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SELENIUM IN THE BLACKFOOT, SALT, AND BEAR RIVER WATERSHEDS

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Abstract. Nine stream sites in the Blackfoot River, Salt River, and Bear River watersheds in southeast Idaho, USA were sampled in May 2001 for water, surficial sediment, aquatic plants, aquatic invertebrates, and fish. Selenium was measured in these aquatic ecosystem components, and a hazard assessment was performed on the data. Water quality characteristics such as pH, hardness, and specific conductance were relatively uniform among the nine sites. Of the aquatic components assessed, water was the least contaminated with selenium because measured concentrations were below the national water quality criterion of 5 μg/L at eight of the nine sites. In contrast, selenium was elevated in sediment, aquatic plants, aquatic invertebrates, and fish from several sites, suggesting deposition in sediments and food web cycling through plants and invertebrates. Selenium was elevated to concentrations of concern in fish at eight sites (>4 μg/g in whole body). A hazard assessment of selenium in the aquatic environment suggested a moderate hazard at upper Angus Creek (UAC) and Smoky Creek (SC), and high hazard at Little Blackfoot River (LiB), Blackfoot River gaging station (BGS), State Land Creek (SLC), upper (UGC) and lower Georgetown Creek (LGC), Deer Creek (DC), and Crow Creek (CC). The results of this study indicate that selenium concentrations from the phosphate mining area of southeast Idaho were sufficiently elevated in several ecosystem components to cause adverse effects to aquatic resources in southeastern Idaho.

Keywords: selenium, phosphate mining, Idaho, hazard assessment

1. Introduction

Phosphorus is present in economically mineable quantities in organic-rich black shales of the Permian Phosphoria Formation, which constitutes the Western U.S. Phosphate Field and includes portions of Idaho, Wyoming, and Utah. There are four active open pit mines in the southeast Idaho Phosphate District that produce phosphate from the Meade Peak Phosphatic Shale Member, and 11 inactive mines (MW, 1999). Most mining of these phosphatic shales is by open-pit or contour strip surface mining, and waste materials are generally deposited on the surface in tailings piles, ponds, landfills, and dumps. Many of the waste piles have drainage systems to move surface water and groundwater away from waste-rock piles. These drainage systems transfer leachates from mining areas to surface waters, eventually draining into tributaries, and later, rivers such as the Blackfoot, Salt, and Bear.

Thus, water movement can carry toxic inorganic elements to aquatic and terrestrial ecosystems.

The Blackfoot, Salt, and Bear river watersheds have several active and inactive phosphate mines that could adversely affect aquatic resources in tributaries of these rivers. As early as 1978 concerns were expressed about contamination of the Blackfoot River and its tributaries by inorganic elements released from phosphate mining (Platts and Martin, 1978). Recent concerns about the potential impact on aquatic and terrestrial ecosystems from phosphate mining have been the subject of several reports (MW, 1999, 2000, 2001a,b; MWH, 2002a,b; Tetra Tech, 2002a,b). Several investigations by the U.S. Geological Survey (USGS) have reported the chemical composition of weathered and less-weathered strata of the Meade Peak Phosphoatic Shale (e.g., Desborough *et al.*, 1999; Herring *et al.*, 2000a,b). Other USGS investigations have reported elevated inorganic element concentrations in aquatic bryophytes and terrestrial plants that were influenced by mining (Herring and Amacher, 2001; Herring *et al.*, 2001).

Release of toxic inorganic elements from phosphate mining in southeast Idaho and accumulation in the food chain has resulted in adverse biological effects. In recent years, seven horses in the Dry Valley and Woddall areas were euthanized, and 60–80 sheep died in the Caribou National Forest on the old Stauffer Mine site due to selenium poisoning (Caribou County Sun, 1999). Twenty-six dead sheep were found at the south end of Rasmussen Ridge Mine near a spring or seep at an overburden ore site. In 2003, an additional 327 sheep were killed from eating high selenium vegetation on claimed mining land above the Conda Mine (Steele, 2003). Elevated concentrations of selenium and other inorganic elements have been reported in samples of fish and aquatic invertebrates from streams below phosphate mining activities (MW, 1999, 2001a,b). Recent USGS reports suggest that selenium concentrations in fish and wildlife in the Blackfoot River watershed were sufficiently elevated to cause adverse effects in sensitive fish species (Piper *et al.*, 2000; Hamilton *et al.*, 2002; Hamilton and Buhl, 2003a,b).

The purpose of this study was to determine the concentrations of selenium in water, surficial sediment, aquatic plants, aquatic invertebrates, and fish from streams in the Blackfoot, Salt, and Bear river watersheds in southeast Idaho near phosphate mining operations. This information was used in a hazard assessment of the potential effects of selenium.

2. Methods

Samples of water, surficial sediment, aquatic plants, aquatic invertebrates, and fish were collected from nine sites in the Blackfoot, Salt, and Bear river watersheds located in southeast Idaho (Figure 1). Sample collection occurred in May 2001, and was a joint effort of the USGS and the U.S. Forest Service (USFS).

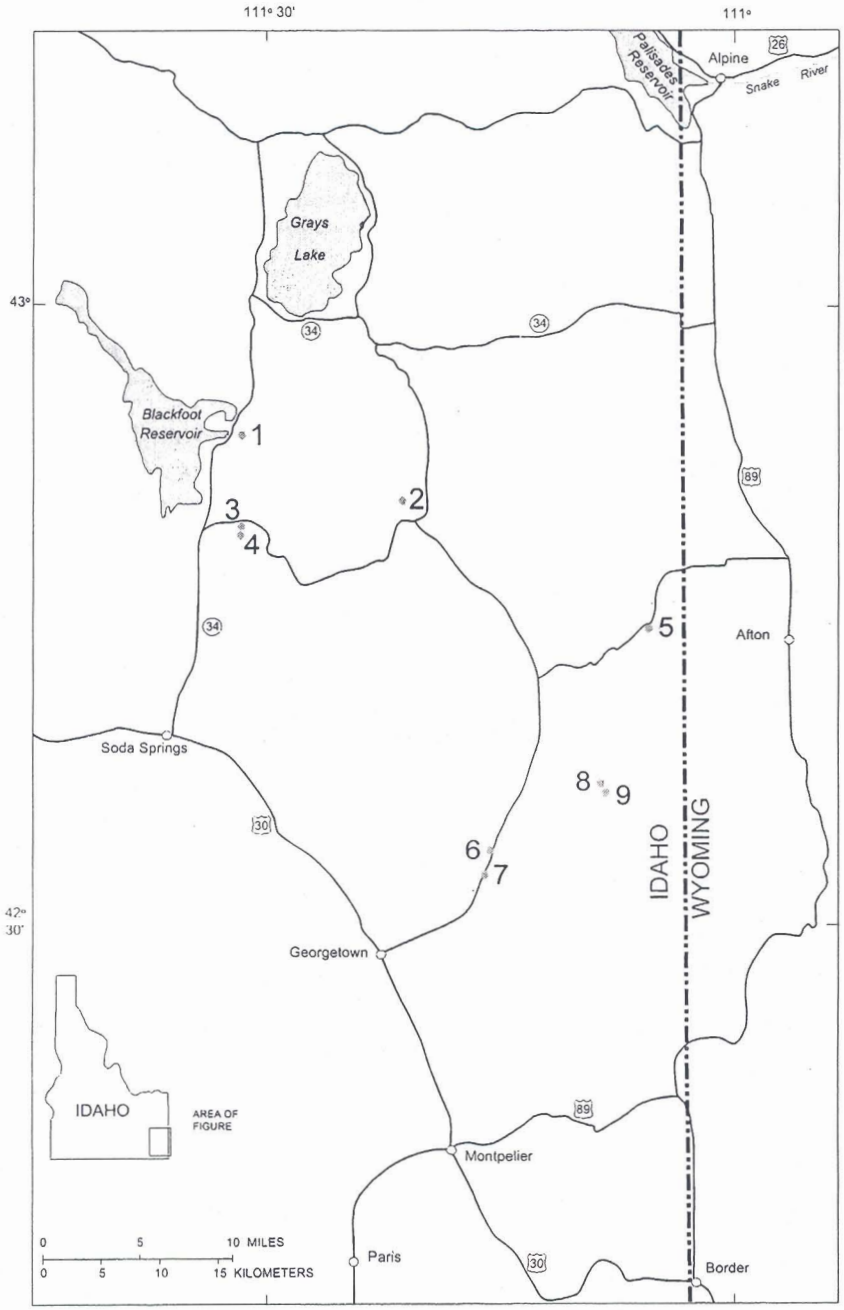

Figure 1. Map of sample sites: 1, Little Blackfoot River; 2, upper Angus Creek; 3, Blackfoot River gaging station; 4, State Land Creek; 5, Smoky Creek; 6, upper Georgetown Creek; 7, lower Georgetown Creek; 8, Deer Creek; 9, Crow Creek (map source: modified from Herring *et al.*, 2001.).

2.1. Site description

The collection sites were as follows:

1. The Little Blackfoot River (LiB) site was located about 1 km above its confluence with Blackfoot Reservoir and about 1 km southeast of State Highway 34. The sampling site was below the active Enoch Valley Mine and the inactive Henry and Wooley Valley mines. The land along the river was primarily open grassland with light grazing.
2. The upper Angus Creek (UAC) site was located about 2 km below the headwater seep in Little Long Valley accessed by Forest Route 205 (USFS map, Caribou National Forest, Montpelier and Soda Springs, Districts, 1988) and a mining road. The sampling site was below the Wooley Valley Mine Unit 4 waste-rock pile, and was composed primarily of grassland habitat with sparse forbs and no grazing activity. Sample collection was in an open area of forbs, grass, and willows. The creek had been previously impacted by upslope runoff of water and sediment from Wooley Valley Mine Unit 4 waste-rock pile located about 2 km upstream.
3. The Blackfoot River gaging station (BGS) site was located 20 m upstream of the crossing of the river by the private haul road and the railroad tracks, accessed from the Blackfoot River Road. The sampling site was below several active and inactive mines. The land on either side of the river was composed of grass and sagebrush and had moderate grazing.
4. The State Land Creek (SLC) site was located about 1.2 km from the private haul road, accessed from the Blackfoot River Road. The site was approximately 0.5 km above the confluence with the Blackfoot River, and below the inactive Conda Mine. Sample collection was in a generally open area of forbs, grass, and spare pine trees with some grazing.
5. The Smoky Creek (SC) site was located in Smoky Canyon about 1.5 km inside the USFS boundary on Forest Route 110, and about 3 km above the confluence with Tygee Creek, which flows into the Salt River. The sampling site was not impacted by mining activity, and the land around the stream was primarily riparian with numerous beaver ponds above and below the collection site. The roadway bordered one side of the creek and the opposite side was heavily forested with no grazing evident.
6. The upper Georgetown Creek (UGC) site was located adjacent to the public parking lot located just inside of the USFS boundary and about 0.5 km above the abandoned mine processing plant in Georgetown Canyon (Forest Route 102) and within the Georgetown Canyon Mine area. The site was approximately 15 km above the confluence with the Bear River, and below the inactive Georgetown Mine. The land on either side of the road was ungrazed, riparian habitat.
7. The lower Georgetown Creek (LGC) site was located about 5 km downstream of the UGC site and below the abandon mine processing plant, but within the

Georgetown Canyon Mine area. The sampling site was below the inactive Georgetown and Montpelier mines. The roadway bordered one side of the stream and the opposite stream bank bordered a steep forested slope with no grazing evident.

8. The Deer Creek (DC) site was located adjacent to Forest Route 111 and about 0.5 km upstream of its confluence with Crow Creek (CC), which flows into the Salt River. The sampling site was not impacted by mining activity, but the upper portion of the DC watershed has outcroppings of Phosphoria (Figure 2 in Tetra Tech, 2002a). The land along the stream was primarily riparian with some forbs, shrubs, and grass. Sample collection was upstream of roadway influences, but the area had light grazing.
9. The CC site was located on USFS land adjacent to Forest Service Route 111 and upstream of the confluence of DC and an in-stream crossing. The sampling site was not impacted by mining activity, but the upper portion of the CC watershed has outcroppings of Phosphoria (Figure 2 in Tetra Tech, 2002a). The sampling site was not impacted by mining activity, but the land along the stream was primarily riparian with some shrubs nearby and light camping activity.

2.2. Sample Collection

Samples of water, surficial sediment, aquatic plants, aquatic invertebrates, and fish were collected in May 2001 at each of nine stream sites. Water sample bottles were conditioned by immersion in site water three times. Water samples were collected by grab sampling. At a mobile laboratory, water was filtered through a 0.45 μm polycarbonate filter using standard sampling techniques. A 200-ml sample of each filtered water was collected in an acid-cleaned polyethylene bottle for analysis of selenium concentrations and acidified with ultrapure hydrochloric acid (HCl). A reagent blank was collected for analysis of selenium concentrations and consisted of deionized water from a mobile laboratory combined with the acid preservative. All samples were stored frozen.

One sediment sample was collected at each site using a plastic scoop to gently acquire surficial sediments including detritus, but not pebbles or plant material. The scoop and acid-cleaned sample container were rinsed in ambient water for sufficient time to condition the equipment to ambient conditions prior to sample collection. After sediments settled, excess water was discarded and the sample stored frozen and used for analysis of selenium and mercury concentrations.

Submerged aquatic plants (white-water buttercup, *Ranunculus longirostris*) were collected by hand from each site. The composite sample consisted of leaf whorls removed from stems using plastic or stainless steel forceps. Additional samples of leaves and stems (minus roots) were collected at the LiB and CC sites for comparison with leaf-only samples. At the UGC site no white-water buttercup

could be found, so a different submerged macrophyte was collected (Hypericum). One composite plant sample was collected from each site, squeezed to remove excess water, weighed, bagged in Whirl-Pak bags, labeled, stored frozen, and analyzed for selenium concentration.

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. Invertebrate samples were separated by taxa group and weighed by taxa group. One half of the weight of each taxa group was combined as a composite invertebrate sample and analyzed for selenium concentration.

Fish were collected by electrofishing with a Coffelt Mark-10 electroshocker provided and operated by the USFS, Caribou National Forest, Soda Springs, ID. The anode and cathode wands were rinsed in ambient water for sufficient time to condition the equipment to ambient conditions. Fish samples were collected from each site, euthanized with MS-222 (tricaine methanesulfonate), identified to species if possible, measured for total length and weight, bagged in Whirl-Pak bags, labeled with identification information, and stored frozen. When possible, one or more fish of each species from each site was analyzed for selenium concentrations in whole body.

2.3. Water quality analyses and flow measurement

Site water was analyzed *in situ* for the following general water quality characteristics: conductivity, pH, temperature, dissolved oxygen, and percent saturation of dissolved oxygen. Flow measurements were taken using a global flow probe (FP101), except at the BGS. At the gaging station, the flow was recorded from the USGS web site (USGS gage 1306000, Blackfoot River above reservoir near Henry, Idaho; http://waterdata.usgs.gov/nwis/discharge). Water samples (~1 l) at each site were collected and analyzed for general water quality characteristics in a mobile laboratory according to standard methods (APHA *et al.*, 1995).

Immediately after arrival of the site water at the mobile laboratory, the following water quality characteristics were measured in unfiltered water: conductivity, pH, alkalinity, hardness, calcium, magnesium, and temperature. A subsample of 200 ml water was collected and stored at 4 °C with no preservative, and transported to the Columbia Environmental Research Center Field Research Station, Yankton, SD, for analysis of sulfate and chloride. A second subsample of 125 ml water was collected, acidified with 0.5 ml concentrated sulfuric acid (H_2SO_4), and transported to Yankton for analysis of ammonia concentrations. All water quality characteristics were measured according to standard methods (APHA *et al.*, 1995), except ammonia and chloride. Ammonia was measured using ion-selective electrodes and following the procedures for low-concentration measurements of the electrode manufacturer (Orion Research, 1990, 1991; ATI

Orion, 1994). Chloride was measured by the mercuric nitrate titration method (Hach Company, 1997).

2.4. Selenium analysis

Water, surficial sediment, aquatic plants, aquatic invertebrates, and fish were analyzed for selenium concentrations by atomic absorption spectroscopy graphite furnace (AA-GF) at the Research Triangle Institute (RTI), Research Triangle Park, NC. 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, duplicate samples, certified reference materials, and spiked samples. Analysis of selenium concentrations was based on U.S. Environmental Protection Agency (USEPA) method 7740 (USEPA, 1983). Results were reported on a dry weight basis for analysis of sediment, aquatic plant, aquatic invertebrate, and fish samples.

Water, surficial sediment, aquatic plants, aquatic invertebrates, and fish were analyzed for mercury concentrations by cold vapor atomic absorption at the RTI. Analyses incorporated the appropriate QA/QC described above. Analysis of mercury was based on USEPA method 7174A (USEPA, 1983). Results are reported on a dry weight basis for analysis of sediment, aquatic plant, aquatic invertebrate, and fish samples.

2.5. Statistical analyses

Data were analyzed (SAS, 2002) to determine the relations among various measures made during the study. Pearson correlation analyses were used to test for relations among water quality characteristics, and selenium concentrations in water, sediment, aquatic plant, aquatic invertebrate, and fish. For fish residue data for each sample location, the geometric mean was used in correlation analyses with other variables.

The nonparametric Friedman test (Conover, 1980) ranked the streams from highest selenium concentrations to lowest using ecosystem components (sediment, plant, invertebrate, and fish). Significant differences ($p = 0.05$) among streams were determined with Friedman's multiple comparison test.

3. Results

3.1. Water quality

Water quality characteristics were relatively uniform among the sites, with two exceptions (Table I). CC had elevated chloride, and LiB had elevated sulfate and

TABLE I
Water quality characteristics measured in water

Measure	Site[a]								
	LiB	UAC	BGS	SLC	SC	UGC	LGC	DC	CC
pH	7.6	8.2	8.5	8.0	7.9	8.2	8.2	8.4	8.7
Conductivity (μmhos/cm)	880	610	390	600	590	500	620	530	1050
Hardness (mg/L as $CaCO_3$)	344	227	174	206	203	201	184	212	220
Calcium (mg/L)	83	66	50	60	53	60	50	56	61
Magnesium (mg/L)	33	15	12	14	17	12	14	18	16
Alkalinity (mg/L as $CaCO_3$)	238	175	166	193	186	185	176	207	200
Chloride (mg/L)	60	4	3	12	4	<2	<2	2	173
Sulfate (mg/L)	113	54	10	11	14	18	8	6	19
Un-ionized ammonia (mg/L NH_3-N)	<0.001	0.001	0.004	<0.001	<0.001	<0.001	<0.001	0.001	0.003
Total ammonia (mg/L as N)	<0.01	0.02	0.04	0.02	0.02	0.02	0.02	0.02	0.02
Dissolved oxygen (mg/L)	6.7	8.9	9.3	8.2	9.3	10.2	10.7	10.8	10.5
% Saturation dissolved oxygen	69	79	99	73	79	81	93	100	116
Discharge (cfs)	1.95	0.58	54	0.07	0.60	4.97	24.84	2.74	13.75

$n = 1$; the symbol (<) denotes values below limit of measurement.
[a]LiB: Little Blackfoot River, UAC: upper Angus Creek, BGS: Blackfoot River gaging station, SLC: State Land Creek, SC: Smoky Creek, UGC: upper Georgetown Creek, LGC: lower Georgetown Creek, DC: Deer Creek, CC: Crow Creek.

slightly elevated chloride relative to the other seven sites. The nine sites were well oxygenated at the time of sampling (Table I).

3.2. Selenium

The results of QA/QC sample analysis by AA-GF at RTI for the determination of selenium concentrations are given in Table II. The procedure blank had background concentration less than the LOD, which indicated no contamination from reagents or sample handling. The percent relative standard deviation (duplicate preparation and analysis) ranged from <LOD to 11%, which indicated consistent sample handling during preparation, digestion, and analysis. Percent recovery of selenium from certified material ranged from 83 to 108%, which indicated the digestion and analysis procedure accurately measured selenium concentrations. Percent recovery

TABLE II

Quality assurance and quality control measures of selenium analysis of water, sediment, aquatic plants, aquatic invertebrates, and fish

	Ecosystem component				
Measure	Water	Sediment	Aquatic plant	Aquatic invertebrate	Fish
Limit of detection (LOD)	2 μg/L	0.5 μg/g	0.5 μg/g	0.5 μg/g	0.5 μg/g
Procedural blank	<LOD	<LOD	<LOD	<LOD	<LOD
% RSD[a]	<LOD	11	2.3	7.0	4.8 (2.0)
% Recovery of reference material	89[b]	83[c]	NR[d,e]	108[f]	108[f]
% Recovery of digested spike	93	82	80	84	110 (4)

$n = 1$ for water, sediment, aquatic plants, and aquatic invertebrates; $n = 2$ for fish (mean and standard error in parentheses); <: less than.

[a]%RSD: percent relative standard deviation for duplicate preparation and analysis.

[b]Leeman Labs commercial standard solution (lot number 480801).

[c]National Institute of Standards and Technology (NIST) standard reference material 2709 (San Joaquin soil; 1.57 μg/g).

[d]NR: not reported.

[e]NIST standard reference material 1547 (peach leaves, 0.12 μg/g).

[f]National Resource Council of Canada standard reference material TORT-2 (lobster hepatopancreas, 5.63 μg/g).

of selenium from samples spiked before digestion ranged from 80 to 110%, which indicated the digestion procedure did not alter the amount of spiked selenium in the sample, i.e., suggested no loss of selenium during digestion.

3.3. Water

Selenium concentrations in water from eight sites were less than the LOD (<2 μg/L), but elevated at UGC, which contained 11 μg/L (Table III). Although UGC water contained elevated selenium, it was not among the highest in other inorganic element concentrations (Hamilton and Buhl, 2003b).

3.4. Sediment

Selenium concentrations in surficial sediment were relatively low at BGS, UAC, SC, and LiB ($\leq$2 μg/g), moderately elevated at the SLC and CC (2.1 μg/g), and elevated at DC and UGC (4.5 μg/g) and LGC (7.5 μg/g) (Table III). Based on selenium concentrations, the streams from highest concentration to lowest were: LGC, UGC, DC, CC, SLC, LiB, UAC, SC, BGS.

TABLE III

Selenium concentrations (μg/L for water and μg/g dry weight for sediment, aquatic plants, and aquatic invertebrates) in water sediment, aquatic plants, and aquatic invertebrates

Ecosystem component	Site[a]								
	LiB	UAC	BGS	SLC	SC	UGC	LGC	DC	CC
Water	<2	<2	<2	<2	<2	11	<2	<2	<2
Sediment	1.8	1.2	1.0	2.1	1.2	4.5	7.5	4.5	2.1
Aquatic plant	1.8 (2.0)	2.8	7.2	4.3	2.5	3.7	1.6	4.3	4.6 (3.1)
Aquatic invertebrate	5.4	5.0	10.8	9.7	4.1	9.3	7.8	8.7	6.7

Concentrations in parentheses are for leaves and stems. $n = 1$; <: less than limit of detection.
[a]LiB: Little Blackfoot River, UAC: upper Angus Creek, BGS: Blackfoot River gaging station, SLC: State Land Creek, SC: Smoky Creek, UGC: upper Georgetown Creek, LGC: lower Georgetown Creek, DC: Deer Creek, CC: Crow Creek.

3.5. Aquatic Plants

Selenium concentrations in aquatic plants were relatively low at LGC and LiB (1.6–1.8 μg/g), intermediate at SC, UAC, UGC, SLC, DC, and CC (2.5–4.6 μg/g), and high at BGS (7.2 μg/g) (Table III). Selenium concentrations in plants at BGS (7.2 μg/g) and LGC (1.6 μg/g) seemed inconsistent with selenium concentrations in sediments at those two sites (1.0 and 7.5 μg/g, respectively). LGC contained the highest selenium concentration in sediment of all the sites, and the lowest selenium concentration in plants. Based on selenium concentrations, the streams from highest concentration to lowest were: BGS, CC, SLC, DC, UGC, UAC, SC, LiB, LGC. Selenium concentrations in aquatic plants were not significantly correlated with those in sediments ($r = -0.37$, $p = 0.32$, $n = 9$), which was probably due to the inconsistent selenium concentrations between aquatic plants and sediments at the BGS and LGC.

3.6. Aquatic Invertebrates

Selenium concentrations in aquatic invertebrates were relatively low at SC, UAC, and LiB (4.1–5.4 μg/g), intermediate at CC, LGC, and DC (6.7–8.7 μg/g), and high at UGC, SLC, and BGS (9.3–10.8 μg/g) (Table III). Selenium concentrations in invertebrates were significantly correlated with selenium concentrations in aquatic plants ($r = 0.70$, $p = 0.04$, $n = 9$), but not with sediments ($r = 0.27$, $p = 0.48$, $n = 9$).

3.7. Fish

Nine fish species were collected at the nine stream sites, but no one species was collected at all nine sites (Table IV). Fish collected included cutthroat trout

TABLE IV
Selenium concentrations (μg/g dry weight) in whole-body fish ($n = 1$)

	Site[a]								
Species	LiB	UAC	BGS	SLC	SC	UGC	LGC	DC	CC
Brook trout	–[b]	–	–	–	–	9.8	6.7[c], 6.9[d]	–	–
Cutthroat trout	–	6.6	12.2	–	5.0[c], 3.5[d]	–	–	9.3[c], 11.0[d]	–
Brown trout	–	–	–	–	–	–	–	–	9.7
Mottled sculpin	–	6.2	12.3	–	–	–	–	12.0	8.2
Longnose dace	–	–	10.9	–	–	–	–	–	10.8[c], 13.4[d]
Speckled dace	5.8	–	–	15.2	–	–	–	–	–
Redside shiner	–	–	13.6	–	–	–	–	–	–
Other	9.8[e]	–	11.1[f]	–	–	–	–	–	–
Geometric mean	7.6	6.4	12.0	15.2	4.2	9.8	6.8	11.5	10.4

[a]LiB: Little Blackfoot River, UAC: upper Angus Creek, BGS: Blackfoot River gaging station, SLC: State Land Creek, SC: Smoky Creek, UGC: upper Georgetown Creek, LGC: lower Georgetown Creek, DC: Deer Creek, CC: Crow Creek.
[b]Not collected.
[c]Young of year.
[d]Subadult.
[e]Unknown minnow.
[f]Unknown chub.

(*Oncorhynchus clarki*), brook trout (*Salvelinus fontinalis*), brown trout (*Salmo trutta*), mottled sculpin (*Cottus bairdi*), longnose dace (*Rhinichthys cataractae*), speckled dace (*Rhinichthys osculus*), redside shiner (*Richardsonius balteatus*), an unknown minnow, and an unknown chub. The most commonly collected fish were cutthroat trout (four sites) and mottled sculpin (four sites). Only one fish species was collected at each of four sites in spite of substantial electrofishing effort: speckled dace at SLC, cutthroat trout at SC, and brook trout at UGC and LGC.

Geometric mean selenium concentrations in whole-body fish were relatively low at SC, UAC, and LGC (4.2–6.8 μg/g), intermediate at LiB, UGC, and CC (7.7–10.4 μg/g), and high at DC, BGS, and SLC (11.5–15.2 μg/g) (Table IV). Selenium concentrations in fish were significantly correlated with selenium concentrations in aquatic plants ($r = 0.69$, $p = 0.04$, $n = 9$) and aquatic invertebrates ($r = 0.82$, $p = 0.006$, $n = 9$), but not in sediments ($r = -0.04$, $p = 0.92$, $n = 9$). In general, selenium concentrations in young-of-year cutthroat trout and longnose dace were

similar to those in subadult fish (Table IV). Based on selenium concentrations, the streams from highest concentration to lowest were: SLC, BGS, DC, CC, UGC, LiB, LGC, UAC, SC.

3.8. Streams

There were significant differences among streams based on selenium concentrations in water, sediment, plant, invertebrate, and fish using the Friedman test. Streams were ranked from highest selenium concentration to lowest as follows (streams with lower case letters in common are not significantly different): UGC_a, SLC_a, BGS_a, DC_{ab}, CC_{ab}, LGC_{abc}, LiB_{bc}, UAC_c, SC_c. Testing the same rankings based on selenium concentrations, but without the water component (only UGC water contained measurable selenium in water), streams were ranked from highest selenium concentration to lowest as follows (streams with lower case letters in common are not significantly different): SLC_a, BGS_a, DC_a, UGC_{ab}, CC_{ab}, LGC_{abc}, LiB_{bc}, UAC_c, SC_c. Only the position of UGC in the ranking changed between the two approaches.

4. Discussion

4.1. Water

UGC contained substantially elevated selenium concentrations in water, whereas the other eight sites contained concentrations below the limit of detection. Selenium in UGC was substantially higher than the current national water quality criterion for the protection of aquatic life of 5 μg/L (USEPA, 1987). Most of the nine stream sites contained inorganic element concentrations in water that were below the limit of detection (Hamilton and Buhl, 2003b). Consequently, no one stream stood out as being impacted by water-borne inorganic elements other than selenium in UGC.

The Idaho Mining Association Selenium Subcommittee (Selenium Subcommittee) investigated concentrations of selenium, cadmium, manganese, nickel, vanadium, and zinc in water from numerous sites in the Southeast Idaho Phosphate Resource Area during 1998–2000 and concluded that selenium was the major contaminant of potential concern (MW, 1999, 2000, 2001a,b). In May 1998, selenium concentrations in water at 12 of 37 stream sites exceeded the USEPA criteria of 5 μg/L, whereas in September 1998 only one stream, East Mill Creek (32 μg/L), exceeded the USEPA criteria (MW, 1999).

Selenium concentrations in water at sites sampled by Montgomery Watson (MW) in May 1998 that were located close to our sites included the following: <1 μg/L in the LiB, 3 μg/L in Angus Creek (about 2 km below our site), 7 μg/L in the Blackfoot River at the gaging station, 9 μg/L in SLC (about 2 km upstream of our site), 1 μg/L in SC, and 6 μg/L in Georgetown Creek (about 6 km below our LGC site) (MW, 1999). Most of these sites contained selenium concentrations

less than in the September sampling (MW, 1999). Overall, our selenium concentrations in water were lower than those measured in 1998 by MW, except at UGC.

In May 1999, MW conducted additional water sampling, but only one site was close to the sites in the current study. They reported 8.2 μg/L at the BGS (MW, 1999), whereas in September 1999 they reported <1 μg/L selenium in water at the BGS, LiB, Angus Creek, and SC (MW, 2001a).

Selenium concentrations in water at sites sampled by MW in May 2000 that were close to our sites included the following: <1 μg/L in the LiB, 2 μg/L in Angus Creek, 4 μg/L in the Blackfoot River at the gaging station, 10 μg/L in SLC, 7 μg/L in Georgetown Creek, and 18 μg/L downstream Georgetown Creek (MW, 2001b). Overall, our selenium concentrations in water were lower than those measured in 2000, except at UGC. The high selenium concentration in SLC water reported by MW was probably due to selenium loading from two unnamed tributaries downstream of waste rock dumps (each contained 160 μg/L selenium, MW, 2001b).

In 2001, Tetra Tech collected water samples for analysis of elemental concentrations at 31 sites in the Blackfoot, Bear, and Salt river watersheds in May, June, and September 2001, and six of those sites were close to the locations sampled in the present study (Tetra Tech, 2002a). Tetra Tech reported similar selenium concentrations as those in the present study for LiB (close to our site), SLC (about 4 km upstream of our site), middle Angus Creek (about 3 km downstream of our site), SC (close to our site), DC (close to our site), and CC (close to our site) (Tetra Tech, 2002a). They reported 1.9 μg/L in May, 1.5 μg/L in June, and 2.0 μg/L in September in Georgetown Creek, which was similar to our LGC site (<2 μg/L), but lower than our UGC site (11 μg/L). This difference in selenium concentrations suggests relatively high variability in selenium concentrations due to dilution from surface and ground water sources. Overall, Tetra Tech (2002a) reported that most selenium loading of watersheds was occurring in the Blackfoot River watershed, less loading in the Salt River watershed coming mostly from Sage Creek with lesser amounts from DC and CC, and lower loading in the Bear River watershed coming from Georgetown Creek and Montpelier Creek. They also reported that chronic selenium criteria was exceeded at least once in Georgetown Creek, Sage Creek, East Mill Creek, Spring Creek, Maybe Creek, Dry Valley Creek, Trail Creek, SLC, and the Blackfoot River. Much of the selenium in those surface waters came from elevated selenium concentrations in seeps, ponds, and drains associated with waste rock piles (MWH, 2002a).

In a follow-up study in May 2002, Tetra Tech sampled water at 10 sites they previously sampled in 2001, but only two were close to those in the present study (Tetra Tech, 2002b). They reported selenium concentrations in water were 3.0 μg/L at SLC and 2.0 μg/L at Georgetown Creek (about 7 km downstream of LGC), which were higher than concentrations we found. They attributed the higher selenium concentrations and selenium loading in the streams they monitored to the higher

snowmelt runoff. They also concluded that selenium loading occured primarily in the spring and was associated with spring runoff.

4.2. Sediment

Selenium concentrations in surficial sediment from BGS, UAC, SC, and LiB were 1.0–1.2 μg/g, which were above the value that Presser *et al.* (1994) and Moore *et al.* (1990) used (0.5 μg/g) as a reasonable selenium concentration in sediment to represent the threshold between uncontaminated, background conditions and environments with elevated selenium concentrations. Selenium in surficial sediment from SLC, CC, DC, UGC and LGC were elevated, suggesting a substantial contamination concern.

Elevated selenium in sediments is an important consideration in assessing the health of aquatic ecosystems and has been considered as a federal criterion for selenium in a workshop on selenium aquatic toxicity and bioaccumulation (USEPA, 1998). However, the workshop participants concluded that the sediment compartment was a poor choice for a criterion. Two papers have proposed the use of a sediment-based criterion for selenium expressed on a particulate basis, such as sediment selenium concentration or a measure of the organic content of sediment (Canton and Van Derveer, 1997; Van Derveer and Canton, 1997). Hamilton and Lemly (1999) reviewed these two papers and pointed out how they incorrectly interpreted contaminant survey reports as being exposure-response studies, did not acknowledge the importance of the water-borne entry of selenium in aquatic food webs, overlooked key studies from the extensive body of selenium literature, and failed to consider the off-stream consequences of proposing high in-stream selenium standards.

The sediment component of aquatic ecosystems is an important pathway of inorganic element movement through the food web (Seelye *et al.*, 1982). Specific to selenium, Woock (1984) demonstrated in a cage study with golden shiner (*Notemigonus crysoleucas*) that fish in cages with access to bottom sediments accumulated more selenium than fish held in cages suspended about 1.5 m above the sediments. This study revealed that effects in fish were linked to selenium exposure via sediment, benthic invertebrates, or detritus, or a combination of sediment components. A similar finding was presented by Barnhart (1957) who reported that "numerous species of game fish" lived at least 4 months when held in a livebox, which limited access to food organisms and sediment, but fish lived less than 2 months when released in selenium-contaminated Sweitzer Lake, CO. The highly toxic nature of benthic invertebrates from selenium-contaminated Belews Lake, NC, was reported by Finley (1985) in an experiment where bluegill died in 17–44 days after being fed Hexagenia nymphs containing 13.6 μg/g wet weight selenium. Elevated selenium in sediments at North Pond at Walter Walker State Wildlife Area near Grand Junction, CO (geometric mean 25.1 μg/g in 1996 and 38.9 μg/g in 1997) were associated with elevated selenium in the food chain, and

increased mortality of endangered larval razorback sucker (*Xyrauchen texanus*) in two 30-day water and dietary exposure studies (Hamilton *et al.*, 2001a,b).

MW (1999) evaluated selenium concentrations in sediment at sites in September 1998 that were close to our sites included: 1.3 μg/g in the LiB, 0.5 μg/g in Angus Creek, 0.9 μg/g in the Blackfoot River at the gaging station, 9.4 μg/g in SLC, 1.0 μg/g in SC, 2.6 μg/g in Georgetown Creek, and 0.95 μg/g in DC. In contrast, our SLC site (2.1 μg/g) was a substantial distance downstream of their site (9.4 μg/g) and sediment-bound selenium might not have reached the lower creek area, due to numerous depositional areas above our site. Likewise, our selenium concentrations in sediments at the two Georgetown Creek sites (4.5–7.5 μg/g) were substantially higher that that reported by MW (1999) (2.6 μg/g), which may have been due to their downstream site location. In contrast, the other Georgetown Creek site sampled by MW (1999) was located above the Georgetown Mine and contained only 0.34 μg/g selenium in sediment. Their site on DC (0.95 μg/g) was near the mined area, but also was high elevation and high stream gradient, which may have reduced deposition of selenium in sediments, whereas our site was located closer to depositional areas near the mouth of DC (4.5 μg/g).

MW (2001a) reported selenium concentrations in sediment at sites in September 1999 at LiB (1.6 μg/g), UAC (1.0 μg/g), BGS (1.1 μg/g), SLC (2.1 μg/g), and SC (1.1 μg/g) that were similar to those in the present study. The closeness of selenium concentrations in sediments to those in the present study was because MW had shifted the location of several stations relative to the sites sampled in 1998. The only disparity occurred in sediment selenium concentrations in Georgetown Creek where MW (2001a) reported 6.9 μg/g, which was close to our UGC site (4.5 μg/g), and 1.2 μg/g, which was close to our LGC site (7.5 μg/g). One other possible disparity occurred in Angus Creek where MW (2001a) reported 5.1 μg/g in sediment collected downstream of the sediment pond and about 2 km above our site (1.2 μg/g).

MW (1999) monitored 54 sites in September 1998 and 11 contained selenium concentrations of 2–4 μg/g in sediment including Slug Creek, Dry Valley Creek, Rasmussen Creek (tributary to Angus Creek), and East Mill Creek, whereas SLC contained sediment values greater than 4 μg/g. Overall, the elevated concentrations of selenium in sediments from several streams in the Blackfoot, Salt, and Bear river watersheds in the present study coincides with reports by others (MW, 1999, 2001a; Hamilton *et al.*, 2002; Hamilton and Buhl, 2003a) and suggested widespread contamination of the aquatic environment by phosphate mining.

4.3. Aquatic Plants

No guidelines were found that propose toxicity threshold concentrations for selenium in aquatic plants that might be considered hazardous to aquatic organisms. However, most domestic animals exhibit signs of selenium toxicity on terrestrial vegetative diets containing $\geq$3–5 μg/g natural selenium (NRC, 1980; Eisler, 1985;

Olson, 1986). Selenium concentrations in aquatic plants from LGC and LiB were 1.8 μg/g or less, which was similar to the lower range of concentrations in the previous studies (Hamilton *et al.*, 2002; Hamilton and Buhl, 2003a), and thus, this concentration might be considered near background. By comparison, selenium concentrations in aquatic plants at SC, UAC, UGC, SLC, DC, and CC were relatively elevated, and those at BGS were elevated.

Substantial accumulation of selenium has been reported in aquatic macrophytes by Saiki (1986), Schuler *et al.* (1990), Gutenmann *et al.* (1976), and Barnum and Gilmer (1988) in selenium-contaminated environments. Submerged macrophytes provide a substrate upon which periphyton and some macroinvertebrates colonize, and which benthic invertebrates and some aquatic and semi-aquatic birds and mammals feed. Although fish typically do not feed on macrophytes, when macrophytes die, they become an important contributor to the detrital food chain. Detritus has been reported to contain highly elevated selenium concentrations in selenium-contaminated environments (9.8–440 μg/g; Saiki, 1986; 7–22 μg/g, Saiki *et al.*, 1993; 36–307 μg/g, Saiki and Lowe, 1987), whereas reference areas contained 1 μg/g or less (Saiki and Lowe, 1987). Benthic invertebrates readily accumulate selenium from detritus (Alaimo *et al.*, 1994). Saiki *et al.* (1993) concluded that high concentrations of selenium in aquatic invertebrates and fish in selenium-contaminated areas of central California were the result of food-chain transfer from selenium-enriched detritus rather than other pathways. Thus, aquatic plants with elevated selenium concentrations from several of the stream sites in the present study were probably contributing to the selenium transfer in the aquatic food web and sediments. Although selenium in aquatic plants was not significantly correlated with selenium in sediments in the present study ($r = -0.37$, $p = 0.32$), it was significant in two previous studies ($r = 0.96$, $p = 0.0001$, Hamilton *et al.*, 2002; $r = 0.97$, $p = 0.0001$, Hamilton and Buhl, 2003a).

MW (2001a) measured selenium concentrations in macrophytes collected in September 1999, but only two sites were close those in the present study. They reported 4.2–4.6 μg/g at the BGS, whereas we found 7.2 μg/g. At Angus Creek they reported 1.4–1.7 μg/g, whereas we found 2.8 μg/g. They reported 3.3, 5.1, and 9.2 μg/g in periphyton collected downstream of the sediment pond on Angus Creek about 2 km above our site.

MW (2001b) measured selenium concentrations in macrophytes collected in May 2000, but only two sites were close to those in the present study. They reported 2.2–2.8 μg/g at the BGS, whereas we found 7.2 μg/g. At Angus Creek they reported 0.6–2.2 μg/g, whereas we found 2.8 μg/g. They reported 8.4, 9.6, and 11 μg/g in submergent macrophytes collected downstream of the sediment pond on Angus Creek.

Selenium concentrations in aquatic plants in the present study exceeded the typical background concentration in submerged macrophytes (<1.5 μg/g) (USDOI, 1998). Likewise, selenium concentrations in terrestrial plants collected from selenium-impacted riparian sites such as UAC contained 0.9–1.1 μg/g

(MWH, 2002b), and in grasses (64 μg/g), forbs (78 μg/g), and shrubs (11 μg/g) in riparian areas next to Maybe Creek (TRC Environmental, 1999), a tributary of Dry Valley Creek. These concentrations were above the typical selenium concentrations in terrestrial plants from nonseleniferous soils (<0.25 μg/g) (USDOI, 1998).

MW (2001a,b) reported elevated selenium concentrations in periphyton in the Blackfoot River (3.0–4.3 μg/g), Angus Creek (3.3–9.2 μg/g), Spring Creek (4.2–7.5 μg/g), and very high values in East Mill Creek (12–25 μg/g).

Taking the periphyton and submerged macrophyte data together, the elevated selenium concentrations demonstrated that aquatic plants were accumulating selenium from both water and sedimentary sources in the Blackfoot River watershed. MW (2001a,b) acknowledged that submerged aquatic plants were efficient accumulators of selenium. Their values were similar to data in the present report and previous studies (Hamilton *et al.*, 2002; Hamilton and Buhl, 2003a). Aquatic plants are the foundation of the food web including detritus, and as such, they are the first link in the bioaccumulation of selenium to higher trophic consumers such as aquatic invertebrates and fish.

4.4. Aquatic Invertebrates

Selenium concentrations in aquatic invertebrates from SC and UAC (4.1–5.0 μg/g) were the lowest of the sites investigated, but were above the proposed dietary selenium threshold of 3 μg/g for fish. Several other studies summarized in Hamilton (2002) have reported that dietary selenium concentrations of 4–6 μg/g have caused adverse effects in larval fish. Consequently, the moderate dietary selenium concentrations in LiB, CC, and LGC (5.4–7.8 μg/g), and the elevated concentrations in DC, UGC, SLC, and BGS (8.7–10.8 μg/g) were of concern to the health of fishery resources and species that use these resources.

Although UAC and LiB contained relatively low selenium concentrations in water, surficial sediments, and aquatic plants, selenium concentrations in aquatic invertebrates were elevated. Benthic invertebrates can be efficient accumulators of selenium and can retain elevated concentrations over long periods. For example, Maier *et al.* (1998) reported that aquatic invertebrates contained selenium concentrations of 1.7 μg/g at pretreatment of a watershed with selenium fertilizer, and elevated concentrations during post-treatment monitoring: 4.7 μg/g at 11 days, 4.0 μg/g at 2 months, 5.0 μg/g at 4 months, 4.2 μg/g at 6 months, 4.3 μg/g at 8 months, and 4.5 μg/g at 11 months.

Much of the selenium concentrations in invertebrates likely came from the food web transfer from detritus, which have been reported as the important route of uptake by aquatic invertebrates and fish (Maier and Knight, 1994; Lemly, 1993, 1996b). Three investigations have reported high correlations between selenium concentrations in sediment and benthic invertebrates ($r = 0.94$, Zhang and Moore, 1996; $r = 0.87$, Malloy *et al.*, 1999; Hamilton *et al.*, 2001b), which suggested that selenium concentrations in invertebrates were linked with sedimentary selenium.

Recently, Peters *et al.* (1999) reported that two benthic organisms accumulated selenium directly from spiked sediments. In our study, the linkage between selenium concentrations in invertebrates, sediment, and plants, was supported by the significant correlation between aquatic invertebrates and aquatic plants in the present study ($r = 0.70$) and previous studies ($r = 0.91$, $p = 0.0001$, Hamilton *et al.*, 2002; $r = 0.74$, $p = 0.04$, Hamilton and Buhl, 2003a).

MW (2001a) measured selenium concentrations in aquatic invertebrates collected in September and October 1999, but only two sites were close to those in the present study. They reported 5.0 μg/g at the BGS, whereas we found 10.8 μg/g. At Angus Creek they reported 6.2 μg/g, whereas we found 5.0 μg/g. In contrast, they reported 12 μg/g in invertebrates collected below the sediment pond on Angus Creek and about 2 km upstream of our site.

MW (2001b) measured selenium concentrations in aquatic invertebrates collected in May and June 2000, but only two sites were close those in the present study. They reported 4, 12, and 20 μg/g at the BGS, whereas we found 10.8 μg/g. At Angus Creek they reported 1, 10, and 37 μg/g, whereas we found 5.0 μg/g. In contrast, they reported 12, 20, and 22 μg/g in invertebrates collected below the sediment pond on Angus Creek.

Selenium concentrations in aquatic invertebrates in the present study exceeded the typical background concentration in aquatic invertebrates (<2 μg/g) (USDOI, 1998). Likewise, selenium concentrations in terrestrial invertebrates collected from selenium-impacted riparian sites such as UAC contained 1.6, 2, and 6 μg/g (MWH, 2002b), which was above the typical selenium concentrations in terrestrial invertebrates (<1.5 μg/g) (USDOI, 1998).

Elevated selenium concentrations in aquatic invertebrates were reported in two previous studies in the Blackfoot River watershed (Hamilton *et al.*, 2002; Hamilton and Buhl, 2003a), and by others. Elevated selenium concentrations have been reported in benthic invertebrates collected from ponds (110–390 μg/g) and a lotic area (14 μg/g) of Maybe Creek, a tributary of Dry Valley Creek (TRC Environmental, 1999). Benthic invertebrate samples collected from various sites in Blackfoot Reservoir contained ≤ 2 μg/g in September 1999, except for three samples, which contained selenium concentrations of 3.8, 4.6, and 10 μg/g (MW, 2001a). However, in the May 2000 sampling, 8 of 12 samples from Blackfoot Reservoir contained a geometric mean selenium concentration of 7.8 μg/g (range 5.3–12 μg/g; MW, 2001b).

Benthic invertebrates collected in September 1999 from numerous stream sites in the Blackfoot River watershed contained low selenium concentrations in 5 of 26 samples (3.0–4.6 μg/g), moderately elevated concentrations in five samples (5.0–15 μg/g), and highly elevated concentrations at East Mill Creek (72 μg/g) (MW, 2001a). In the May 2000 sampling, low selenium concentrations occurred in 11 of 42 samples (3.0–4.9 μg/g), 17 samples contained moderately elevated concentrations (5.0–37 μg/g), and East Mill Creek contained 100, 120, and 170 μg/g (MW, 2001b).

The large number of samples with substantial selenium concentrations in aquatic invertebrates from the Blackfoot, Salt, and Bear river watersheds above the proposed toxic threshold of 3 μg/g for fish suggested that benthic invertebrate populations were highly contaminated with selenium. Similar to aquatic plants, benthic invertebrates also demonstrated that selenium accumulation was occurring. Bioaccumulation of selenium through the food web from invertebrates to higher trophic organisms such as fish have been reported by several investigators (Sandholm *et al.*, 1973; Finley, 1985; Bennett *et al.*, 1986; Dobbs *et al.*, 1996; Hamilton *et al.*, 2001a,b).

4.5. Fish

Selenium concentrations in fish from the nine sites, based on geometric mean values, followed the same pattern of accumulation as in aquatic plants and aquatic invertebrates, but not surficial sediments. The similarity in selenium accumulation between aquatic ecosystem components also paralleled the significant correlations between selenium concentrations in fish, aquatic plants, and aquatic invertebrates, which demonstrated the interconnectedness of the aquatic ecosystem components. This accumulation pattern was supported in reviews of the selenium literature (Maier and Knight, 1994; Lemly, 1993, 1996b).

SLC contained the highest selenium concentrations in whole-body fish and also the highest inorganic element concentrations based on the Friedman test (Hamilton and Buhl, 2003b). There seemed to be no parallel accumulation between selenium concentrations and concentrations of inorganic elements in fish from the other streams, especially for DC, UGC, LiB, and UAC. Consequently, SLC seemed to standout as a potentially highly impacted stream. In contrast, elevated selenium concentrations in fish from BGS, DC, CC, and UGC were cause for concern. There were no significant differences among the streams based on inorganic element concentrations including selenium in the combined dataset of sediment, aquatic plant, aquatic invertebrate, and fish (but not water) using the Friedman test (Hamilton and Buhl, 2003b).

This scenario of selenium being a more important contaminant than other inorganic elements in the present study has occurred in other contaminant investigations. For example, Furr *et al.* (1979) examined contaminated food chains in coal ash settling basins and concluded that only selenium was of concern to biota. Other investigations reaching similar conclusions were reported by Sorensen (1988), Lemly (1985), Saiki and Lowe (1987), Nakamoto and Hassler (1992), Gillespie and Baumann (1986), Bryson *et al.* (1984), MW (1999), and Hamilton *et al.* (2001a,b).

A workshop on selenium aquatic toxicity and bioaccumulation concluded that the tissue-based criterion might be the best approach for a national criterion because tissue residues accounted for selenium's biogeochemical pathways by integrating the route, duration, and magnitude of exposure, chemical form, metabolic transformations, and modifying biotic and abiotic factors (USEPA, 1998). A recent paper gave the rationale for a tissue-based criterion for selenium in fish, and proposed a

national criterion of 4 μg/g in whole body based on the review of several laboratory and field studies (Hamilton, 2002). This concentration was the same as the whole-body toxicity threshold for fish proposed earlier by Lemly (1993, 1996b) and similar to the threshold of 4.5 μg/g proposed by Maier and Knight (1994). Other papers have proposed selenium toxicity thresholds of 6 μg/g for coldwater anadromous fish and 9 μg/g for warm water fish (DeForest *et al.*, 1999; Brix *et al.*, 2000). The approach, information, and conclusions presented in DeForest *et al.* (1999) and Brix *et al.* (2000) have been reviewed and problems in their interpretation and conclusions have been discussed in Hamilton (2003).

Based on a whole-body toxicity threshold of 4–4.5 μg/g, the geometric mean selenium concentrations in fish from SC, UAC and LGC (range 4.2–6.8 μg/g) would probably have some effects on early life stages of sensitive species. Fish in LiB and UGC (7.6–9.8 μg/g) contained selenium concentrations above the 4–4.5 μg/g threshold value, thus suggesting possible effects in sensitive fish species in these streams. Elevated whole-body residues of selenium in fish from CC, DC, BGS, and SLC (10.4–15.2 μg/g) suggested sensitive and moderately sensitive fish are probably being adversely affected by selenium exposure.

MW (2001a) measured selenium concentrations in fish collected in September 1999, but only two sites were close to those in the present study. They reported 10.0 μg/g (converted from 2.8 μg/g wet weight) in unidentified forage fish at the BGS, whereas we found 12.0 μg/g. At Angus Creek they reported 3.4 μg/g (converted from 0.74 μg/g wet weight) in unidentified forage fish, whereas we found 6.4 μg/g. In contrast, they reported 7.1 μg/g (converted from 1.7 μg/g wet weight) collected below the sediment pond on Angus Creek. For other sites, nine of 13 forage fish samples contained elevated selenium concentrations in fish (5.2–8.3 μg/g, after conversion to dry weight using the percent moisture given for each sample), and two samples contained high selenium concentrations of 10 and 12.9 μg/g (MW, 2001a).

MW (2001b) measured selenium concentrations in fish collected in May 2000, but only two sites were close to those in the present study. They reported 5.1, 6.2, and 10.0 μg/g (assuming dry weight) in unidentified forage fish (they collected two redside shiner and one sucker species, but did not match the species with the residue) at the BGS, whereas we found 12.0 μg/g. At Angus Creek they reported 4.2, 7.4, and 37 μg/g (assuming dry weight) in cutthroat trout, whereas we found 6.4 μg/g. For other sites, 13 of 36 samples of forage fish collected in the Blackfoot River watershed in May 2000 contained selenium concentrations of 5.0–9.4 μg/g, and 13 samples contained concentrations of 10–37 μg/g (MW, 2001b).

Elevated selenium concentrations in fish were reported in two previous studies in the Blackfoot River watershed (Hamilton *et al.*, 2002; Hamilton and Buhl, 2003a), and by others. Rich and Associates (1999) reported concentrations of inorganic elements in cutthroat trout, rainbow trout, 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. They

concluded that selenium and other elements (cadmium, copper, lead, vanadium, and zinc) were probably causing stress in fish populations in Dry Valley Creek.

Selenium concentrations in whole-body salmonids collected in September 1999 from Blackfoot Reservoir and the mainstem and tributaries of the Blackfoot River were elevated in 21 of 50 samples (4.2–9.7 μg/g) and high in seven samples (12–31 μg/g) (converted to dry weight using the appropriate percent moisture from MW (2001a), and whole body using a factor of 1.667, Lemly and Smith, 1987). For salmonids collected in May 2000 from various locations in the Blackfoot River, selenium concentrations in whole body were elevated in 13 of 27 samples (5.2–9.2 μg/g) and high in 12 samples (10–48 μg/g) (converted to dry weight using the appropriate percent moisture from MW (2001b), and whole body using a factor of 1.667, Lemly and Smith, 1987). These selenium residues in forage fish and salmonids were substantially above background concentrations in fish from laboratory and field investigations, which are typically 1–2 μg/g (Maier and Knight, 1994; Hamilton *et al.*, 2000). More importantly, the selenium residues were above those reported to cause adverse effects in early life stages of fish, including salmonids (4–5 μg/g; Hamilton *et al.*, 2000). In particular, selenium residues of 5.2 μg/g in rainbow trout were associated with reduced survival (Hunn *et al.*, 1987), and 3.8–4.9 μg/g in chinook salmon (*Oncorhynchus tshawytscha*) were associated with reduced survival and growth (Hamilton *et al.*, 1986; Hamilton and Wiedmeyer, 1990).

Based on the above discussion, selenium contamination of the Blackfoot River and its tributaries is most likely adversely affecting aquatic resources, especially early life stages of fish. The large number of samples in the present study with substantial selenium concentrations above the proposed toxic whole-body threshold of 4 μg/g suggested that fish populations have accumulated elevated selenium concentrations similar to aquatic plants and benthic invertebrates. Thus, forage fish and salmonids probably pose a hazard from dietary selenium toxicity to predatory fish and fish-eating wildlife.

One concern may be the presence of elevated selenium residues in fish from the Blackfoot River and tributaries of the Salt and Bear rivers without readily apparent biological effects. However, data in the current study and studies by others (Rich and Associates, 1999; MW, 1999, 2000, 2001a,b) were from contaminant surveys and not biological effects studies. Secondly, residues measured in fish were for adults or subadults. This life stage is generally less sensitive to the effects of environmental contaminants than are early life stages (Rand and Petrocelli, 1985). The third consideration was the movement of fish in the Blackfoot, Salt, and Bear river watersheds or in any open river system. Adverse effects on a demographically open fish population in a section of the river with contaminant impacts would be difficult to detect and must be confirmed with detailed biological studies because of immigration of individuals from the portion of the population in non-affected river reaches or tributary streams. The review by Skorupa (1998) addresses this concern succinctly and stated, "It is common for instream studies to report the counterintuitive combination of abnormally elevated levels of selenium in fish tissue

associated with what is viewed as a normally abundant and diverse fish fauna." 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 (Stephan *et al.*, 1985).

MW (2001b) acknowledged that higher than expected selenium 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 selenium concentrations of 10, 12, and 22 μg/g. However, in spite of high selenium residues in whole-body forage fish collected in May 2000, MW (2001b) stated that, "There is no evidence of forage fish in the Blackfoot Reservoir being impacted by either selenium or cadmium." Likewise, MW (2001a) reported elevated selenium concentrations in forage fish collected in September 1999, yet stated that, "Evaluation of forage fish data show no evidence that this medium is impacted in the reservoir." Because no biological effects were assessed in fish collections in September 1999 or May 2000, their statements were unsupported.

4.6. Hazard assessment

Lemly (1995) presented a protocol for aquatic hazard assessment of selenium, which was 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: 1, no identifiable hazard (no toxic threat is identified and selenium concentrations are not elevated in any ecosystem component); 2, minimal hazard (no toxic threat identified but concentrations of selenium are slightly elevated in one or more ecosystem components [water, sediment, benthic invertebrates, fish eggs, bird eggs] compared to uncontaminated reference sites); 3, low hazard (a periodic or ephemeral toxic threat that could marginally affect the reproductive success of some sensitive species, but most species will be unaffected); 4, moderate hazard (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); 5, high hazard (an imminent, persistent toxic threat sufficient to cause complete reproductive failure in most species of fish and aquatic birds). 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, and 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. Table V gives the

TABLE V

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

	Hazard														
	None			Minimal			Low			Moderate			High		
Ecosystem component	Concentration	Lemly[a] score	Modified score	Concentration	Lemly[a] score	Modified score	Concentration	Lemly[a] score	Modified score	Concentration	Lemly[a] score	Modified score	Concentration	Lemly[a] score	Modified score
Water (μg/L)	<1	1	1	1–2	2	2	2–3	3	3	3–5	4	4	>5	5	5
Sediment (μg/g)	<1	1	1	1–2	2	2	2–3	3	3	3–4	4	4	>4	5	5
Benthic invertebrate (μg/g)	<2	1	2	2–3	2	4	3–4	3	6	4–5	4	8	>5	5	10
Fish eggs (μg/g)	<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[a])		4			5–7			8–10			11–14			15–20	
Final hazard (Modified)			7			8–13			14–20			21–27			28–35

[a]Lemly (1996a).

hazard term and corresponding selenium concentration range for each of the four ecosystem components in the four-component model (Lemly, 1996a).

These protocols have been used to assess the selenium hazard to aquatic ecosystems at Ouray NWR, UT (Lemly, 1995, 1996a), the Animas, LaPlata, and Mancos rivers in the San Juan River basin (Lemly, 1997), three Wildlife Management Areas in Nevada (Lemly, 1996a), and three sites near Grand Junction, CO (Hamilton *et al.*, 2001a,b).

The selenium hazard protocols give equal weight to each component (Lemly, 1995, 1996a). However, there may be 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 2 for the score for benthic invertebrate information and a factor of 3 for the score for fish eggs and bird eggs. Similar concerns have been raised by a USGS scientist (M. Sylvester, Menlo Park, CA, 2002, written communication), and a USFWS Environmental Contaminant Specialist (B. Osmundson, Grand Junction, CO, 2001, written communication). The weighting of the three biological components seems justified based on the repeated expression of their importance in the selenium 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 results in modified final hazard 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 V). 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 V). However, if in an environmental situation all measured selenium 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) set 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 seems biologically reasonable and is referred to here as the offset summation approach. Similar offsets for other final hazards are given in Table V. 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. This modified hazard assessment was used in previous investigations in the Blackfoot River watershed (Hamilton *et al.*, 2002; Hamilton and Buhl, 2003a,b).

In the present study, fish eggs were not collected. In the hazard assessment, we converted the geometric mean whole-body concentrations of selenium in fish to fish egg concentrations using the conversion factor based on Lemly (1995, 1996a), who reported: whole-body $\times$ 3.3 = fish egg. The hazard assessment for the nine sites is given in Table VI.

TABLE VI
Hazard assessment of selenium at nine sites in southeastern Idaho using modified scores

Site[a] and ecosystem component	Selenium concentration[b]	Evaluation by component		Total for the site	
		Hazard	Score	Score	Hazard
LiB					
Water	<2	None	1		
Sediment	1.8	Minimal	2	28	High
Benthic invertebrate	5.4	High	10		
Fish eggs[c]	25.1	High	15		
UAC					
Water	<2	None	1		
Sediment	1.2	Minimal	2	26	Moderate
Benthic invertebrate	5.0	Moderate	8		
Fish eggs	21.1	High	15		
BGS					
Water	<2	None	1		
Sediment	1.0	Minimal	2	28	High
Benthic invertebrate	10.8	High	10		
Fish eggs	39.6	High	15		
SLC					
Water	<2	None	1		
Sediment	2.1	Low	3	29	High
Benthic invertebrate	9.7	High	10		
Fish eggs	50.2	High	15		
SC					
Water	<2	None	1		
Sediment	1.2	Minimal	2	23	Moderate
Benthic invertebrate	4.1	Moderate	8		
Fish eggs	13.9	Moderate	12		
UGC					
Water	11	High	5		
Sediment	4.5	High	5	35	High
Benthic invertebrate	9.3	High	10		
Fish eggs	32.3	High	15		
LGC					
Water	<2	None	1		
Sediment	7.5	High	5	31	High
Benthic invertebrate	7.8	High	10		
Fish eggs	22.4	High	15		
DC					
Water	<2	None	1		
Sediment	4.5	High	5	31	High
Benthic invertebrate	8.7	High	10		
Fish eggs	38.0	High	15		
CC					
Water	<2	None	1		
Sediment	2.1	Low	2	28	High
Benthic invertebrate	6.7	High	10		
Fish eggs	34.3	High	15		

[a]LiB: Little Blackfoot River, UAC: upper Angus Creek, BGS: Blackfoot River gaging station, SLC: State Land Creek, SC: Smoky Creek, UGC: upper Georgetown Creek, LGC: lower Georgetown Creek, DC: Deer Creek, CC: Crow Creek.

[b]Selenium concentrations in μg/L for water, μg/g for sediment, benthic invertebrates, and fish eggs.

[c]Fish eggs: fish egg values converted from whole-body residues using: whole-body $\times$ 3.3 = fish egg (Lemly, 1995, 1996a).

The two sites with minimal to moderate selenium concentrations in most aquatic ecosystem components had moderate overall hazard rating: UAC and SC. Although selenium concentrations were none or minimal in water and sediment at LiB and BGS, they were elevated in benthic invertebrates and whole-body residues converted to fish egg concentrations, resulting in an overall high hazard rating. Selenium concentrations in water or sediment were in the none or low categories at SLC and CC, but high in benthic invertebrates and whole-body residues. Thus these two sites received high final hazards. UGC, LGC and DC consistently contained elevated selenium concentrations in sediment, invertebrates, and whole-body residues, thus resulting in an overall high hazard rating. Using the original Lemly (1996a) approach, LiB, BGS, SLC, and CC would have received moderate final hazards in spite of the high score for benthic invertebrates and fish eggs (converted from whole-body residues).

Based on the final hazard score (in parentheses) the streams can be listed from highest selenium hazard to lowest as follows: UGC (35), DC (31), LGC (31), SLC (29), BGS (28), LiB (28), CC (28), UAC (26), SC (23). This ranking is very similar to the results of the Friedman test using the ranked selenium concentrations: UGC_a, SLC_a, BGS_a, DC_{ab}, CC_{ab}, LGC_{abc}, LiB_{bc}, UAC_c, SC_c. The only major disparity seemed to occur in the ranking of LGC. Thus, the selenium hazard protocol seems to be a useful tool in assessing the differences among sites due to the comparable outcome of statistical approach such as the Friedman test.

Reports by MW (1999, 2000, 2001a,b) do not present hazard assessments. However, the data evaluations of the various aquatic ecosystem components for water, sediment, submerged macrophytes, benthic invertebrates, forage fish, and salmonid fillets, tend to suggest no major impacts from selenium and other elements, with the exception of creeks influenced directly by phosphate mining.

A preliminary assessment of selenium hazard in the Caribou National Forest was conducted using selenium 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 occurring in fish and wildlife associated with the Blackfoot River, its tributaries, and tributaries of the Salt and Bear rivers. The results of the present study and two previous studies add substantially more support to the premise that selenium concentrations in several aquatic ecosystem components were sufficiently elevated to cause adverse effects to aquatic resources and terrestrial species that utilize these resources in the Blackfoot River watershed.

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