Se concentrations.

Selenium in surficial sediment from the lower Blackfoot River site (2.1 $\mu g\ g^{-1}$) falls in the concentration-of-concern range of >2–4 $\mu g\ g^{-1}$ of Stephens et al. (1997) and the moderate hazard of 2–4 $\mu g\ g^{-1}$ of Lemly (1995). Selenium in surficial sediment from the Dry Valley Creek (5.2 $\mu g\ g^{-1}$) and upper and lower East Mill Creek (32.5–45.8 $\mu g\ g^{-1}$) sites were higher than the toxicity threshold of $>4\ \mu g\ g^{-1}$ proposed by Stephens et al. (1997) and the high hazard rating of $>4\ \mu g\ g^{-1}$ proposed by Lemly (1995). Elevated Se concentrations in surficial sediment at upper and lower East Mill Creek suggest a substantial contamination concern, as do the elevated Se concentrations in Dry Valley Creek.

The peer consultation workshop on Se aquatic toxicity and bioaccumulation (US Environmental Protection Agency, 1998b) discussed the possibility of a sediment-based criterion. However, they concluded that sediment was a poor criterion because of spatial heterogeneity of Se deposition, variable water retention time, variable volatilization rates, heterogeneity of the benthic phytoplankton community and benthic invertebrates, and variable feeding habitats of higher trophic organisms (US Environmental Protection Agency, 1998b).

Here, the strong correlation for Se concentrations between surficial sediments and aquatic plants (0.96–0.97) and between surficial sediment and aquatic invertebrates (0.92–0.94) indicate ready movement of Se among aquatic ecosystem components and accumulation in the food web (Table 18-VII). Similar movements of Se through the food web were reported in two field studies of seleniferous areas of the upper Colorado River (Hamilton et al., 2001a,b).

Specific to Se, Woock (1984) demonstrated in a cage study with golden shiner (*Notemigonus crysoleucas*) that fish with access to bottom sediments accumulated more Se than fish held in cages suspended about 1.5 m above the sediment. That study revealed that effects in fish were linked to Se exposure via sediment, benthic invertebrates, detritus, or a combination of sediment components.

TABLE 18-VI

Element concentrations ($\mu g\, g^{-1}$ dry weight; $n = 1$) in sediment from nine locations listed in Table 18-I; analyses from USGS lab

Element	ACM	UEMC	LEMC	TC	USC	LSC	DVC	ShpC	LBR
Ca	16,200	31,200	36,600	37,000	16,000	47,600	16,900	20,200	82,100
Mg	6500	7630	7500	6360	4960	2400	6280	5680	7070
P	3820	6860	4940	2180	2120	14,100	2930	1680	1880
K	14,800	16,500	17,100	12,900	17,300	7550	16,500	15,500	14,100
Na	6760	6800	6910	6610	6580	3190	6750	6220	5780
Al	47,700	46,500	48,800	39,200	43,200	20,000	52,200	39,200	39,100
Sb	0.39	1.10	0.84	0.26	0.44	0.20	0.48	0.21	0.34
As	7.4	8.2	8.6	2.5	6.4	4.5	4.5	3.2	3.3
Ba	603	391	400	446	527	279	525	354	460
Be	1.7	1.7	1.7	1.2	1.3	0.8	1.7	1.3	1.4
Bi	0.14	0.14	0.15	0.09	0.12	0.03	0.17	0.06	0.10
Cd	3.3	9.6	8.4	1.0	3.6	2.4	8.8	0.3	1.4
Ce	66.0	56.3	51.4	49.4	53.7	42.1	58.4	50.9	46.6
Cs	4.2	4.5	4.7	3.3	4.4	1.6	4.9	3.1	3.8
Cr	93	329	343	114	78	124	128	40	54
Co	13.3	8.8	8.1	5.8	7.0	3.9	9.8	6.7	6.8
Cu	15.3	33.6	30.9	12.6	14.6	5.9	22.3	8.9	12.8
Ga	11.7	11.2	11.4	9.3	10.1	4.6	12.8	8.8	9.0
Fe	25,400	22,100	22,000	16,100	16,600	8340	24,900	16,300	17,600
La	39.3	47.0	39.0	32.4	34.6	49.8	36.9	30.2	27.0
Pb	17.2	14.3	13.4	10.8	14.6	7.5	15.6	11.4	12.2
Li	30.1	31.2	31.7	21.5	29.7	13.8	29.6	25.0	23.5
Mn	5740	1990	1290	1120	676	363	1970	348	1110
Mo	0.64	3.50	3.50	0.56	0.33	0.57	0.56	0.25	0.28
Ni	28.9	67.9	64.5	19.0	19.5	11.6	75.2	15.7	17.6

Nb	9.2	9.5	6.4	8.1	14	5.0	14	6.5	13
Rb	73.4	74.2	78.8	61.7	87.0	31.8	81.8	62.8	68.0
Sc	7.8	8.7	8.5	6.2	5.8	3.5	8.3	6.0	6.8
Ag	0.34	1.30	1.40	0.27	0.28	0.36	0.30	0.24	0.28
Sr	125	184	182	134	146	194	140	82	193
Tl	0.60	0.78	0.75	0.45	0.56	0.27	0.81	0.41	0.48
Th	8.9	8.4	8.6	7.2	8.6	4.1	9.8	7.2	7.3
Ti	1740	2110	1620	1820	2360	839	2280	1510	2370
U	4.7	8.3	6.0	2.9	3.4	11.4	3.5	2.2	2.4
V	74	141	141	49	52	45	79	49	49
Y	37.0	49.2	39.6	31.3	27.6	66.7	32.6	23.9	22.4
Zn	121	316	326	86	119	48	904	54	87
Hg	0.03	0.10	0.10	0.03	0.04	0.02	0.04	<0.02	0.02
Se	1.0	35	52	1.4	1.0	1.3	4.3	0.4	2.1

TABLE 18-VII

Significant ($P < 0.05$) Pearson correlation coefficients for various aquatic ecosystem components

Ecosystem component	Ecosystem component							
	Sediment (ESTL)		Aquatic plant		Aquatic invertebrate		Fish	
Sediment (USGS lab)	Se	0.99	Se	0.97	Se	0.92	Se	0.95
					Al	0.85		
					As	0.92		
	Ba	0.69			Ba	0.92		
					Cr	0.90		
	Be	0.93						
	Cd	0.98	Cd	0.73				
	Cu	0.97			Cu	0.78		
					Fe	0.91		
	Pb	0.78						
	Mg	0.76					Mg	−0.66
	Mn	0.83			Mn	0.87		
	Ni	0.99	Ni	0.71	Ni	0.79		
	Sr	0.80						
	V	0.96			V	0.93		
	Zn	0.99	Zn	0.91	Zn	0.83		
Sediment (ESTL)			Se	0.96	Se	0.94	Se	0.96
					Al	0.78		
					Cu	0.79	Cu	0.79
					Mn	0.87		
			Ni	0.74	Ni	0.84		
					V	0.92	V	0.81
			Zn	0.90	Zn	0.83		
Aquatic plant					Se	0.91	Se	0.95
					Cd	0.99		
					Mn	0.85	Mn	0.71
					Mo	0.83		
					Ni	0.98		
					Zn	0.84		
Aquatic invertebrate							Se	0.99
							Cu	0.84

Other elements

Concentrations of other elements in surficial sediments reported by USGS and ESTL are comparable (Tables 18-VI and 18-VIII) with correlation coefficients of >0.90

TABLE 18-VIII

Element concentrations ($\mu g\, g^{-1}$ dry weight; $n = 1$) in sediment from nine locations listed in Table 18-I; analyses from ESTL

Element	ACM	UEMC	LEMC	TC	USC	LSC	DVC	ShpC	LBR
Al	1650	1020	1320	1160	1010	1090	1330	944	1240
As	7.7	6.6	8.0	6.5	3.9	7.5	4.5	6.3	2.0
Ba	131	7.8	9.4	11.2	85	10.4	10.9	7.8	108
Be	0.8	0.7	0.7	0.6	<0.1	0.5	0.7	0.6	<0.1
B	12	9.0	13	9.4	7.3	8.7	8.3	9.0	9.0
Cd	2.8	5.2	4.4	0.6	1.9	1.4	5.5	0.4	0.8
Cr	33	10.2	12	6.7	20	7.6	7.7	6.2	16
Cu	8.8	18	18	5.9	7.0	5.4	10.7	5.8	5.7
Fe	184	161	175	156	710	130	171	140	782
Pb	8.1	5.6	5.8	4.8	5.0	4.3	6.3	5.1	3.9
Mg	453	461	498	388	258	315	363	367	516
Mn	280	117	107	91	51	173	142	29	99
Mo	<0.5	1.9	1.9	<0.5	<0.5	<0.5	<0.5	<0.5	<0.5
Ni	18	43	43	11	12	12	54	12	13
Sr	70	78	69	40	29	94	34	20	73
V	39	58	62	15	16	21	24	15	16
Zn	69	175	182	41	61	48	555	34	41

($P < 0.001$–0.003) for Se, Be, Cd, Cu, Ni, V, and Zn; 0.60–0.89 ($P = 0.01$–0.006) for Ba, Pb, Mg, Mn, and Sr; and <0.59 ($P = 0.26$–0.76) for Al, As, Cr, and Fe (Table 18-VII). The difference in concentrations of some elements was probably due to the four-acid digestion used at the USGS lab and the single-acid digestion at ESTL.

Concentrations of other elements in surficial sediments followed a similar trend as Se in sediments (Tables 18-VI and 18-VIII). Upper and lower East Mill Creek tended to have the highest concentrations of Sb, Cd, Cr, Cu, Mo, Ni, Ag, V, and Zn, whereas Dry Valley Creek had a moderate amount of these elements relative to the other six sites. Other sites such as lower Slug Creek had elevated Mn and Sr compared to upper Slug Creek and Angus Creek had elevated B, Mn, Sr, V, and Zn compared to upper Slug Creek, thus suggesting some element contamination in sediments at these sites. Mercury was not elevated at any of the nine sites (Table 18-VI).

The magnitude of difference in concentrations at upper and lower East Mill Creek sites compared to those in the non-impacted upper Slug Creek site ranged from 1.8- to 11-fold. In contrast, the magnitude of difference for Se concentrations between upper and lower East Mill Creek and upper Slug Creek ranged from 35- to 57-fold, thus suggesting a major disparity between Se enrichment in the East Mill Creek compared to upper Slug Creek.

Of all the elements measured, Se was the most bioaccumulative in the food web and the element of principal concern for effects in biota.

The sediment in aquatic ecosystems is an important pathway of element movement through the food web (Seelye et al., 1982). Sediment contains the most concentrated pool of elements in the aquatic environment and many types of aquatic organisms ingest sediment during the foraging process (Luoma, 1983), for example, fish (Dallinger and Kautzky, 1985; Dallinger et al., 1987; Kirby et al., 2001a,b).

Comparison to other Idaho data

Overall, the elevated concentrations of Se and other elements in sediments from several streams in the Blackfoot River watershed that were reported by TRC Environmental (1999), Montgomery Watson (1999, 2001a), and in the present study suggest widespread contamination of the aquatic environment.

Selenium concentrations in sediment from Maybe Creek, a tributary of Dry Valley Creek, near its mouth were 261 $\mu g\,g^{-1}$ (TRC Environmental, 1999), indicating that much of the Se loading in Dry Valley Creek comes from Maybe Creek. They reported that other segments of Maybe Creek contained 12–77 $\mu g\,g^{-1}$ of Se in sediment.

The Selenium Subcommittee investigated concentrations of Se, Cd, Mn, Ni, V, and Zn in sediment from numerous sites in the Southeastern Idaho Phosphate Resource Area in September 1998 (Montgomery Watson, 1999). Out of 54 sites investigated, 11 had Se concentrations in sediment of 2–4 $\mu g\,g^{-1}$, thus placing them in the concentration-of-concern range proposed by Stephens et al. (1997) and the moderate hazard range of Lemly (1995). These sites included Slug Creek, Dry Valley Creek, Rasmussen Creek (tributary to Angus Creek), and East Mill Creek. Three sites had sediment values greater than 4 $\mu g\,g^{-1}$ (State Land Creek, Sage Creek below Smoky Canyon Mine, North Fork Sage Creek below Pole Creek), which places them above the toxicity threshold of Stephens et al. (1997) and the high hazard of Lemly (1995). The Se concentrations in sediment reported by Montgomery Watson (1999) in East Mill Creek (2.9 $\mu g\,g^{-1}$) were substantially lower than those in the present investigation (35–52 $\mu g\,g^{-1}$), whereas their values for Dry Valley Creek (3.3 $\mu g\,g^{-1}$) were slightly lower than our values (4.3–5.2 $\mu g\,g^{-1}$). Other sediment contents reported by Montgomery Watson (1999) for similar sites were similar to those reported here.

Montgomery Watson (2001a) reported elevated Se concentrations in sediment collected in September 1999 from State Land Creek (2.1 $\mu g\,g^{-1}$), Pedro Creek (3.1 $\mu g\,g^{-1}$), Dry Valley Creek (3.9 $\mu g\,g^{-1}$), Rasmussen Creek (2.2 $\mu g\,g^{-1}$), and the Blackfoot River (2.1–3.0 $\mu g\,g^{-1}$), which were in the concentration-of-concern range of Stephens et al. (1997) and the moderate hazard of Lemly (1995). They reported high Se concentrations in sediment from Dry Valley Creek (6.2 $\mu g\,g^{-1}$), Angus Creek (5.1 $\mu g\,g^{-1}$), and East Mill Creek (5.0 $\mu g\,g^{-1}$), which were in the toxicity threshold range of Stephens et al. (1997) and the high hazard of Lemly (1995).

Aquatic plants

Selenium

Selenium concentrations in aquatic plants followed a similar pattern as in surficial sediments: low at five sites (0.6–1.5 $\mu g\,g^{-1}$, Sheep Creek, Trail Creek, Angus Creek, upper Slug Creek, lower Slug Creek), moderately elevated at Dry Valley Creek and lower Blackfoot River (3.8–4.5 $\mu g\,g^{-1}$), and very elevated at the upper and lower East Mill Creek (30–74 $\mu g\,g^{-1}$) (Table 18-IV). Sheep Creek and Trail Creek had the lowest Se concentrations in aquatic plants and surficial sediment. The correlation coefficient between Se concentrations in sediment and aquatic plants is 0.97 ($P < 0.0001$; Table 18-VII).

No guidelines were found in the literature proposing toxicity threshold concentrations for Se in aquatic plants that might be hazardous to aquatic organisms. However, most domestic animals exhibit signs of Se toxicosis when consuming terrestrial vegetation containing $\geq$3–5 $\mu g\,g^{-1}$ Se (National Research Council, 1980; Eisler, 1985; Olson, 1986). The low Se concentrations in aquatic plants from upper Slug Creek, Trail Creek, Sheep Creek, Angus Creek, and lower Slug Creek might be considered near background for the Blackfoot River watershed. By comparison, Se concentrations at Dry Valley Creek (3.8 $\mu g\,g^{-1}$) and lower Blackfoot River (4.5 $\mu g\,g^{-1}$) were elevated and those at upper and lower East Mill Creek (30–74 $\mu g\,g^{-1}$) were substantially elevated.

Although fish typically do not feed on macrophytes, dead macrophytes become an important contributor to detrital food. Benthic invertebrates readily accumulate Se from detritus (Alaimo et al., 1994), which in turn is bioaccumulated through the food web to predators such as fish. Saiki et al. (1993) concluded that high concentrations of Se in aquatic invertebrates and fish in Se-contaminated areas of central California were the result of food-chain transfer from Se-rich detritus rather than from other pathways. Likewise, elevated Se concentrations in aquatic plants from four of the stream sites in the Blackfoot River watershed probably contribute to Se transfer in the aquatic food web.

Other elements

Concentrations of elements in aquatic plants follow a similar trend as Se in surficial sediments (Table 18-IX). Lower East Mill Creek tended to have the highest concentrations of Al, As, Cd, Cu, Fe, Pb, Mg, Mo, Ni, Sr, V, and Zn, whereas Dry Valley Creek had an intermediate amount of these elements relative to the other seven sites. In contrast, lower Slug Creek tended to have slightly higher concentrations of Al, As, B, Ba, Cr, Fe, and Sr than Dry Valley Creek. This relative elevation of elements in aquatic plants in lower Slug Creek compared to Dry Valley Creek was not present in sediment data. In general, aquatic plants from Sheep Creek and Trail Creek tended to have low element concentrations compared to the other sites, except for Ba, which was elevated in aquatic plants from Trail

TABLE 18-IX

Element concentrations ($\mu g\, g^{-1}$ dry weight; $n = 1$) in aquatic plants from nine locations listed in Table 18-I

Element	ACM	UEMC	LEMC	TC	USC	LSC	DVC	ShpC	LBR
Al	168	69	1796	465	290	699	549	90	153
As	1.4	1.8	7.6	1.9	2.5	3.4	1.6	1.0	1.4
Ba	92	14	94	156	65	281	106	32	60
Be	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1
B	12	10	28	14	12	38	17	10	14
Cd	2.3	22	13	0.6	3.2	1.1	1.9	1.7	3.2
Cr	3.7	8.1	13.8	7.1	8.5	18	15	2.4	3.5
Cu	5.3	4.4	18	4.7	5.4	4.0	5.0	3.5	5.6
Fe	103	64	834	267	163	408	315	68	113
Pb	<2	<2	4.8	<2	<2	2.4	2.3	<2	<2
Mg	288	269	575	334	306	375	389	232	416
Mn	888	85	75	749	544	266	419	242	442
Mo	2.2	1.7	2.1	0.8	1.6	<0.5	1.4	0.5	0.8
Ni	7.4	7.7	41	5.7	7.6	8.1	27	4.0	6.4
Sr	64	53	99	39	36	61	53	21	45
V	3.8	3.6	79	8.1	6.0	15	14	2.1	4.0
Zn	32	166	197	22	40	34	300	29	115

Creek. Although upper Slug Creek is considered the reference site, it had higher concentrations of some elements in aquatic plants than did Sheep Creek plants.

There were strong correlation between element concentrations in aquatic plants and surficial sediments for Cd, Ni, and Zn (0.71–0.91, $P < 0.03$) and for V (0.59, $P < 0.10$) using the USGS sediment data (Tables 18-VI and 18-VII) and also for Cu and Sr using the ESTL data (Tables 18-VII and 18-VIII).

Elements accumulate in aquatic plants from both the water column (Bryson et al., 1984; Devi et al., 1996) and from sediment (Cherry and Guthrie, 1977; Dallinger and Kautzky, 1985; Dallinger et al., 1987). Significant correlations for several elements (Cd, Cu, Ni, Sr, V, Zn) between surficial sediments and aquatic plants suggest a strong interconnectedness in element cycles. Although few herbivores feed on aquatic plants directly, when rooted aquatic plants die, greater than 90% of their biomass comprises 90% of the detrital-food chain, whereas the remaining 10% is from algal and animal detritus (Teal, 1962; Mann, 1972). Much of the nutritional content in detritus comes from microbe enrichment and metabolic products, which add proteins and amino acids to detritus (Odum and de la Cruz, 1967; Foda et al., 1983). Plant litter and other coarse debris that enter a stream are a major source of energy that fuels higher trophic levels (Allan, 1995).

Thus, uptake of elements by aquatic plants might seem unimportant; however, elements in dead-plant material can play an important role in movement of mass and energy through the detrital food web to aquatic invertebrates and fish.

Comparison to other Idaho data

Periphyton (attached algae and detritus), plankton, and submerged macrophyte data demonstrate that aquatic plants were accumulating Se from water and sedimentary sources. Montgomery Watson (2001a,b) acknowledged that submerged aquatic plants were efficient accumulators of Se. Their measurements are similar to data presented here.

Elevated Se concentrations were found for grasses (mean 64 $\mu g\,g^{-1}$), forbs (78 $\mu g\,g^{-1}$), and shrubs (11 $\mu g\,g^{-1}$) in Maybe Creek (TRC Environmental, 1999). Montgomery Watson (1999) did not collect or analyze element concentrations in aquatic plants, however, Montgomery Watson (2001a) reported Se concentrations in periphyton collected from artificial substrates placed in streams between September and October 1999. Elevated Se concentrations were found in periphyton in the Blackfoot River (3.0 $\mu g\,g^{-1}$), Angus Creek (3.3–9.2 $\mu g\,g^{-1}$), Spring Creek (4.2–7.5 $\mu g\,g^{-1}$), and very high values in East Mill Creek (12–25 $\mu g\,g^{-1}$). Montgomery Watson (2001b) reported Se concentrations in periphyton collected from artificial substrates placed in streams between May and June 2000, but fewer streams than investigated in Montgomery Watson (2001a). Elevated Se concentrations were found in the Blackfoot River (4.3 $\mu g\,g^{-1}$) and Angus Creek (6.0 $\mu g\,g^{-1}$).

Plankton samples collected from various sites in Blackfoot Reservoir contained Se concentrations of $\leq$1.5 $\mu g\,g^{-1}$ in September 1999 (Montgomery Watson, 2001a). However, in the May 2000 sampling, 9 of 12 samples contained a geometric mean Se concentration of 3.3 $\mu g\,g^{-1}$ (Montgomery Watson, 2001b).

Submerged macrophytes were collected in September 1999 from numerous stream sites in the Blackfoot River watershed and analyzed for Se concentrations (Montgomery Watson, 2001a). Several samples had elevated concentrations that range from 3.2 to 4.8 $\mu g\,g^{-1}$, 10 samples had high concentrations ranging from 5.1 to 8.8 $\mu g\,g^{-1}$, and one site, East Mill Creek, with very high concentrations ranging from 31 to 46 $\mu g\,g^{-1}$. Submerged macrophytes collected in May 2000 contained similar Se concentrations to those sampled in the September 1999 (Montgomery Watson, 2001b).

A native bryophyte that was collected from a seep at the base of the Wooley Valley phosphate mine Unit 4 waste-rock pile in the headwater area of Angus Creek contained very elevated concentrations of several elements including Cd (160 $\mu g\,g^{-1}$), Co (180 $\mu g\,g^{-1}$), Cr (210 $\mu g\,g^{-1}$), Mn (33,000 $\mu g\,g^{-1}$), Ni (2000 $\mu g\,g^{-1}$), V (1000 $\mu g\,g^{-1}$), Zn (11,000 $\mu g\,g^{-1}$), and Se (750 $\mu g\,g^{-1}$) (Herring et al., 2001). This site and others on Angus Creek were monitored for element accumulation in late spring and late summer 1999 using an introduced bryophyte, *Hygrohypnum ochraceum* (Herring et al., 2001). The same elements as present in the native bryophyte accumulated in the introduced bryophyte, but Se was the most enriched of the elements measured.

Aquatic invertebrates

Selenium

The opportunistic sample of crane fly nymphs collected at the Angus Creek headwater contained 186 $\mu g\,g^{-1}$ Se (Table 18-IV). Aquatic invertebrates collected from Sheep Creek and Trail Creek had low Se concentrations (2.6–2.9 $\mu g\,g^{-1}$), upper Slug Creek, lower Blackfoot River, and Angus Creek had moderate concentrations (4.9–7.2 $\mu g\,g^{-1}$), and Dry Valley Creek and upper East Mill Creek had elevated concentrations (19–36 $\mu g\,g^{-1}$).

Correlation coefficients between Se concentrations in aquatic invertebrates were strong with aquatic plants (0.91, $P < 0.004$), and surficial sediment (0.94, $P < 0.002$; Table 18-VII).

In the hazard assessment protocol proposed by Lemly (1995), Se concentrations in benthic invertebrates of <2 $\mu g\,g^{-1}$ represent no identifiable hazard, 2–3 $\mu g\,g^{-1}$ minimal hazard, 3–4 $\mu g\,g^{-1}$ low hazard, 4–5 $\mu g\,g^{-1}$ moderate hazard, and >5 $\mu g\,g^{-1}$ high hazard. Using these values suggests that Se concentrations in invertebrates at upper Slug Creek, lower Blackfoot River, Angus Creek, Dry Valley Creek, and upper East Mill Creek probably were adversely affecting sensitive larval fish, but not necessarily older life stages such as subadults or adults.

Selenium concentrations in aquatic invertebrates from Sheep Creek and Trail Creek (2.6–2.9 $\mu g\,g^{-1}$) were the lowest of the sites, but were close to the proposed dietary Se threshold of 3 $\mu g\,g^{-1}$ (Lemly, 1993, 1996b; Hamilton, 2002). Selenium concentrations of 4.6 $\mu g\,g^{-1}$ in zooplankton caused nearly complete mortality of razorback sucker in about 10–13 days (Hamilton et al., 2001a,b). Several other studies summarized in Hamilton (2002) reported that dietary Se concentrations of 4–6 $\mu g\,g^{-1}$ caused adverse effects in larval fish. Consequently, the moderate concentrations in upper Slug Creek, lower Blackfoot River, and Angus Creek (4.9–7.2 $\mu g\,g^{-1}$), and the elevated concentrations in Dry Valley Creek and upper East Mill Creek (19–36 $\mu g\,g^{-1}$) were of concern to the health of fishery resources.

The high Se concentrations in crane fly nymphs (186 $\mu g\,g^{-1}$) probably pose an acute toxicity dietary hazard to predators that might feed on them. Sandhill cranes (*Grus canadensis*) have been observed feeding in the marshy area within a few meters of the location where the nymphs were collected (L. Stillings, US Geological Survey, personal communication).

Although upper Slug Creek was the reference site and contained relatively low Se concentrations in water, surficial sediments, and aquatic plants, Se concentrations in aquatic invertebrates were elevated. Benthic invertebrates can be efficient accumulators of Se and can retain elevated concentrations over a long time. For example, Maier et al. (1998) reported that aquatic invertebrates contained Se concentrations of 1.7 $\mu g\,g^{-1}$ at pretreatment of a watershed with Se fertilizer and elevated concentrations during post-treatment monitoring, 4.0–5.0 $\mu g\,g^{-1}$ for six samplings over 11 months.

The linkage between Se concentrations in invertebrates and sediment and detritus was supported by the strong correlation between aquatic invertebrates and surficial sediments

(0.92–0.94) and between aquatic invertebrates and aquatic plants (0.91). Selenium concentrations in invertebrates and its bioaccumulation through the food web to higher trophic organisms such as fish have been reported by several investigators (Finley, 1985; Hamilton et al., 2001a,b).

Other elements

Crane fly larvae nymphs collected from the Angus Creek headwater contained elevated concentrations of Cr, Fe, Ni, Sr, V, and Zn compared to invertebrates from the other eight sites (Table 18-X). Concentrations of elements in aquatic invertebrates from seven sites (none collected at lower East Miller Creek or lower Slug Creek) followed a similar trend to that found in surficial sediments and aquatic plants (Table 18-X). Low concentrations occur in aquatic invertebrate specimens from Sheep Creek and Trail Creek, moderate concentrations in upper Slug Creek and lower Blackfoot River, and elevated concentrations in upper East Mill Creek, Angus Creek, and Dry Valley Creek specimens. At the three sites with elevated concentrations, the elements generally elevated include Al, B, Ba, Cr, Fe, V, and Zn.

TABLE 18-X

Element concentrations ($\mu g\, g^{-1}$ dry weight; $n = 1$) in composite samples of aquatic invertebrates from seven locations listed in Table 18-I and site ACH[1]

Element	ACH	ACM	UEMC	TC	USC	DVC	ShpC	LBR
Al	49	459	257	72	224	381	59	213
As	2.6	3.0	3.6	<2	3.0	2.7	<2	2.5
Ba	14	215	30	112	145	140	22	30
Be	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1
B	4.3	8.9	9.5	6.8	6.6	8.2	3.7	7.1
Cd	12	2.7	31	1.0	3.7	1.7	0.6	0.5
Cr	31	11	31	2.7	14	12	2.6	5.6
Cu	12	21	51	11	29	17	16	27
Fe	374	296	251	68	174	262	59	160
Pb	<2	<2	<2	<2	<2	<2	<2	<2
Mg	308	238	191	135	175	227	129	181
Mn	168	434	29	196	109	198	21	68
Mo	3.5	1.5	1.0	0.6	0.7	1.0	0.7	0.6
Ni	46	16	12	1.8	6.9	50	2.2	6.3
Sr	32	11	21	8.2	47	30	3.8	17
V	46	11	18	1.7	5.7	11	1.7	5.5
Zn	326	330	595	87	143	659	138	127

[1]Crane fly nymphs collection site at Angus Creek headwaters.

At upper East Mill Creek, other elevated elements include Cd, Cu, and Ni, whereas in Dry Valley Creek samples, Ni and Sr are elevated.

There are strong correlations between element concentrations in aquatic invertebrates and surficial sediments for Al, As, Ba, Cr, Cu, Fe, Mn, Ni, V, and Zn (0.78–0.93, $P < 0.05$). There also are strong correlations for elements between aquatic invertebrates and aquatic plants for Cd, Mn, Mo, Ni, and Zn (0.83–0.99, $P < 0.05$). Several elements elevated in aquatic invertebrates from upper East Mill Creek, Angus Creek, and Dry Valley Creek were probably linked with concentrations in aquatic surficial sediment and aquatic plants.

Comparison to other Idaho data

Elevated Se concentrations have been reported in benthic invertebrates collected from ponds (110–390 $\mu g\,g^{-1}$) and a flowing section (14 $\mu g\,g^{-1}$) of Maybe Creek, a tributary of Dry Valley Creek (TRC Environmental, 1999). Those Se concentrations are substantially higher than those measured here in benthic invertebrates from Dry Valley Creek.

Benthic invertebrates collected from various sites in Blackfoot Reservoir in September 1999 contain $\leq 2\ \mu g\,g^{-1}$ Se, except for three samples, which contain Se concentrations of 3.81 $\mu g\,g^{-1}$ (Montgomery Watson, 2001a). However, 8 of 12 samples collected in May 2000 from Blackfoot Reservoir contain a geometric mean Se concentration of 7.8 $\mu g\,g^{-1}$ (range 5.3–12 $\mu g\,g^{-1}$; Montgomery Watson, 2001b).

Benthic invertebrates collected in September 1999 from numerous stream sites in the Blackfoot River watershed had moderately elevated Se concentrations in 5 of 26 samples (3.0–4.6 $\mu g\,g^{-1}$), elevated concentrations in five samples (5.0–15 $\mu g\,g^{-1}$), and very elevated concentrations at East Mill Creek (72 $\mu g\,g^{-1}$) (Montgomery Watson, 2001a). In the May 2000 sampling, moderately elevated Se concentrations occurred in 11 of 42 samples (3.0–4.9 $\mu g\,g^{-1}$), 17 samples had elevated concentrations (5.0–37 $\mu g\,g^{-1}$), and East Mill Creek had 100, 120, and 170 $\mu g\,g^{-1}$ (Montgomery Watson, 2001b).

Selenium concentrations reported by Montgomery Watson (2001a,b) tended to be higher than those determined here for similar collection sites. The large number of samples with substantial Se concentrations far above the proposed toxic threshold of 3 $\mu g\,g^{-1}$ suggest that benthic invertebrate populations were highly contaminated with Se.

Fish

Selenium

Six fish species were collected, but no one fish species was collected at all eight sites. Mottled sculpin were collected at six sites and Se in these fish followed the same pattern as present in the geometric mean of combined fish data (Table 18-XI). Speckled dace were collected at five sites and in general had the highest whole-body Se concentrations of the species collected. Although redside shiners were collected from only four sites, they

consistently had the lowest whole-body Se concentrations of the species collected at those sites. Geometric mean Se concentrations in fish were low from upper Slug Creek (4.0 $\mu g\ g^{-1}$), Trail Creek (5.1 $\mu g\,g^{-1}$), Sheep Creek (5.2 $\mu g\,g^{-1}$), and lower Slug Creek (5.3 $\mu g\,g^{-1}$), intermediate from Angus Creek (6.4 $\mu g\,g^{-1}$) and lower Blackfoot River (7.8 $\mu g\,g^{-1}$), and elevated from Dry Valley Creek (16.1 $\mu g\,g^{-1}$) and upper East Mill Creek (32.2 $\mu g\,g^{-1}$) (Table 18-XI).

Selenium concentrations in fish from the eight sites, based on geometric-mean values, followed the same pattern of accumulation as in surficial sediments, aquatic plants, and aquatic invertebrates. This accumulation pattern is supported by other studies, which show that fish bioaccumulate Se primarily from the dietary route of exposure (Lemly, 1993, 1996b; Maier and Knight, 1994).

Here, there were consistent differences in Se concentrations between fish species at a site. The particular feeding niche can influence the residues accumulated. Ney and Van Hassel (1983) reported that the benthic species fantail darter (*Etheostoma flabellare*) and blacknose dace (*Rhinichthys atratulus*) had the highest accumulations of Cd, Pb, Ni, and Zn, bottom-dwelling northern hog sucker (*Hypentelium nigricans*) and white sucker (*Catostomus commersoni*) had intermediate accumulations, and water-column dwelling redbreast sunfish (*Lepomis auritus*) and rock bass (*Ambloplites rupestris*) had the least accumulations. Others have reported that element residues vary among fish species, but variations were not conclusively related to food habits and trophic status (summarized in Wiener and Giesy, 1979).

We found that speckled dace tended to have the highest Se concentrations of the six species collected, whereas redside shiner had the lowest concentrations. Speckled dace is a bottom browser that feeds on invertebrates and plant material (Lee et al., 1980), possibly detritus, and thus seemed to accumulate elevated elements similar to bottom-feeding

TABLE 18-XI

Se concentrations ($\mu g\,g^{-1}$ dry weight; $n = 1$) in whole-body fish from eight locations listed in Table 18-I

Species	ACM	UEMC	TC	USC	LSC	DVC	ShpC	LBR
Brook trout	–[1]	42.7	–	2.4	–	–	–	–
Cutthroat trout	6.7	24.3	–	–	–	16.6	3.6	–
Mottled sculpin	7.1	–	4.4	3.7	–	15.6	5.6	6.7
Longnose dace	7.0	–	5.9	–	–	–	7.1	–
Speckled dace	–	–	6.3	7.2	5.8	16.1	–	6.7 14.1
Redside shiner	5.3	–	4.0	–	4.9	–	–	5.8
Geometric mean	6.4	32.2	5.1	4.0	5.3	16.1	5.2	7.8

[1]Dash means not collected.

redear sunfish reported by Campbell (1994). Redside shiners are omnivores (Lee et al., 1980) and thus seemed to accumulate low element concentrations similar to omnivorous bluegill reported by Campbell (1994). We conclude that feeding niche differences result in different dietary exposures and more importantly, different Se bioaccumulation in fish.

Other elements

Several elements are elevated in whole-body fish from the eight sites (Table 18-XII). Similar to Se concentrations, highest element concentrations are in speckled dace and the lowest concentrations tended to occur in redside shiner. The elements that were somewhat elevated include Al, Ba, Cu, Mn, Ni, Sr, and Zn, but no site had fish with consistently elevated values based on geometric means (Table 18-XIII). Fish from upper Slug Creek, the reference site, had the highest geometric means for Al, Ba, Cr, and Mn. Upper East Mill Creek fish had the highest geometric-mean Cu concentrations. Lower Slug Creek fish had the highest geometric-mean Sr and Zn.

There were few correlations between elements in fish (using the geometric mean for all fish at each site) and surficial sediment, aquatic plants, or aquatic invertebrates. There are three correlations with surficial sediment data: Cu (0.68, $P = 0.06$), Mg (-0.66, $P = 0.08$), and V (0.72, $P = 0.07$). For aquatic plant data, there is one correlation: Mg (0.71, $P = 0.05$). For aquatic invertebrate data there are two correlations: Cu (0.84, $P = 0.02$) and Mn (0.73, $P = 0.06$). Copper and V are the only two elements in surficial sediment and aquatic invertebrates that are consistently correlated with Se concentrations in fish.

There seemed to be no parallel bioaccumulation of elements with Se in fish from the eight sites. In contrast to Se concentrations in fish, concentrations of some elements were highest in upper Slug Creek, which was the reference site. Only elevated Cu occurred in fish from upper East Mill Creek, which had the highest Se concentrations in fish as well as other aquatic ecosystem components, compared to the other sites.

Selenium-dominant contamination found in the Blackfoot River watershed was also found in other contaminant investigations. For example, contaminated food chains in coal ash settling basins showed elevated Al, Co, Eu, Fe, La, Lu, Sm, Se, and Ti in biota, but only Se was of concern in terms of detriment to biota (Furr et al., 1979). Lemly (1985) reviewed information in 10 studies of potential causes of fishery problems at Belews Lake (16 species eliminated, two species present as adults only, one species recolonized, and one species unaffected), and of the 16 elements of concern only Se was present at elevated concentrations in water and fish. Montgomery Watson (1999) concluded that Se was the major element of concern associated with phosphate mining activities in the Blackfoot River watershed of southeast Idaho and not other elements (Cd, Mn, Ni, V, Zn). In the two reproduction studies with endangered razorback sucker, several elements (Al, Sb, As, B, Ba, Be, Bi, Cd, Cr, Co, Cu, Fe, Pb, Li, Mg, Mn, Mo, Ni, Si, Ag, Sr, Tl, Sn, Ti, V, Zn) were elevated in water, zooplankton, sediment, and fish eggs, but only Se was high enough to cause adverse effects in fish (Hamilton et al., 2001a,b).

TABLE 18-XII

Element concentrations ($\mu g\, g^{-1}$ dry weight; $n = 1$) in whole-body fish from eight locations listed in Table 18-I

Element	ACM				UEMC	TC			
	Cutthroat trout	Mottled sculpin	Longnose dace	Redside shiner	Cutthroat trout	Mottled sculpin	Longnose dace	Speckled dace	Redside shiner
Al	40	86	166	10	41	24	4	226	8
As	<2	<2	<2	<2	<2	<2	<2	<2	<2
Ba	1.5	6.6	9.2	3.5	3.0	4.2	11.1	14.9	4.1
Be	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1
B	2.1	2.2	2.3	2.0	2.1	1.8	1.8	2.0	2.0
Cd	<0.3	<0.3	<0.3	<0.3	<0.3	<0.3	<0.3	<0.3	<0.3
Cr	1.2	1.6	1.9	1.3	1.6	1.2	1.0	1.4	1.1
Cu	3.4	2.4	7.3	2.4	11	2.9	3.8	8.3	3.5
Fe	20	36	30	20	19	24	23	40	20
Pb	<2	<2	<2	<2	<2	<2	<2	<2	<2
Mg	121	137	109	120	104	124	135	147	128
Mn	25	72	78	16	5.4	27	17	56	11
Mo	<0.5	<0.5	<0.5	<0.5	<0.5	<0.5	<0.5	<0.5	<0.5
Ni	1.7	4.6	2.7	2.3	1.6	3.0	2.8	2.7	2.2
Sr	10	29	24	20	4	20	26	23	17
V	<0.2	2.1	0.8	<0.2	<0.2	0.9	<0.2	0.7	<0.2
Zn	81	64	98	142	62	69	96	149	126

Continued

TABLE 18-XII *Continued*

Element	USC			LSC		DVC		
	Brook trout	Mottled sculpin	Speckled dace	Speckled dace	Redside shiner	Cutthroat trout	Mottled sculpin	Speckled dace
Al	518	194	162	213	16	49	112	160
As	<2	<2	<2	<2	<2	<2	<2	<2
Ba	5.6	7.6	22	11	5.1	6.5	3.3	5.9
Be	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1
B	2.8	1.6	1.6	2.2	2.0	1.6	1.7	1.8
Cd	<0.3	<0.3	<0.3	<0.3	<0.3	<0.3	<0.3	<0.3
Cr	5.9	1.4	1.4	1.8	1.3	1.3	1.4	1.5
Cu	2.9	2.2	6.3	7.2	2.8	2.2	2.0	2.2
Fe	34	27	42	38	22	18	23	29
Pb	<2	<2	<2	<2	<2	<2	<2	<2
Mg	121	96	132	151	132	123	113	125
Mn	20	14	204	42	13	24	46	40
Mo	<0.5	<0.5	<0.5	<0.5	<0.5	<0.5	<0.5	<0.5
Ni	2.0	1.8	2.5	4.1	2.5	1.4	2.7	2.7
Sr	14	10	19	38	30	9	16	20
V	2.0	0.8	0.7	0.8	<0.2	<0.2	1.3	0.7
Zn	50	38	109	134	122	82	69	116

Element	ShpC			LBR			
	Cutthroat trout	Mottled sculpin	Longnose dace	Mottled sculpin	Speckled dace	Speckled dace	Redside shiner
Al	24	104	195	69	16	127	16
As	<2	<2	<2	<2	<2	<2	<2
Ba	1.5	3.7	8.8	2.9	8.3	6.3	2.7
Be	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1
B	1.7	1.8	1.8	1.7	1.6	2.0	1.6
Cd	<0.3	<0.3	<0.3	<0.3	<0.3	<0.3	<0.3
Cr	1.3	1.3	1.5	1.2	1.1	1.5	1.0
Cu	2.7	2.5	5.7	2.4	3.2	6.0	1.8
Fe	16	27	29	21	24	25	17
Pb	<2	<2	<2	<2	<2	<2	<2
Mg	104	113	102	112	131	112	102
Mn	5	11	16	18	15	29	8
Mo	<0.5	<0.5	<0.5	<0.5	<0.5	<0.5	<0.5
Ni	1.5	2.9	2.2	2.2	3.2	2.2	1.6
Sr	4	13	24	17	32	20	18
V	<0.2	0.6	0.7	0.9	<0.2	0.6	0.3
Zn	62	50	59	50	98	101	128

TABLE 18-XIII

Geometric mean of element concentrations ($\mu g\, g^{-1}$ dry weight) in whole-body fish from eight locations listed in Table 18-I

Element	ACM	UEMC	TC	USC	LSC	DVC	ShpC	LBR
Al	49	41	20	253	58	96	79	39
As	<2	<2	<2	<2	<2	<2	<2	<2
Ba	4.2	3.0	7.3	9.8	7.5	5.0	3.7	4.5
Be	<0.1	0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1
B	2.1	2.1	1.9	1.9	2.1	1.7	1.8	1.7
Cd	<0.3	<0.3	<0.3	<0.3	<0.3	<0.3	<0.3	<0.3
Cr	1.5	1.6	1.2	2.3	1.5	1.4	1.4	1.2
Cu	3.5	11	4.2	3.4	4.5	2.1	3.4	3.0
Fe	26	19	26	34	29	23	23	22
Pb	<2	<2	<2	<2	<2	<2	<2	<2
Mg	121	104	133	115	141	120	106	114
Mn	39	5.4	23	39	23	35	10	16
Mo	<0.5	<0.5	<0.5	<0.5	<0.5	<0.5	<0.5	<0.5
Ni	2.6	1.6	2.7	2.1	3.2	2.2	2.1	2.2
Sr	19	4	21	14	34	14	11	21
V	1.3	<0.2	0.8	1.0	0.8	1.0	0.6	0.5
Zn	92	62	106	59	128	87	57	89

The possibility of a tissue-based criterion was discussed at the peer consultation workshop on Se aquatic toxicity and bioaccumulation (US Environmental Protection Agency, 1998b). The conclusion was that the tissue-based criterion might be the best approach because tissue residues accounted for the biogeochemical pathways of Se by integrating the route, duration, and magnitude of exposure, chemical form, metabolic transformations, and modifying biotic and abiotic factors. Hamilton (2002) proposed a national criterion of 4 $\mu g\,g^{-1}$ in whole-body based on the review of several laboratory and field studies. This value was the same as the toxicity threshold for fish proposed earlier by Lemly (1993, 1996b) and similar to the threshold of 4.5 $\mu g\,g^{-1}$ proposed by Maier and Knight (1994). Recent papers proposed Se toxicity thresholds of 6 $\mu g\,g^{-1}$ for coldwater anadromous fish and 9 $\mu g\,g^{-1}$ for warm-water fish (DeForest et al., 1999; Brix et al., 2000), although these proposals are controversial (Hamilton, 2003).

Comparison to other Idaho data

Rich and Associates (1999) reported concentrations of elements in cutthroat trout, rainbow trout (*Oncorhynchus mykiss*), brook trout, sculpin species, dace species, and redside shiner collected from Dry Valley Creek immediately upstream of the Blackfoot River, and Dry Valley Creek directly below Maybe Creek. In general, their concentrations were higher than those presented here for Cd, Cr, Cu, V, and Zn, but lower for Se. They concluded that Se and other elements were probably causing stress in fish populations in Dry Valley Creek.

Montgomery Watson (1999) reported that salmonid fillets from a 1998 sampling contained Se concentrations of 6 $\mu g\,g^{-1}$ wet weight (maximum 7.9 $\mu g\,g^{-1}$) from East Mill Creek, whereas fish from two other sites (Blackfoot River above Wooley Range Ridge Creek and South Fork Sage Creek) had 1.2–1.3 $\mu g\,g^{-1}$. Converting these values to a dry weight basis (dry weight = wet weight $\times$ 4; assuming 75% moisture) results in 24 $\mu g\,g^{-1}$ in fish fillets from East Mill Creek and 4.8–5.2 $\mu g\,g^{-1}$ in fish fillets from the reference sites.

Selenium concentrations in fillets reported by Montgomery Watson (1999) underestimate the concentrations in whole body fish, which is the dominant matrix of Se residues in fish reported in the literature. Muscle contains less Se than whole-body due to the relatively high amounts of Se in spleen, liver, kidney, heart, and other tissues, especially mature ovaries (Adams, 1976; Lemly, 1982; Hilton et al., 1982). Consequently, the actual whole-body Se concentrations in trout would be about 40 $\mu g\,g^{-1}$ in fish from East Mill Creek and 8–8.7 $\mu g\,g^{-1}$ in fish from the Blackfoot River and South Fork Sage Creek (based on a conversion factor of 1.667 $\times$ muscle concentration = whole-body concentration; Lemly and Smith, 1987). Other conversion factors reported in the literature were 2.355 based on data from Adams (1976) for rainbow trout and 1.745 from Lemly (1982) for bluegill and largemouth bass. Here, the Se concentrations in whole-body brook trout (42.7 $\mu g\,g^{-1}$) and cutthroat trout (24.3 $\mu g\,g^{-1}$) from East Mill Creek were comparable to those reported by Montgomery Watson (1999) after conversion to dry weight and whole body (40 $\mu g\,g^{-1}$).

Selenium concentrations in salmonids collected in September 1999 were elevated in 21 of 50 samples (4.2–9.7 $\mu g\,g^{-1}$) and high in seven samples (12–31 $\mu g\,g^{-1}$) (converted to dry weight and whole body). For salmonids collected in May 2000, Se concentrations were elevated in 13 of 27 samples (5.2–9.2 $\mu g\,g^{-1}$) and high in 12 samples (10–48 $\mu g\,g^{-1}$) (converted to dry weight and whole body).

These Se residues in salmonids were substantially above background concentrations in fish from laboratory and field investigations. Background Se concentrations in fish are typically 1-2 $\mu g\,g^{-1}$ (Maier and Knight, 1994; Hamilton et al., 2000). More importantly, the Se residues were above those reported to cause adverse effects in early life stages of fish, including salmonids (4–5 $\mu g\,g^{-1}$; Hamilton et al., 2000). In particular, Se residues of 5.2 $\mu g\,g^{-1}$ in rainbow trout were associated with reduced survival (Hunn et al., 1987), and 3.8–4.9 $\mu g\,g^{-1}$ in chinook salmon (*Oncorhynchus tshawytscha*) were associated with reduced survival and growth (Hamilton et al., 1986; Hamilton and Wiedmeyer, 1990). Older life stages typically are more tolerant of contaminant stresses than are early life stages (Rand and Petrocelli, 1985). Consequently, effects in adults may not be as readily apparent as effects in early life stages.

Based on the above discussion, Se contamination of the Blackfoot River and its tributaries is most likely adversely affecting aquatic resources, especially the early life stages of fish. Thurow et al. (1981) reported that 13 fish species used the Blackfoot River and its tributaries and that the indigenous cutthroat trout was the dominant species. They noted that cutthroat trout used several tributaries, as well as the main stem river and the Blackfoot Reservoir during their life cycle. Thurow et al. (1981) noted the potential for mining activities to cause negative effects on trout and others species, primarily from erosion, sedimentation, and nutrient loading from phosphorous, but did not specially mention impacts from other elements.

Eggs from cutthroat trout in 1999 contained Se concentrations of 4.4 and 6.7 $\mu g\,g^{-1}$ dry weight in two ripe females, 4.0 $\mu g\,g^{-1}$ in two partially spawned females, and 1.4 $\mu g\,g^{-1}$ in females from the reference site, Henry's Lake (Montgomery Watson, 2000). These data demonstrate that trout were accumulating Se and depositing it in their eggs. However, the Se concentrations in eggs were less than the toxic effects threshold of 10 $\mu g\,g^{-1}$ proposed by Lemly (1993, 1996b). The low number of egg samples in Montgomery Watson (2000) precludes further speculation on the extent of Se contamination of fish eggs.

Forage fish samples collected in September 1999 from various sites in the Blackfoot River watershed contained Se concentrations of $\geq$4 $\mu g\,g^{-1}$ (Montgomery Watson, 2001a), which are above the generally accepted toxic threshold of 4 $\mu g\,g^{-1}$ (Lemly, 1993, 1996b; Maier and Knight, 1994; Hamilton, 2002). Nine of 13 samples had elevated Se concentrations in fish (5.2–8.3 $\mu g\,g^{-1}$, after conversion to dry weight), and two samples had high Se concentrations of 10 and 12.9 $\mu g\,g^{-1}$ (Montgomery Watson, 2001a). For forage fish collected from various sites in the Blackfoot River watershed in May 2000, 13 of 36 samples contained Se concentrations of 5.0–9.4 $\mu g\,g^{-1}$, and 13 samples had concentrations of 10–37 $\mu g\,g^{-1}$ (Montgomery Watson, 2001b). Selenium concentrations in forage fish reported by Montgomery Watson (2001a,b) were similar to those determined here.

Other considerations

One consideration is the presence of elevated Se residues in fish without readily apparent biological effects. However, it should be considered that the data in the current study and studies by others (Rich and Associates, 1999; Montgomery Watson, 1999, 2000, 2001a,b) were contaminant surveys and not biological effect studies. No biological effects such as survival, growth, reproduction, diversity, population structure, community structure, or other biological effects were measured.

A second consideration is that residues measured in fish were for adults or subadults. Fish in these life stages are generally not as sensitive to the effects of environmental contaminants as those in earlier stages (Rand and Petrocelli, 1985).

Monitoring of fish populations in rivers is an insensitive measure of contaminant effects unless substantial effort is made to assess the health of the fish community. This assertion was addressed by the US Environmental Protection Agency in their guidelines for deriving water-quality criteria. Stephan et al. (1985) found that most monitoring programs are insensitive for number of taxa and individuals, which limits their usefulness in developing water-quality criteria because some relevant changes may go undetected. Limited field studies may be able to determine whether criteria are under-protective, but only high-quality field studies can clearly determine that criteria are over-protective.

The claim of no biological effects in stream or river studies cannot often be confirmed without appropriate biological effects tests and often fall into the null fallacy trap: (a) there is no evidence for adverse effects, vs. (b) there is evidence for no adverse effects (J. Skorupa, US Fish and Wildlife Service, personal communication). The null fallacy occurs when statement (a) (a null finding) is given equal weight as statement (b) (a positive finding). What often is overlooked is that a null finding usually implies a lack of positive evidence in both directions – for effects or for absence of effects.

Montgomery Watson (2001b) found that higher than expected Se concentrations in forage fish from a reference site on Spring Creek above influences of East Mill Creek were probably due to the mobility of fish. Forage fish in the upper Spring Creek contained Se concentrations of 10, 12, and 22 $\mu g\, g^{-1}$. However, in spite of high Se residues in whole-body forage fish collected in May 2000, Montgomery Watson (2001b) concluded that neither Se nor Cd was impacting forage fish in the Blackfoot Reservoir. Likewise, Montgomery Watson (2001a) reported elevated Se concentrations in forage fish collected in September 1999, yet concluded that forage fish in the reservoir show no evidence of impact. Because no biological effects were assessed in fish collections in September 1999 or May 2000, the null fallacy trap applies to those conclusions.

HAZARD ASSESSMENT

Lemly (1995) presented a protocol for aquatic hazard assessment of Se, formulated primarily in terms of the potential for food-chain bioaccumulation and reproductive impairment in fish and aquatic birds. The protocol incorporated five ecosystem components

including water, sediment, benthic invertebrates, fish eggs, and bird eggs. Each component was given a numeric score based on the degree of hazard: (a) no identifiable hazard; (b) minimal hazard; (c) low hazard; (d) moderate hazard; (e) high hazard. The final hazard characterization was determined by adding the individual scores and comparing the total to the following evaluation criteria: 5, no hazard; 6–8, minimal hazard; 9–11, low hazard; 12–15, moderate hazard; 16–25, high hazard. Lemly (1996a) modified his protocol for use with four ecosystem components due to the difficulty in collecting residue information for all five components in an assessment. He adjusted the final ecosystem-level hazard assessment to the following four-component evaluation criteria: 4, no hazard; 5–7, minimal hazard; 8–10, low hazard; 11–14, moderate hazard; 15–20, high hazard.

Lemly (1995) defined the five categories of hazards as follows: (a) high hazard denotes an imminent, persistent toxic threat sufficient to cause complete reproductive failure in most species of fish and aquatic birds; (b) moderate hazard indicates a persistent toxic threat of sufficient magnitude to substantially impair but not eliminate reproductive success; some species will be severely affected whereas others will be relatively unaffected; (c) low hazard denotes a periodic or ephemeral toxic threat that could marginally affect the reproductive success of some sensitive species, but most species will be unaffected; (d) minimal hazard indicates that no toxic threat identified but concentrations of Se are slightly elevated in one or more ecosystem components compared to uncontaminated reference sites; (e) no hazard denotes that no toxic threat is identified and Se concentrations are not elevated in any ecosystem component. Table 18-XIV gives the Se concentrations for each hazard level for the four-component model.

The Se hazard protocols give equal weigh to each component (Lemly, 1995, 1996a). However, a critique of the protocol pointed out the need to give more weight to the biological components: benthic invertebrates, fish eggs, and bird eggs (H. Ohlendorf, 1996, written communication). Ohlendorf suggested a multiplication factor of two for benthic invertebrate data and a factor of three for fish eggs and bird eggs. Similar concerns have been raised by others (M. Sylvester, US Geological Survey, 2002; B. Osmundson, US Fish Wildlife Service, 2001, written communications). The weighting of the three biological components seems justified based on the repeated expression of their importance in the Se literature (reviews by Lemly, 1985, 1993; Maier and Knight, 1994; Presser et al., 1994; Hamilton and Lemly, 1999; Hamilton 2002, 2003).

Incorporating these factors into the protocol using the offset summation approach used by Lemly (1995, 1996a) results in modified final characterizations for the four-component protocol of 7, no hazard; 8–13, minimal hazard; 14–20, low hazard; 21–27, moderate hazard; and 28–35, high hazard (Table 18-XIV). The offset summation is explained as follows: for the low hazard column, Lemly (1996a) gives a score of 3 for each of the four components being evaluated (water, sediment, benthic invertebrate, and fish eggs), which results in a summed score of 12 (Table 18-XIV). However, if in an environmental situation all measured Se concentrations of the four components fell into the "low" column, the additive effect of the combined low exposures would most likely result in a "moderate" final hazard to biota. Thus, Lemly (1996a) sets the final hazard range for a "low" final hazard at 8–10, instead of closer to the summed total of 12. This offsetting of the final hazard total

TABLE 8-XIV

Aquatic ecosystem components and concentrations posing various hazards (based on Lemly, 1996a)

Ecosystem component	Hazard														
	None			Minimal			Low			Moderate			High		
	Conc.	Lemly score	Current score	Conc.	Lemly score	Current score	Conc.	Lemly score	Current score	Conc.	Lemly score	Current score	Conc.	Lemly score	Current score
Water ($\mu g\,L^{-1}$)	<1	1	1	1–2	2	2	2–3	3	3	3–5	4	4	>5	5	5
Sediment ($\mu g\,g^{-1}$)	<1	1	1	1–2	2	2	2–3	3	3	3–4	4	4	>4	5	5
Benthic invertebrate ($\mu g\,g^{-1}$)	<2	1	2	2–3	2	4	3–4	3	6	4–5	4	8	>5	5	10
Fish eggs ($\mu g\,g^{-1}$)	<3	1	3	3–5	2	6	5–10	3	9	10–20	4	12	>20	5	15
Sum		4	7		8	14		12	21		16	28		20	35
Final hazard (Lemly)		4			5–7			8–10			11–14			15–20	
Final hazard (Current)			7			8–13			14–20			21–27			28–35

TABLE 18-XV

Hazard assessment of Se at nine locations listed in Table 18-I

Site and ecosystem component	Selenium[1]	Evaluation by component		Totals for site	
		Hazard	Score	Score	Hazard
ACM					
Water	<0.5	None	1	28	Moderate
Sediment	1.0	Minimal	2		
Benthic invertebrate	7.2	High	10		
Fish eggs[2]	21.1	High	15		
UEMC					
Water	29.9	High	5	35	High
Sediment	32.5	High	5		
Benthic invertebrate	35.7	High	10		
Fish eggs	106.3	High	15		
LEMC					
Water	14.9	High	5	10	High[4]
Sediment	45.8	High	5		
Benthic invertebrate	NS[3]	–	–		
Fish eggs	NS	–	–		
TC					
Water	<0.5	None	1	18	Low
Sediment	0.8	None	1		
Benthic invertebrate	2.9	Minimal	4		
Fish eggs	16.9	Moderate	12		
USC					
Water	<0.5	None	1	22	Moderate
Sediment	0.8	None	1		
Benthic invertebrate	4.9	Moderate	8		
Fish eggs	13.2	Moderate	12		
LSC					
Water	<0.5	None	1	14	Moderate[4]
Sediment	1.7	Minimal	2		
Benthic invertebrate	NS	–	–		
Fish eggs	17.5	Moderate	12		
DVC					
Water	0.5	None	1	31	High
Sediment	5.2	High	5		
Benthic invertebrate	19.5	High	10		
Fish eggs	53.1	High	15		

Continued

TABLE 18-XV *Continued*

Site and ecosystem component	Selenium[1]	Evaluation by component		Totals for site	
		Hazard	Score	Score	Hazard
ShpC					
Water	<0.5	None	1		
Sediment	0.5	None	1	18	Low
Benthic invertebrate	2.6	Minimal	4		
Fish eggs	17.2	Moderate	12		
LBR					
Water	<0.5	None	1		
Sediment	2.1	Low	3	29	Moderate
Benthic invertebrate	5.4	High	10		
Fish eggs	25.7	High	15		

[1]Concentrations in $\mu g\,L^{-1}$ for water, $\mu g\,g^{-1}$ for sediment, benthic invertebrates, and fish eggs.
[2]Fish eggs: fish egg values converted from whole-body residues using: whole-body × 3.3 = fish egg (Lemly1995, 1996a).
[3]NS: not sampled.
[4]Based on available aquatic ecosystem components.

seems biologically reasonable and is referred to here as the offset summation approach. Similar offsets for other final hazards are given in Table 18-XIV. For the five-component protocol, the modified final hazard characterization would be, 10, no hazard; 11–19, minimal hazard; 20–28, low hazard; 29–38, moderate hazard; and 39–50, high hazard.

In the present study, fish eggs were not collected. Consequently, as a proxy we converted the geometric mean whole-body concentrations of Se to fish eggs concentrations using the conversion factor based on Lemly and Smith (1987), who reported: viscera (fish eggs or liver) to whole-body = viscera × 0.33. The conversion factor to eggs only would be 3.3 (Lemly, 1995, 1996a).

Trail Creek and Sheep Creek had low Se concentrations in most aquatic ecosystem components, a low hazard rating (Table 18-XV). Although Se concentrations were low in water and sediment at upper Slug Creek and lower Slug Creek, they were elevated in benthic invertebrates or whole-body residues converted to fish egg concentrations or both, resulting in a moderate hazard rating. Selenium concentrations in water or sediment were in the none, minimal, or low categories at Angus Creek and the lower Blackfoot River, but high in benthic invertebrates and whole-body residues, thus receiving high final hazards. Using the original Lemly (1996a) approach, these two sites would have received moderate final hazards in spite of the high score for benthic invertebrates and fish eggs. Upper East Mill Creek and Dry Valley Creek consistently had elevated Se concentrations in sediment, invertebrates, or whole-body residues, thus resulting in a high hazard rating. The hazard rating at lower East Mill Creek was considered high because of the elevated

Se concentrations in water and sediment, which no doubt would have resulted in high concentrations in invertebrates and fish had they been found.

A preliminary assessment of Se hazard in the Caribou National Forest was conducted using Se residue data in water and fish collected from 1997 to 1998 (Lemly, 1999). Lemly (1999) concluded that there was a high potential for toxic impacts to fish and wildlife associated with the Blackfoot River, its tributaries, and Blackfoot Reservoir. The results of the present study add substantially more support to the premise that Se concentrations in several aquatic ecosystem components were sufficiently elevated to cause adverse effects to aquatic resources in the Blackfoot River watershed.

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REFERENCES

Adams, W.J., 1976. The toxicity and residue dynamics of selenium in fish and aquatic invertebrates. PhD dissertation, Michigan State University, Lansing, 109 pp.

Alaimo, J., Ogle, R.S. and Knight, A.W., 1994. Selenium uptake by larval *Chironomus decorus* from a *Ruppia maritima*-based benthic/detrital substrate. Arch. Environ. Contam. Toxicol., 27: 441–448.

Allan, J.D., 1995. Stream Ecology – Structure and Function of Running Waters. Chapman and Hall, London, 388 pp.

American Public Health Association, American Water Works Association, and Water Environment Federation, 1995. Standard Methods for the Examination of Water and Wastewater (19th edn.). American Public Health Association, Washington.

Analytical Technology Incorporated Orion, 1994. Model 9346 Nitrite Electrode Instruction Manual. Analytical Technology Incorporated, Boston, 44 pp.

Brix, K.V., DeForest, D.K., Fairbrother, A. and Adams, W.J., 2000. Critical review of tissue-based selenium toxicity thresholds for fish and birds. In: Planning for End Land Use in Mine Reclamation, Proceedings of the 24th Annual British Columbia Reclamation Symposium. BC Technical and Research Committee on Reclamation, June 19–22, Williams Lake, BC, pp. 220–230.

Bryson, W.T., Garrett, W.R., Mallin, M.A., MacPherson, K.A., Partin, W.E. and Woock, S.E., 1984. Roxboro Steam Electric Plant Environmental Monitoring Studies, Vol. II, Hyco Reservoir Bioassay Studies 1982. Carolina Power and Light, New Hill, NC, 84 pp.

Campbell, K.R., 1994. Concentrations of heavy metals associated with urban runoff in fish living in stormwater treatment ponds. Arch. Environ. Contam. Toxicol., 27: 352–356.

Caribou County Sun, 1999. Toxicologist and vet say dead sheep likely died from selenium. Soda Springs, ID, November 11.

Cherry, D.S. and Guthrie, R.K., 1977. Toxic metals in surface waters from coal ash. Water Resour. Bull., 13: 1227–1236.

Dallinger, R. and Kautzky, H., 1985. The importance of contaminated food for the uptake of heavy metals by rainbow trout (*Salmo gairdneri*): a field study. Oecologia, 67: 82–89.

Dallinger, R., Prosi, F., Segner, H. and Back, H., 1987. Contaminated food and uptake of heavy metals by fish: a review and a proposal for further research. Oecologia, 73 :91–98.

DeForest, D.K., Brix, K.V. and Adams, W.J., 1999. Critical review of proposed residue-based selenium toxicity thresholds for freshwater fish. Hum. Ecol. Risk Assess., 5: 1187–1228.

Desborough, G., DeWitt, E., Jones, J., Meier, A. and Meeker, G., 1999. Preliminary mineralogical and chemical studies related to the potential mobility of selenium and associated elements in Phosphoria Formation strata, southeastern Idaho. US Geol. Surv., Open File Report, 99-129, 20 pp.

Devi, M., Thomas, D.A., Barber, J.T. and Fingerman, M., 1996. Accumulation and physiological and biochemical effects of cadmium in a simple aquatic food chain. Ecotoxicol. Environ. Saf., 33: 38–43.

Eisler, R., 1985. Selenium hazards to fish, wildlife, and invertebrates: a synoptic review. US Fish and Wildlife Service, Contaminant Hazard Reviews Report No. 5, 57 pp.

Finley, K.A., 1985. Observations of bluegills fed selenium-contaminated Hexagenia nymphs collected from Belews Lake, North Carolina. Bull. Environ. Contam. Toxicol., 35: 816–825.

Foda, A., Vandermeulen, J.H. and Wrench, J.J., 1983. Uptake and conversion of selenium by a marine bacterium. Can. J. Fish. Aquat. Sci., 40 (Supplement 2): 215–220.

Furr, A.K., Parkinson, T.F., Youngs, W.D., Berg, C.O., Gutenmann, W.H., Pakkala, I.S. and Lisk, D.J., 1979. Elemental content of aquatic organisms inhabiting a pond contaminated with coal fly ash. NY Fish Game J., 26: 154–161.

Hach Company, 1997. Hach Water Analysis Handbook (3rd edn.). Hach Company, Loveland, CO., 1309 pp.

Hamilton, S.J., 2002. Rationale for a tissue-based selenium criterion for aquatic life. Aquat. Toxicol., 57: 85–100.

Hamilton, S.J., 2003. Review of residue-based selenium toxicity thresholds for freshwater fish. Ecotoxicol. Environ. Saf., 56: 201–210.

Hamilton, S.J. and Hoffman, D.J., 2003. Trace element and nutrition interactions in fish and wildlife. In: D.J. Hoffman, B.A. Rattner, G.A. Burton and J. Cairns (eds.), Handbook of Ecotoxicology (2nd edn.). CRC Press, Boca Raton, pp. 1197–1235.

Hamilton, S.J. and Lemly, A.D., 1999. Water-sediment controversy in setting environmental standards for selenium. Ecotoxicol. Environ. Saf., 44: 227–235.

Hamilton, S.J. and Wiedmeyer, R.H., 1990. Concentrations of boron, molybdenum, and selenium in chinook salmon. Trans. Am. Fish. Soc., 119: 500–510.

Hamilton, S.J., Palmisano, A.N., Wedemeyer, G.A. and Yasutake, W.T., 1986. Impacts of selenium on early life stages and smoltification of fall chinook salmon. Trans. N. Am. Wildl. Nat. Res. Conf., 51: 343–356.

Hamilton, S.J., Muth, R.T., Waddell, B. and May, T.W., 2000. Hazard assessment of selenium and other trace elements in wild larval razorback sucker from the Green River, Utah. Ecotoxicol. Environ. Saf., 45: 132–147.

Hamilton, S.J., Holley, K.M., Buhl, K.J., Bullard, F.A., Weston, L.K. and McDonald, S.F., 2001a. The evaluation of contaminant impacts on razorback sucker held in flooded bottomland sites near Grand Junction, Colorado – 1996. Final Report. US Geol. Surv., Yankton, SD, 302 pp.

Hamilton, S.J., Holley, K.M., Buhl, K.J., Bullard, F.A., Weston, L.K. and McDonald, S.F., 2001b. The evaluation of contaminant impacts on razorback sucker held in flooded bottomland sites near Grand Junction, Colorado – 1997. Final Report. US Geol. Surv., Yankton, SD, 229 pp.

Herring, J.R. and Amacher, M.C., 2001. Chemical composition of plants growing on the Wooley Valley phosphate mine waste pile and on similar rocks in nearby Dairy Syncline, Caribou County, southeast Idaho. US Geol. Surv., Open File Report, 01-0025, 47 pp.

Herring, J.R., Wilson, S.A., Stillings, L.A., Knudsen, A.C., Gunter, M.E., Tysdal, R.G., Grauch, R.I., Desborough, G.A. and Zielinski, R.A., 2000a. Chemical composition of weathered and less weathered strata of the Meade Peak Phosphatic Shale Member of the Permian Phosphoria Formation B. Measured sections C and D, Dry Valley, Caribou County, Idaho. US Geol. Surv., Open File Report, 99-147-B, 33 pp.

Herring, J.R., Grauch, R.I., Desborough, G.A., Wilson, S.A. and Tysdal, R.G., 2000b. Chemical composition of weathered and less weathered strata of the Meade Peak Phosphatic Shale Member of the Permian Phosphoria Formation C. Measured sections E and F, Rasmussen Ridge, Caribou County, Idaho. US Geol. Surv., Open File Report, 99-147-C, 35 pp.

Herring, J.R., Castle, C.J., Brown, Z.A. and Briggs, P.H., 2001. Chemical composition of deployed and indigenous aquatic bryophytes in a seep flowing from a phosphate mine waste pile and in the associated Angus Creek drainage, Caribou County, southeast Idaho. US Geol. Surv., Open File Report, 01-0026, 20 pp.

Hilton, J.W., Hodson, P.V. and Slinger, S.J., 1982. Absorption, distribution, half-life and possible routes of elimination of dietary selenium in juvenile rainbow trout (*Salmo gairdneri*). Comp. Biochem. Physiol., 71C: 49–55.

Hunn, J.B., Hamilton, S.J. and Buckler, D.R., 1987. Toxicity of sodium selenite to rainbow trout fry. Water Res., 21: 233–238.

Kiffney, P.M. and Clements, W.H., 1993. Bioaccumulation of heavy metals by benthic invertebrates at the Arkansas River, Colorado. Environ. Toxicol. Chem., 12: 1507–1517.

Kirby, J., Maher, W. and Harasti, D., 2001a. Changes in selenium, copper, cadmium, and zinc concentrations in mullet (*Mugil cephalus*) from the southern basin of Lake Macquarie, Australia, in response to alteration of coal-fired power station fly ash handling procedures. Arch. Environ. Contam. Toxicol., 41: 171–181.

Kirby, J., Maher, W. and Krikowa, F., 2001b. Selenium, cadmium, copper, and zinc concentrations in sediments and mullet (*Mugil cephalus*) from the southern basin of Lake Macquarie, NSW, Australia. Arch. Environ. Contam. Toxicol., 40: 246–256.

Lee, D.S., Gilbert, C.R., Hocutt, C.H., Jenkins, R.E., McAllister, D.E. and Stauffer, J.R. Jr., 1980. Atlas of North American Freshwater Fishes. North Carolina State Museum of Natural History, Raliegh, NC, 854 pp.

Lemly, A.D., 1982. Response of juvenile centrarchids to sublethal concentrations of waterborne selenium. I. Uptake, tissue distribution, and retention. Aquat. Toxicol., 2: 235–252.

Lemly, A.D., 1985. Toxicology of selenium in a freshwater reservoir: implications for environmental hazard evaluation and safety. Ecotoxicol. Environ. Saf., 10: 314–338.

Lemly, A.D., 1993. Guidelines for evaluating selenium data from aquatic monitoring and assessment studies. Environ. Monit. Assess., 28: 83–100:

Lemly, A.D., 1995. A protocol for aquatic hazard assessment of selenium. Ecotoxicol. Environ. Saf., 32: 280–288.

Lemly, A.D., 1996a. Evaluation of the hazard quotient method for risk assessment of selenium. Ecotoxicol. Environ. Saf., 35: 156–162.

Lemly, A.D., 1996b. Selenium in aquatic organisms. In: W.N. Beyer, G.H. Heinz and A.W. Redmon-Norwood (eds.), Environmental Contaminants in Wildlife – Interpreting Tissue Concentrations. CRC Lewis Publ., Boca Raton, pp. 427–445.

Lemly, A.D., 1999. Preliminary assessment of selenium hazards on Caribou National Forest, Idaho. Report, US Forest Service, Blacksburg, VA, 16 pp.

Lemly, A.D. and Smith, G.J., 1987. Aquatic cycling of selenium: Implications for fish and wildlife. Fish and Wildlife Leaflet 12. US Fish and Wildlife Service, Washington, 10 pp.

Luoma, S.N., 1983. Bioavailability of trace metals to aquatic organisms – a review. Sci. Total Environ., 28: 1–22.

Maier, K.J. and Knight, A.W., 1994. Ecotoxicology of selenium in freshwater systems. Rev. Environ. Contam. Toxicol., 134: 31–48.

Maier, K.J., Nelson, C.R., Bailey, F.C., Klaine, S.J. and Knight, A.W., 1998. Accumulation of selenium by the aquatic biota of a watershed treated with seleniferous fertilizer. Bull. Environ. Contam. Toxicol., 60: 409–416.

Mann, K.H., 1972. Macrophyte production and detritus food chains in coastal waters. Mem. Ist. Ital. Idrobiol., 29: 353–383.

Montgomery Watson, 1999. Final 1998 Regional Investigation Report, Southeast Idaho Phosphate Resource Area Selenium Project. Report to Idaho Mining Association Selenium Subcommittee. Montgomery Watson, Steamboat Springs, CO.

Montgomery Watson, 2000. 1999 Interim Investigation Data Report, Southeast Idaho Phosphate Resource Area Selenium Project. Report to Idaho Mining Association Selenium Subcommittee. Montgomery Watson, Steamboat Springs, CO.

Montgomery Watson, 2001a. Draft 1999–2000 Regional Investigation Data Report for Surface Water, Sediment and Aquatic Biota Sampling Activities, September 1999. Southeast Idaho Phosphate Resource Area Selenium Project. Report to Idaho Mining Association Selenium Subcommittee. Montgomery Watson, Steamboat Springs, CO.

Montgomery Watson, 2001b. Draft 1999–2000 Regional Investigation Data Report for Surface Water, Sediment and Aquatic Biota Sampling Activities, May–June 2000, Southeast Idaho Phosphate Resource Area Selenium Project. Report to Idaho Mining Association Selenium Subcommittee. Montgomery Watson, Steamboat Springs, CO.

Moore, S.B., Winckel, J., Detwiler, S.J., Klasing, S.A., Gaul, P.A., Kanim, N.R., Kesser, B.E., DeBevec, A.B., Beardsley, K. and Puckett, L.K., 1990. Fish and Wildlife Resources and Agricultural Drainage in the San Joaquin Valley, California (vol. 2). San Joaquin Valley Drainage Program, Sacramento, CA.

Ney, J.J. and Van Hassel, J.H., 1983. Sources of variability in accumulation of heavy metals by fishes in a roadside stream. Arch. Environ. Contam. Toxicol., 12: 701–706.

National Research Council, 1980. Recommended daily allowances (9th edn.). National Research Council, Food and Nutrition Board, Committee on Dietary Allowances. National Academy Press, Washington, 187 pp.

Odum, E.P. and de la Cruz, A.A., 1967. Particulate organic detritus in a Georgia salt marsh-estuarine ecosystem. In: G.H. Lanff (ed.), Estuaries. American Association for the Advancement of Science Publication 83, pp. 383–388.

Olson, O.E., 1986. Selenium toxicity in animals with emphasis on man. J. Am. College Toxicol., 5: 45–70.

Orion Research, 1990. Model 95–12 Ammonia Electrode Instruction Manual. Orion Research Incorporated, Boston, 36 pp.

Orion Research, 1991. Model 93-07 Nitrate Electrode Instruction Manual. Orion Research Incorporated, Boston, 31 pp.

Piper, D.Z., Skorupa, J.P., Presser, T.S., Hardy, M.A., Hamilton, S.J., Huebner, M., and Gulbrandsen, R.A., 2000. The Phosphoria Formation at the Hot Springs Mine in southeast Idaho: a source of selenium and other trace elements to surface water, ground water, vegetation, and biota. US Geol. Surv., Open File Report, 00–050, 73 pp.

Platts, W.S. and Martin, S.B., 1978. Hydrochemical influences on the fishery within the phosphate mining area of eastern Idaho. US Department of Agriculture Forest Service Research Note INT-246. Intermountain Forest and Range Experiment Station, Odgen, UT, 15 pp.

Presser, T.S., Sylvester, M.A. and Low, W.H., 1994. Bioaccumulation of selenium from natural geologic sources in western states and its potential consequences. Environ. Manage., 18: 423–436.

Rand, G.M. and Petrocelli, S.R. (eds.), 1985. Fundamentals of Aquatic Toxicology: Methods and Applications. Hemisphere, Washington, 666 pp.

Rich and Associates, 1999. FMC phosphate mine expansion. Fishery Resources Technical Report, A.A. Rich and Associates, San Anselmo, CA, 125 pp. plus appendices.

Saiki, M.K., Jennings, M.R. and Brumbaugh, W.G., 1993. Boron, molybdenum, and selenium in aquatic food chains from the lower San Joaquin River and its tributaries, California. Arch. Environ. Contam. Toxicol., 24: 307–319.

Statistical Analysis System, Inc., 1985. SAS Language Guide for Personal Computers (6th edn.). SAS Institute, Inc., Cary, NC.

Seelye, J.G., Hesselberg, R.J. and Mac, M.J., 1982. Accumulation by fish of contaminants released from dredged sediments. Environ. Sci. Technol., 16: 459–464.

Stephan, C.E., Mount, D.I., Hansen, D.J., Gentile, J.H., Chapman, G.A. and Brungs, W.A., 1985. Guidelines for deriving numerical national water quality criteria for the protection of aquatic organisms and their uses. US Environ. Protection Agency Report PB85-227049, 98 pp.

Stephens, D., Waddell, B., DuBois, K. and Peterson, E., 1997. Field screening of water quality, bottom sediment, and biota associated with the Emery and Scofield Project areas, central Utah, 1994. US Geol. Surv., Water-Resources Investigations Report, 96-4298, 39 pp.

Teal, J.M., 1962. Energy flow in a salt march ecosystem of Georgia. Ecology, 43: 614–624.

Thurow, R., Wishard, L., Christensen, W. and Aebersold, P., 1981. Federal aid to fish and wildlife restoration, Job Completion Report Project F-73-R-3. ID Depart. Fish Game, Boise, 243 pp.

TRC Environmental Corporation, 1999. Maybe Canyon site investigation, Caribou National Forest, Caribou County, Idaho. Report to US Forest Service. TRC Environmental Corporation, Englewood, CO, 122 pp. plus appendices.

US Environmental Protection Agency, 1983. Methods for chemical analysis of water and wastes. Publication EPA 600/4-79-020.

US Environmental Protection Agency, 1987. Ambient water quality criteria for selenium – 1987. Publication EPA 440/5-87-006.

US Environmental Protection Agency, 1998a. National recommended water quality criteria; republication. Fed. Reg., 63: 68354–68364.

US Environmental Protection Agency, 1998b. Report on the Peer Consultation Workshop on Selenium Aquatic Toxicity and Bioaccumulation. Publication EPA 822-R-98-007, Washington, 261 pp.

US Environmental Protection Agency, 1999. National recommended water quality criteria – correction. Publication EPA 822-Z-99-001.

Wiener, J.G. and Giesy, J.P. Jr., 1979. Concentrations of Cd, Cu, Mn, Pb, and Zn in fishes in a highly organic softwater pond. J. Fish. Res. Bd. Can., 36: 270-279.

Woock, S.E., 1984. Accumulation of Selenium by Golden Shiners *Notemigonus crysoleucas:* Hyco Reservoir N.C. cage study 1981-1982. Carolina Power and Light Company, New Hill, NC, 19 pp.