NOAA PRELIMINARY NATURAL RESOURCE SURVEY

Blackbird Mine

Lemhi County, Idaho

Cerclis # IDD000643122
Site ID: P1

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C.A. Mebane
Coastal Resource Coordination Branch
Hazardous Materials Response and Assessment Division
NOAA Office of Resource Conservation and Assessment
7600 Sand Point Way NE
Seattle, WA 98115

Prepared for:

D. Bennett
Superfund Response and Investigations Branch
Hazardous Waste Division
EPA Region 10
Seattle, WA

Abstract

The Salmon River in Idaho is the largest un-dammed tributary to the Columbia River drainage; it contains the most stream habitat for anadromous salmonids of all the tributary watersheds that make up the Columbia River drainage. Panther Creek, a major tributary to the Salmon, receives metals contaminated mine drainage from the Blackbird Mine. This survey was conducted to: (1) make a reconnaissance of the distribution of metals contamination in Panther Creek sediments, (2) determine whether sediments collected from Panther Creek were toxic to aquatic invertebrates, and (3) assemble other data relating to the exposure potential and effects of contaminants of concern to natural resources for which NOAA has trustee responsibilities (anadromous salmon and trout). Data on mine effluents from headwaters areas, contamination in downstream surface water and sediments and sediment-water partitioning are compiled. These are compared to accompanying benthic communities and fish populations in Panther Creek.

Arsenic, cobalt and copper concentrations in sediments downstream of the mine effluents were elevated up to 300 times the upstream concentrations while only cobalt and copper were elevated in surface water. Cobalt and copper but not arsenic were readily released from sediments in leaching tests. Toxicity of metals in Panther Creek sediments was estimated through laboratory growth and survival testing with the amphipod *Hyalella azteca*. Cobalt and copper but not arsenic concentrations in sediment were strongly correlated with *Hyalella* toxicity. Normalizing metals concentrations to the amount of organic carbon in the sediment significantly increased the strength of the correlations, suggesting that the organic carbon content may moderate the bioavailability of copper and cobalt in sediments. Surface water and sediment contamination patterns corresponded to altered benthic community structure and reduced fish populations. The effects of copper and cobalt to invertebrates and salmonids from controlled experiments reported in the literature and those observed in Panther Creek are similar. Additionally, studies from other areas with similar metals contamination strongly suggest that (1) copper contamination impedes homing and downstream migratory behavior in anadromous salmonids, and (2) metals in the food chain are an important factor in reducing the survival and growth of juvenile salmonids.

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I. Site Description and History

The Blackbird Mine, on a high divide in the Salmon River Mountains at the edge of the River of No Return Wilderness in east-central Idaho, is about 36 km southwest of the town of Salmon. Two streams, Big Deer Creek to the North, and Blackbird Creek to the south, drain the mining area. Both streams flow east into Panther Creek which then flows north to the mainstem of the Salmon River (Figure 1). Panther Creek is one of the seven major tributaries of the Salmon River, which, in turn, is the largest sub-basin of the Columbia River, excluding the Snake River, and has the most stream habitat available for anadromous salmonids. Panther Creek, 71 km long with a drainage area of about 1380 km^2, makes up about 10 percent of the overall Salmon River watershed by area. Panther Creek contributes between about 8 percent (February) and 19 percent (May) of the mean Salmon River flows[1].

The Blackbird mine is one of the largest cobalt deposits in North America. The primary sulfide ores are a cobalt-arsenic sulfide called cobaltite (CoAsS), chalcopyrite ($CuFeS_2$), pyrite (FeS_2), and pyrrhotite (FeS). Mining gold and copper has continued intermittently since the 1890s. The mine has been inactive since 1982. It has about 24 km of underground workings (12 levels with 8 portals), a five-ha open pit, and about 34 ha of exposed metals- contaminated mine waste (Bennett 1977; Reiser 1986).

Panther Creek formerly supported large anadromous chinook salmon and steelhead trout runs. The loss of anadromous fish runs in the Panther Creek drainage has been linked closely with mining activity and mine practices at the site. Some of the earliest reports (ca. 1930) on mining in the area suggest that all mine tailings were channeled directly into Blackbird Creek (Reiser 1986). Settling ponds and tailings pipelines were subsequently constructed in the 1940s and 1950s. These containment methods frequently failed with periodic spills of tailings going unchecked into Blackbird and, ultimately, Panther Creek[2]. Open-pit mining began in about 1954 in the Big Deer drainage, resulting in contaminated mine drainage entering Panther Creek via Big Deer and Bucktail creeks (Reiser 1986).

Welsh et al. (1965) described the mining activity as follows:

> "Above Blackbird Creek, no stream pollution exists. From 1948 to 1961, a mine located at the head of Blackbird Creek employed a floatation process for ore separation. A number of reagents were used, including sodium sulfide, sulfuric acid, and pine oil. The effluent was then run into a settling pond where the sediment dropped out. During the winter the pond would freeze and sediment escaped into Blackbird Creek and Panther Creek. Unless lime was added at the settling pond, the water in Blackbird Creek was highly acidic. After entering Panther Creek, the acidity was dissipated by neutral water and a heavy reddish precipitate which persisted for about 10 miles formed on the bottom of the stream."

In December 1983, the Idaho Attorney General filed suit for recovery of damages for injures to natural resources caused by releases of hazardous substances from the Blackbird Mine site against Noranda Mining Inc., Howmet Turbine Component Corporation, and the Hanna Mining Company, alleging pollution damages to state surface and ground waters. The suit and related claims are still unresolved. In May 1993, EPA proposed the Blackbird Mine for inclusion on the Superfund National Priorities List (NPL).

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1 Gauged near the mouth of Panther Creek from 1945-1977 (CBFWA 1990)

2 Nelson et al. (1991) p. 443 includes a photo of this practice; Blackbird Creek flowed directly through tailings piles.

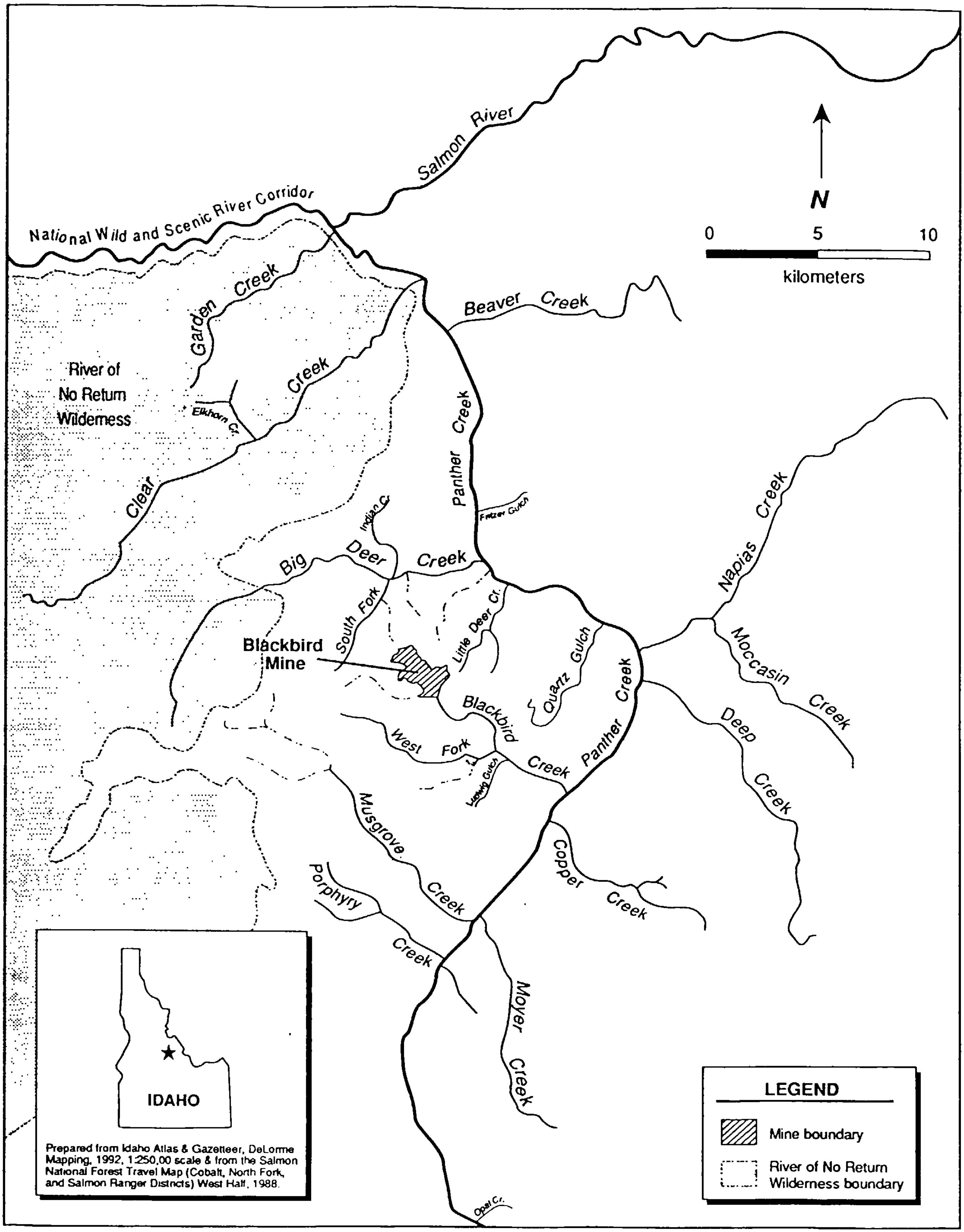

Figure 1. Blackbird Mine study area in the Salmon National Forest, Lemhi County, Idaho.

II. Pathway Characterization

The Blackbird and Big Deer Creek drainages are the pathways for contaminant migration from source areas at the mine to Panther Creek, the habitat of concern for NOAA trust resources (anadromous fish). The Blackbird mine study area is located in an area of high mountain ridges, and steep canyons varying in elevation from 2,370 m at the divide between the two drainages to 990 m at the confluence of Panther Creek with the Salmon River (Figures 1 and 2). A tributary to Blackbird Creek, Meadow Creek flows south through the mine area before joining Blackbird Creek, then flowing about 10 km downstream from the mine to join Panther Creek about 40 km above its confluence with the Salmon River. Bucktail Creek has its headwaters in mine waste just below the open pit; it then joins the South Fork of Big Deer, and then Big Deer Creek, before joining Panther Creek about 19 km above its confluence with the Salmon River.

Besides the Blackbird Mine, there are few likely sources of chemical contamination to the Panther Creek watershed. The patented (privately owned) mine lands are surrounded by the Salmon National Forest. The River of No Return Wilderness borders the Blackbird mining district to the west, including upper Big Deer Creek. There are several comparatively small mining sites on other Panther Creek tributaries, including the Blackpine Mine on Copper Creek and the Copper King Mine on Beaver Creek. These other mines do not appear to measurably contribute to Panther Creek metals contamination. This is discussed further in Section IV.

The Acid Mine Drainage Problem

Mining activities release metals both by breaking up previously impermeable rocks and exposing them to water, and by exposing sulfide-containing rocks to oxygen, resulting in rapid alteration and dissolution. Acid mine drainage (AMD) often results when sulfide minerals are either in the ore or in the surrounding waste rock. Aided by bacterial decomposition, acid is produced when metal sulfide ores react with oxygen-rich water, forming metal ions, sulfate, and hydrogen ions. When these sulfide minerals, particularly pyrite and pyrrhotite, are exposed to oxygen and water, they begin to oxidize almost immediately. In the absence of calcareous materials, the initial chemical reactions produce acid and liberate heavy metals associated with the waste deposit. As the reactions proceed, temperature and acidity increase, resulting in an increased reaction rate. Between pH levels of 2 and 4, bacteria and ferric iron catalyze the reactions, and rates can be five to six orders of magnitude faster than the original inorganic rate. Once established, these reactions may continue undiminished without oxygen and throughout the low temperatures of winter. Spring snowmelt then flushes the metal salts and toxic solution from the interstices of the waste rock into the downstream waters (Nordstrom 1982; Smith et al. 1992).

Surface Runoff Pathway

Infiltration and runoff provide the surface water pathway for transporting contaminants from surface deposits at the site to the downstream aquatic habitats of concern result from. Precipitation, mostly snowfall, liberates soluble metals from contaminated surface deposits and transports them downstream through Meadow and Bucktail creeks (Figure 2). During the stable low-flow conditions of late summer to late winter, acid mine drainage from mine water, and contaminated springs and seeps at the bases of waste piles create surface runoff pathways. During spring runoff, the amount of contaminants and the number of sources greatly increase through the surface runoff pathway. Waste rock is eroded as: sheet runoff removes thin layer containing metal evaporation salts from waste piles, tailings piles, and road surfaces (constructed from waste rock and mill tailings); gully erosion of waste piles

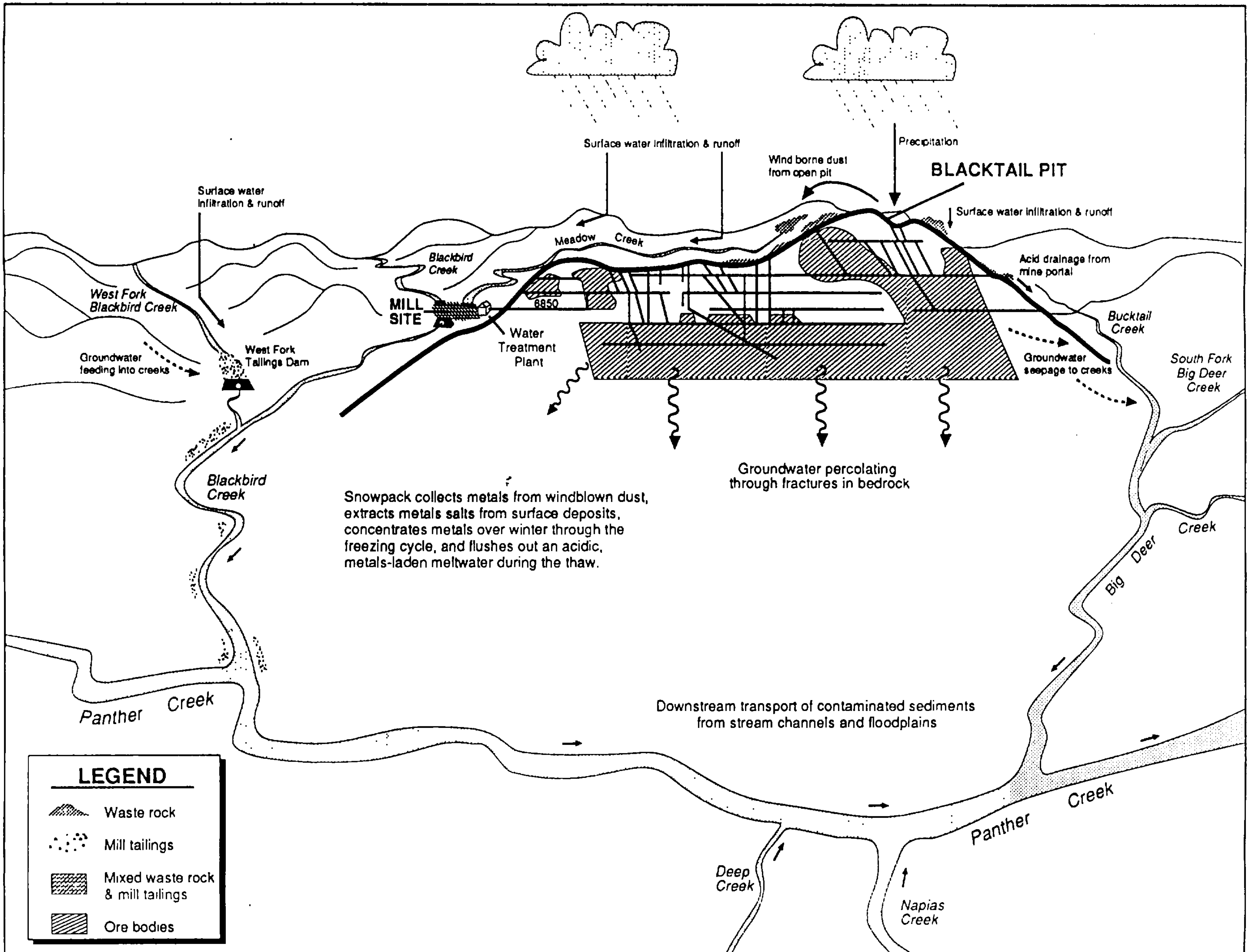

Figure 2. Drainage features and types of contamination associated with the Blackbird mine, Lemhi County, Idaho.

occurs; channel scour resuspends and increases contaminated stream sediment load; and releases metals from snow containing contaminated dust layers (Reiser 1986). The Blackbird Creek floodplain contains sulfidic tailings (Beltman et al. 1993). In semi-arid areas of North America, metal sulfates precipitate in surface sediments as porewaters carrying by-products of sulfide oxidation evaporate. The crusts will form on stream-side tailings on the floodplain. The salts dissolve readily and release metals in water-soluble forms (Nimick and Moore 1991).

Relationship of snowpack to acid mine drainage

Studies of snowpack chemical contamination, sources of snowpack contamination, snow meltwater contamination, and streamflow contamination at the site have shown that the snowpack was a major pollution sink and pathway for acid mine drainage. The steep, open-pit walls and other windblown areas of the mine stay snow-free much of the winter. Highly contaminated blowing "snowdust" from these areas settled and accumulated on the surface of the snowpack. It contributed water-soluble oxidation products directly to runoff and provided reactive material for further oxidation. Snow meltwater from the site was triply contaminated: (1) contaminants from the snowdust probably concentrate on the surface of ice crystals; (2) contaminants are wicked by capillary action into the snowpack base from the ground surface; and (3) sheet flow surface water picks up soluble oxidation products on its way to stream courses. These soluble oxidation products were quickly flushed from the snowpack into streamflow at the first thaw, with a spike of greatly increased metals levels occurring several weeks before peak spring runoff. Sampling later during the spring runoff showed reduced metal concentrations due to dilution (Farmer and Richardson 1980).

Baldwin et al. (1978) reported a similar trend following extensive sampling over several seasons. They reported that streamflow and metals concentrations in the surface water pathways in the Blackbird mining area were closely interrelated. Metal concentrations were: (1) low during winter months, (2) increase sharply during the initial spring runoff period, (3) were very low during the latter part of the spring runoff, and (4) rise gradually during later summer months. About 75% of the total annual metal productions occurred during April and May.

Groundwater Pathway

Groundwater at the Blackbird area occurs in both fracture-controlled bedrock systems and in alluvial (unconsolidated surficial) deposits. Near-surface bedrock groundwater is expressed as seeps, springs, and mine portal discharges. Two major alluvial aquifer systems have been identified at the site: (1) natural alluvial material containing groundwater, which is in direct hydraulic contact with streams at the site, and (2) groundwater flow through the large deposits of mine wastes in the Blackbird and Bucktail drainages (Baldwin et al. 1978). Figure 3 shows potential groundwater flow patterns in the headwaters of Bucktail Creek relating to the mine workings. Since the Baldwin study, Noranda has reportedly routed most of the mine water that flows through the underground workings to the 6850 foot level where they collect and treat the water although no as-built descriptions of the current mine conditions were available (Reiser 1986).

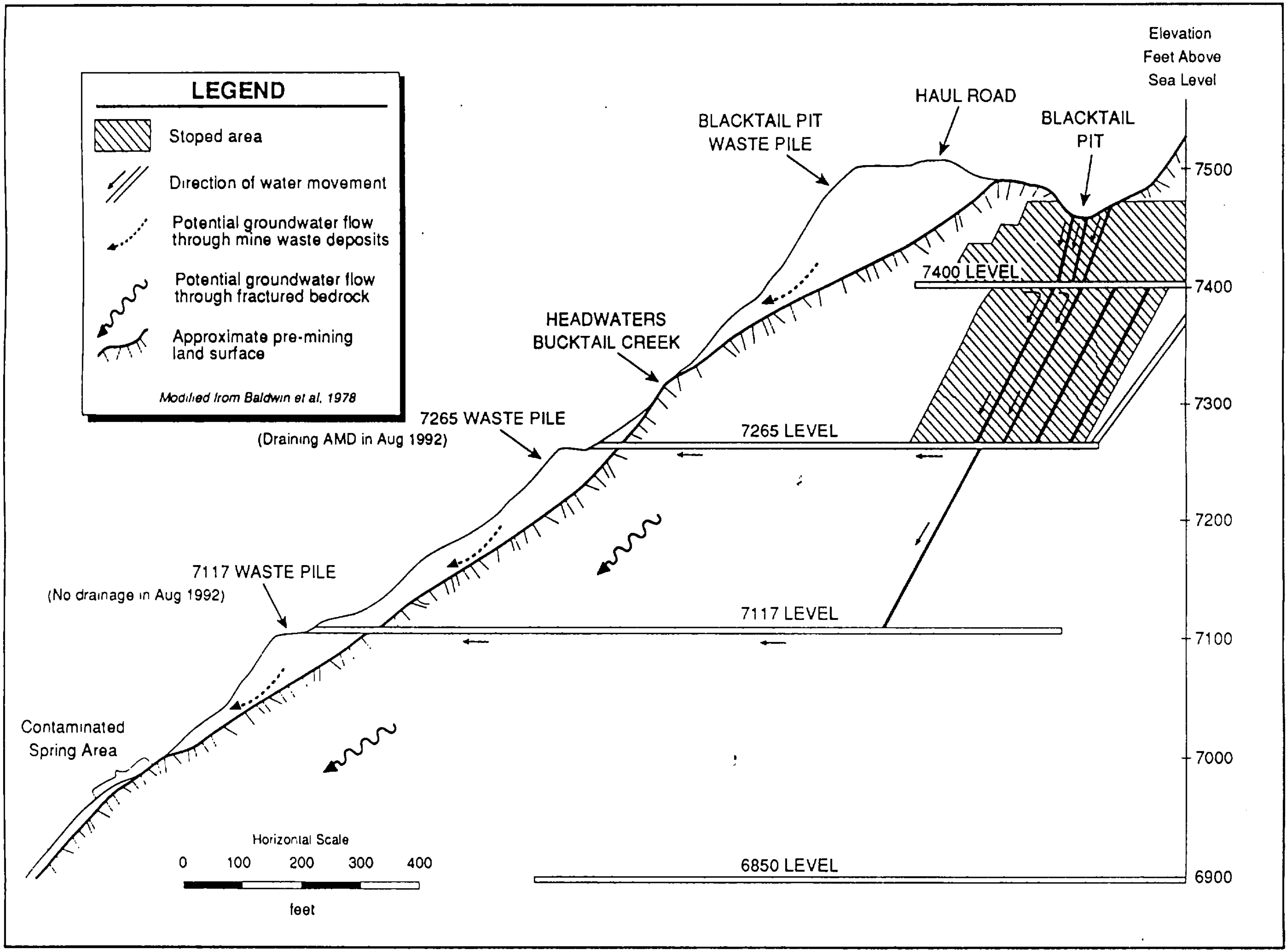

Figure 3. Blacktail Pit and the headwaters of Bucktail Creek. During August 1992 (a drought year), the 7117 portal was dry and the 7265 portal was draining mine water.

III. Potentially Exposed Resources

Habitat Characterization

The habitats for anadromous fish at risk from contamination from the Blackbird Mine include the entire length of Panther Creek and, potentially, the mainstem of the Salmon River; the downstream limit of the chemical contamination signal from Blackbird has not been delineated beyond its confluence with the Salmon River (discussed in Section IV). Panther Creek and its tributaries, as part of the Middle Salmon-Panther hydrologic unit, are designated as critical habitat for the threatened Snake River spring/summer chinook salmon (*Oncorhynchus tshawytscha*[3]; NMFS 1993). Also, all surviving individuals of the endangered Snake River sockeye salmon (*Oncorhynchus nerka*) must pass by Panther Creek on their migratory pathway up the Salmon River to their last remaining spawning grounds in Redfish Lake in the Salmon River Basin upstream from the Panther Creek drainage (NMFS 1991b, 1993). The loss of habitat in Panther Creek resulting from water quality degradation from the Blackbird Mine was specifically cited as a contributing factor leading to the decline of the Snake River spring/summer chinook salmon species (NMFS 1991a).

Essential features of critical habitat for both listed sockeye and chinook salmon species include adequate substrate, water quality, water quantity, water temperature, water velocity, cover/shelter, food, riparian vegetation space, and migration conditions. The water, bottom, and adjacent riparian zone are included as critical habitat including river reaches that are not presently inhabitated by them. Because of the contribution of essential habitat elements such as food, water quality, gravel, and large woody debris, critical habitats were designated on a hydrologic unit rather than an individual reach basis (NMFS 1993).

Habitats for the anadromous chinook salmon and steelhead trout occur in Panther Creek from its mouth for about 65 km upstream to the upstream limit of anadromous salmonid habitats near Opal Creek, and in portions of seven tributary streams. Reiser (1986) made detailed qualitative and quantitative surveys of salmon and steelhead spawning and rearing habitats in the Panther Creek drainage to determine the limiting habitat component for anadromous fish (i.e., rearing or spawning). He stratified Panther Creek into five discrete salmonid habitat types (Table 1).

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[3] The reports referenced here generally refer to three forms of chinook salmon that enter the Snake River (spring-, summer-, or fall-run). Based on evaluating reproductive isolation and genetic/ecological diversity, the NMFS determined that under the Endangered Species Act, Snake River chinook salmon consist of two species, the spring/summer and the fall "species." In the Salmon River sub-basin including Panther Creek, the spring/summer form is the only species occurring. All references to "chinook" or "salmon" in this report refer to Snake River spring/summer chinook salmon.

Table 1: Panther Creek chinook salmon and steelhead trout habitat types(Reiser 1986).

Habitat Description	Area of juvenile rearing habitat provided (m^2)		Overall length of habitat type (km)	Percent of total length
	Chinook	Steelhead		
A- Pool-Cascade-Boulder: Much suitable rearing habitat, long deep pools with instream cover	52,954	153,809	15.8	23.1
B- Spawning (Class 1): Small substrate, shallow, slow water, most suitable spawning habitat.	NR	NR	3.6	5.3
C- Spawning (Class 2) - larger substrate but still usable, shallow, slow water. Some rearing habitats present	36,348	91,412	10.7	15.6
D- Riffle-ripple-run: Predominantly fast, smooth water; rearing habitats provided in backwater areas behind boulders, undercut banks and instream woody debris.	36,086	104,751	14.9	21.8
E- Riffle-Ripple: Fast, broken water. Some rearing habitat in slack waters by boulders and edge of stream	34,511	89,264	23.4	34.2

In the Panther Creek system, summer rearing habitat is the limiting factor for anadromous fish production, rather than spawning areas. Unlike suitable spawning habitat, the majority of habitat required for steelhead and chinook rearing occurs downstream from Big Deer and Blackbird Creeks. For both, about 40% of the rearing habitat in the Panther Creek drainage occurs between the mouth and Big Deer Creek and about 70% occurs between the mouth and Blackbird Creek. These stream surveys emphasize the significance of the lower reaches of Panther Creek below the mine influences for smolt rearing. Juvenile salmonids would rear in lower Panther Creek for several months; water quality needs to be sufficient to not harm juvenile salmonids during this length of exposure for rearing habitat to maintain steelhead or chinook in the Panther Creek drainage. Metals concentrations that have been shown to harm juvenile salmonids are compared to Panther Creek conditions in Section V.

Less than five percent of the of the habitat suitable for chinook salmon spawning in the drainage is below Big Deer Creek, including available habitat in Clear Creek. For steelhead, *only 1.5 %* of the spawning potential exists below Big Deer Creek (Reiser 1986). Thus for successful reproduction in the Panther Creek system, water quality conditions in the lower reaches must not interfere with the fishes' passage to the spawning habitat upstream of Blackbird Creek. Studies linking copper concentrations with disrupted upstream passage of adult salmon and downstream passage of juvenile salmon are also compared to Panther Creek conditions in Section V. Figure 4 shows the approximate locations of chinook salmon spawning habitats in the Panther Creek system.

Cascades in Big Deer Creek block passage to anadromous fish at about 1.0 km above the confluence of Panther. Other than Blackbird Mine effluents, the Big Deer drainage is currently a roadless, largely pristine area, although the Forest Service plans to build a bridge across Panther Creek and a road up the Big Deer drainage in order to promote logging in the drainage (USFS 1993). Blackbird Creek could potentially be habitat for anadromous and resident fish. The drainage is currently uninhabitable by most aquatic life below the confluence of mine effluent from Meadow Creek. The riparian zone in the Blackbird floodplain is denuded of vegetation in areas where mine tailings are deposited near the banks. However, Blackbird Creek has a similar size, gradient, and valley type as Musgrove Creek, which does provide habitat for anadromous and resident fish.

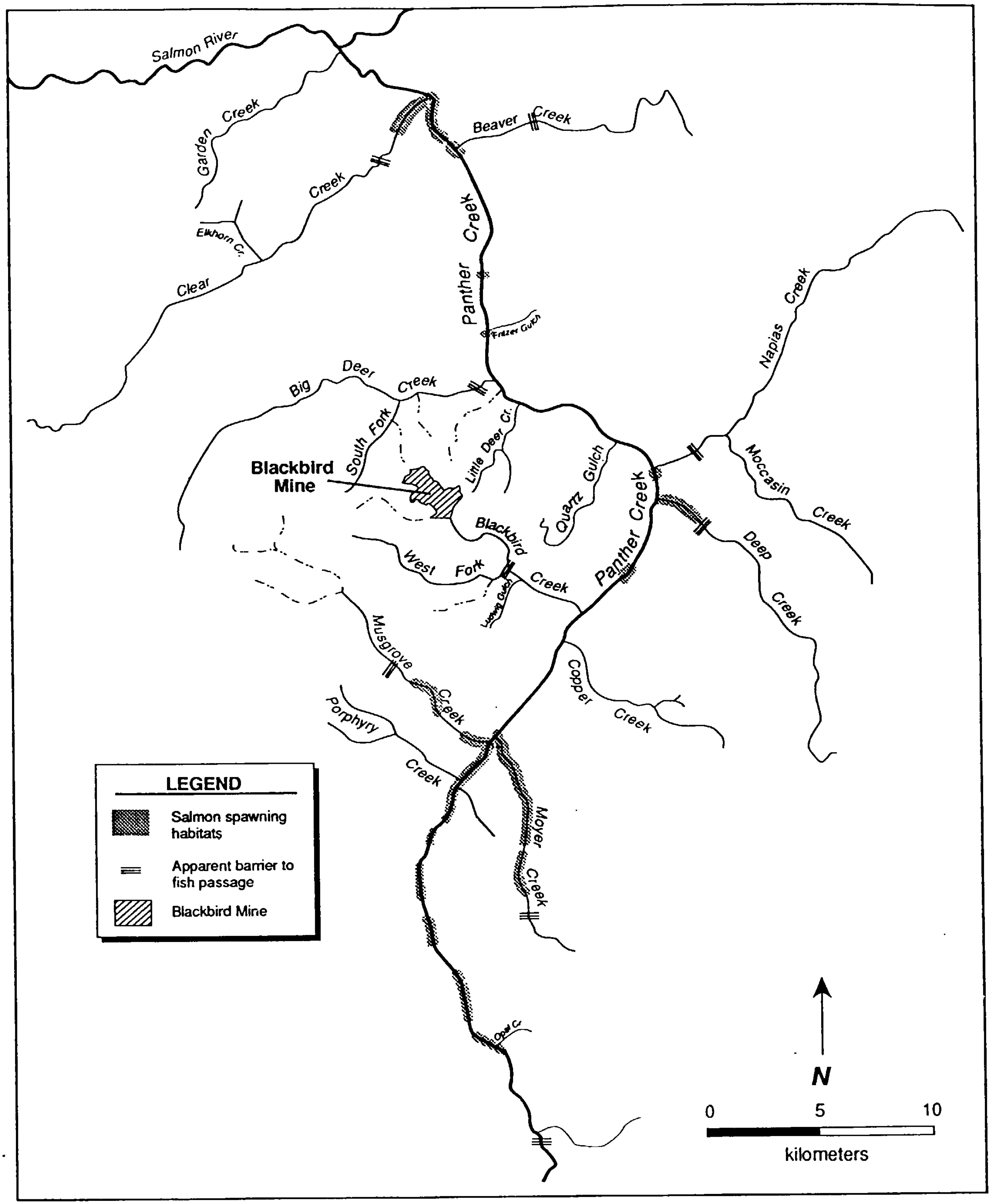

Figure 4. Salmon spawning habitats in the Panther Creek drainage. Rearing habitats occur throughout all reaches of Panther Creek and in the lower reaches of several tributaries (from Reiser 1986).

The substrates of salmonid streams are important habitats for incubating embryos and aquatic invertebrates that provide much of the food of salmonids, and they provide cover for fish in summer and winter. Silt and sand substrates have little or no value as cover for fish. Larger substrate materials provide visual isolation and their interstitial spaces are often the primary cover, along with depth and water turbulence, in some streams. Salmonids will hide in the interstitial in stream substrates, particularly in winter when the voids are accessible. The summer or winter carrying capacity of the stream for fish declines when fine sediments fill the interstitial spaces of the substrate (Bjornn and Reiser 1991).

Reiser (1986) reported that stations in Panther Creek upstream of Blackbird Mine and in six unaffected tributaries, fine sediments filled an average of 10% of the substrate interstitial spaces (range of 0-15%, nine study stations). At the Panther Creek stations downstream of the Blackbird Mine, fine sediments filled an average of 40% of the substrate interstitial spaces (range 25-60%, five study stations).

Resource Utilization

The "spring/summer" chinook are the only chinook that are found in the Salmon River drainage. Two groups of steelhead occur in the Salmon drainage. The "A-run" are fish that pass over the Bonneville Dam by August 25; they are predominately 1-ocean fish[4] that are about 63-70 cm long and average about 3 kg. The "B-run" fish that pass Bonneville later, are predominantly 2-ocean fish, weighing 5-6 kg and are about 80-88 cm long. The Panther Creek steelhead are all "A-run" (CBFWA 1990). Figures 5 and 6 show seasonal distribution and the life history of chinook and steelhead in the Salmon River sub-basin. Fish species occurring in the Panther Creek drainage and chinook salmon redd (gravel nest) are listed in tables 2 and 3. Chinook salmon spawning counts are traditionally made from their distinctive redds which may be counted from the air. No systematic counts of steelhead spawning by stream have been made in Idaho since they spawn in the spring when the water is sometimes turbid.

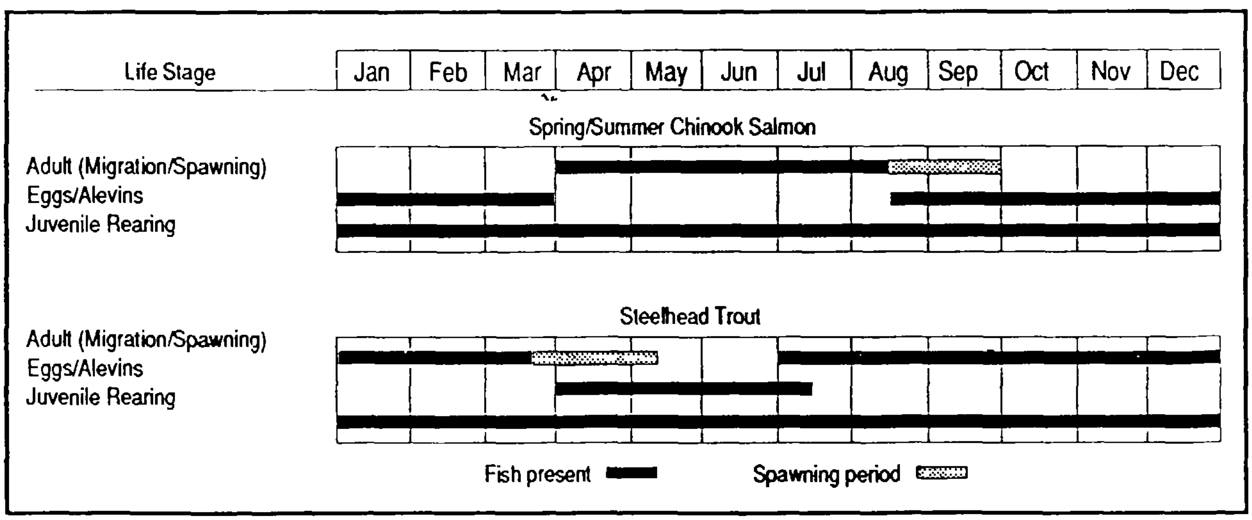

Figure 5. Freshwater life history stages of chinook salmon and steelhead trout in the middle-Salmon River sub-basin (Everest and Chapman 1972, CBFWA 1990).

[4] Refers to the number of years spent in the ocean: a 1-ocean fish has spent one year rearing in the ocean.

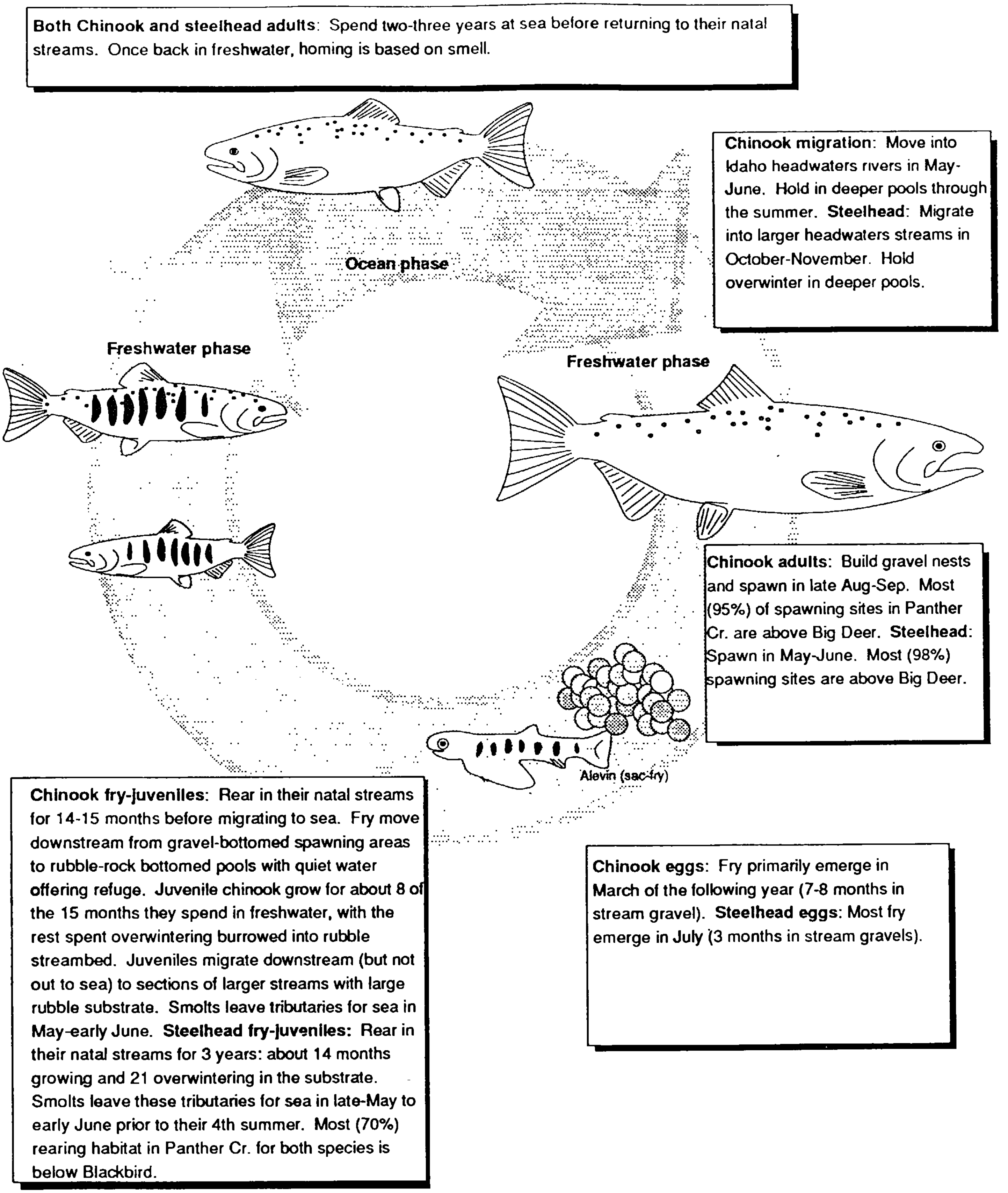

Figure 6. Life history and habitat needs for steelhead trout and chinook salmon in the Salmon River sub-basin (Life history from Orcutt et al. 1968; Chapman and Bjornn 1969; Everest and Chapman 1972; and Healey 1991; habitat needs from Reiser 1986 and Bjornn and Reiser 1991).

Table 2. Fish species occurring in the Salmon River and Panther Creek drainages.

Common Names	Scientific names	Occurrence[1] Salmon River	Panther Creek	Big Deer Creek (2)	Blackbird Creek (3)
Anadromous fishes					
Family Salmonidae					
Snake River sockeye salmon (E)	*Oncorhynchus nerka*	■			
Snake River spring/summer chinook salmon (T)	*Oncorhynchus tshawytscha*	■	■		□
Steelhead trout	*Oncorhynchus mykiss*	■	■		□
Family Petromyzontidae					
Pacific lamprey	*Entoaphenus tridentatus*	□			
Resident Fishes					
Family Salmonidae					
Mountain whitefish	*Prosopium williamsoni*	■	■	□	
Cutthroat trout	*Oncorhynchus clarki*	■	■	□	□
Rainbow trout	*Oncorhynchus mykiss*	■	■	■	□
Brook trout	*Salvelinus fontinalis*	□	■	□	□
Bull trout	*Salvelinus confluentus*	□	■	□	
Arctic grayling	*Thymallus arcticus*	□	□		
Golden trout	*Oncorhynchus aguabonita*	□			
Family Acipenseridae					
White sturgeon	*Acipenser transmontanus*	■			
Family Cyprinidae					
Chiselmouth	*Acrocheilus alutaceus*	■	■		
Northern squawfish	*Ptychocheilus oregonensis*	□	□		
Longnose dace	*Rhinichthys cataractae*	■	■		
Speckled dace	*Rhinichthys osculus*	■	■		
Redside shiner	*Richardsonius balteatus*	■	■		
Family Catostomidae					
Suckers	*Catostomus spp.*	■	■		
Family Cottidae					
Sculpins	*Cottus spp.*	■	■	□	

Key: □ - Potential use - species could potentially occur in these habitats
■ - Known use - species reported in these habitats.
E - Endangered species; T - Threatened species

[1] Potentially occurring and Salmon river species from Sgro et al. (1981). Confirmed species from USFS 1992 snorkeling observations (Smith 1993), IDFG annual monitoring, or Cameron 1993 (Big Deer Creek).

[2] Trout occurred in upper Big Deer Creek only. No fish were observed in snorkeling or electro-shocking in lower Big Deer Creek below Blackbird Mine drainage.

[3] Trout potentially occur in upper Blackbird Creek only (confirmed in upper West Fork Blackbird). No fish have been observed in lower Blackbird Creek (Forster pers. comm. 1992). Potentially occurring species based on their occurrence in Panther Creek tributaries with similar size and gradient.

Table 3: Chinook salmon spawning observations (redd counts) and introductions in Panther Creek

Year	Number of Redds	Comments
<1945	1,000	IDF&G estimate for the entire Panther Creek drainage was 2000 adult spawners. Assumes two adults per redd. (a)
1951	83	(b)
1954	12	200 adult spring chinook killed by sulfuric acid release from Blackbird Mine found below town of Cobalt. Redds located near Beaver & Clear creeks. No live fish seen. (c)
1955	25	(a)
1956	55	Majority of redds above Blackbird Creek (a)
1957	135	Panther Creek closed to fishing; most redds below Blackbird Creek (a)
1958	115	Majority of redds below Blackbird Creek (a)
1959	-	Turbid water from Blackbird Cr. prevented observations (a)
1960	-	Water too turbid for observations due to placer mining (a)
1961	4	Redds were above Blackbird Creek between 4th of July and Porphyry Cr(d)
1962	10	Redds located below Blackbird Creek (a)
1963	0	(a)
1964	0	(a)
1965	0	(a)
1966	0	(a)
1967	0	(a)
1968	0	Spawning counts discontinued. Periodic field surveys were conducted between 1968 and 1977. No redds were observed (e)
1977	0	46,305 spring chinook fry from Rapid R. and 50,000 steelhead fry outplanted in Panther Cr (e,f)
1978	-	25,000 steelhead fry released in Clear Creek (e)
1982	-	118,000 steelhead fry outplanted in Panther Cr (e)
1983	-	One pair adult chinook salmon observed holding in Panther Cr. below the bridge at Beaver Cr. (h). 379 steelhead adults released above Blackbird Creek (e)
1984	-	677 steelhead adults and 265,000 fry released above Blackbird Creek ; 40,000 steelhead fry released in Musgrove Creek (e)
1985	-	150 steelhead adults, 238,000 smolts and 310,000 steelhead fry released in Panther Cr. 175,000 fry released in Moyer Cr. (e,f)
1986	-	3,383 spring chinook adults from released at the mouth of Panther Cr. (e,f)
		121 steelhead adults, 246,000 smolts and 177,500 steelhead fry released in Panther Cr. 182,500 fry released in Moyer Cr. and 265,500 in Musgrove Cr. (e,f)
1987	-	137,000 chinook eggs buried in Panther Cr near the mouth of Clear Cr. (f,)
1987	-	299,700 steelhead smolts and 172,500 steelhead fry released in Panther Cr. 102,500 fry released in Moyer Cr. and 102,500 in Musgrove Cr. (e,f)
1988	-	237,000 steelhead smolts released in Panther Creek (f)
1989	0	Steelhead observed spawning in lower reaches of Panther Creek at mouth of Clear Creek. Chinook spawning counts resumed (g). 282,000 steelhead fry released in Panther Cr. (f)
1990	2	Steelhead again observed spawning at mouth of Clear Creek. Chinook redds observed below confluence of Beaver Cr. (g)
1991	2	Steelhead observed spawning in lower reaches of Panther Creek at mouth of Clear Creek. Highest chinook redd again observed about 8 km above Salmon River confluence. USFS field survey (g)
1992	2	Potential chinook redds observed at Fritzer Gulch, about 1 km below Big Deer Cr. confluence (g,i)

Sources: (a) Corley 1967; (b) Hauck 1952; (c) Pirtle and Keating 1955; (d) Bjornn 1961; (d) Platts et al.1979; (e) CBFWA 1990; (f) Scully and Petrosky 1991; (g) Smith, pers. comm., 1992; (h) Petrosky and Holubetz 1985; (i) C. Mebane pers. observation 10/16/92

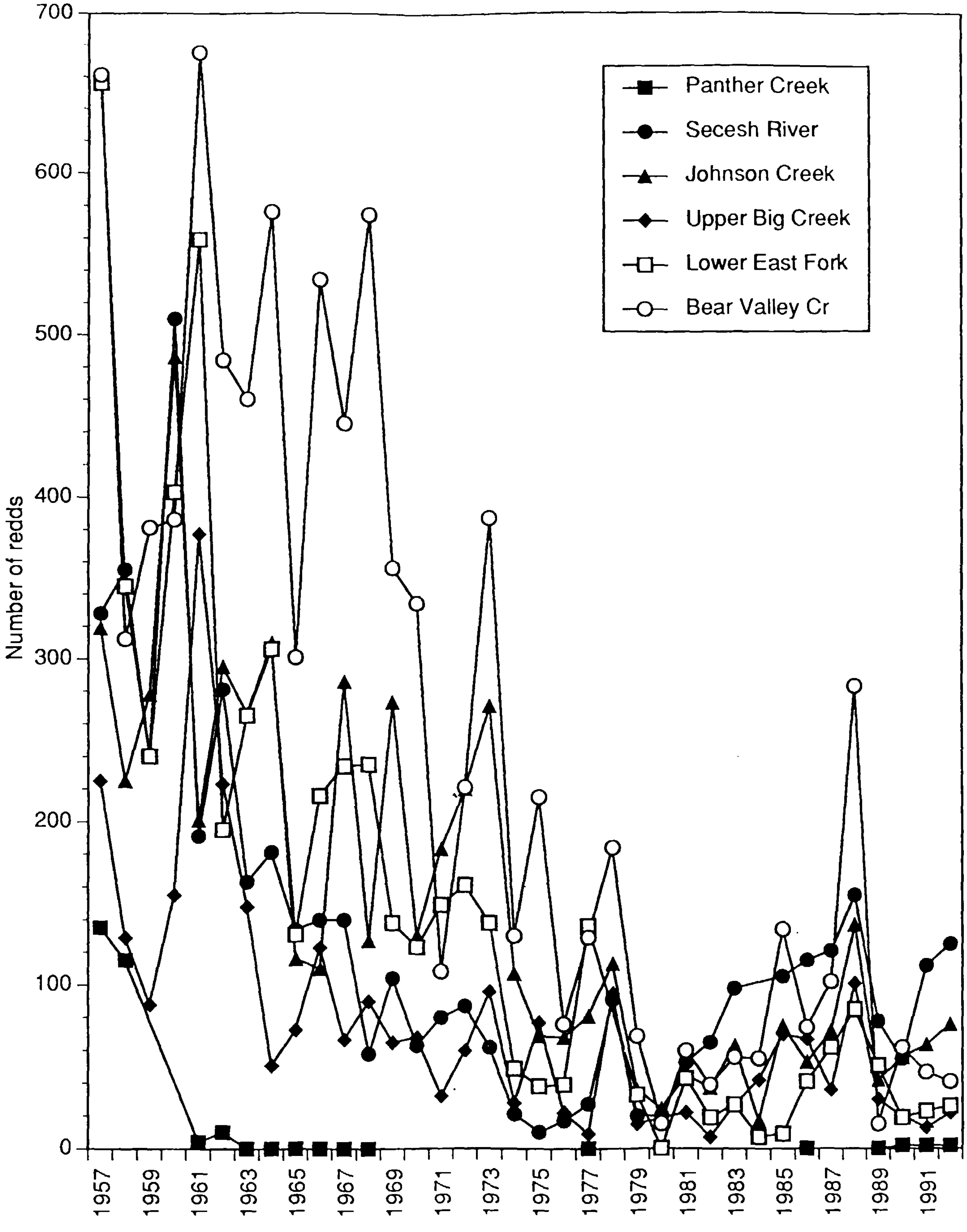

Figure 7. Comparison of chinook salmon redd counts in the Salmon River basin 1957-1992. Panther Creeks runs were already affected by the Blackbird Mine when these counts were started. (Source: Table 3; IDFG unpublished data 1992)

Anadromous fish populations throughout the Snake River drainage have precipitously declined due to many factors (e.g., dams, habitat degradation, overfishing) the decline in Panther Creek populations exceed that of nearby streams that also suffer the same downstream factors. Figure 7 compares the numbers of chinook salmon redds from various streams in the Salmon River drainage. Chinook spawning runs in Panther Creek began clearly declining in the 1950s, long before major downstream dam effects, and were completely eliminated by 1962 (Chapman et al. 1991). Efforts to reintroduce chinook and steelhead into Panther Creek have had less success than stocking efforts in other Salmon River streams (Table 3).

Commercial and Recreational Fisheries

Based on interviews of senior residents on anadromous fish catches in Lemhi county in the mid 1960s, the Idaho Department of Fish and Game (IDFG) estimated that, prior to 1945, salmon and steelhead returns in Panther Creek were similar to the Lemhi River's returns. From this the department estimated the annual return of adult chinook to Panther Creek to about 2,000 adults prior to 1945 (Corley 1967). Recently, two independent estimates of anadromous fish potential productions have been made. Reiser and coworkers estimated salmon and steelhead production based on detailed habitat inventories of Panther Creek and tributaries expected to support anadromous fish. Based on detailed field surveys of available habitats, limiting factors, and estimates of survival at different life stages, they estimated the smolt productions for Panther Creek and tributaries. The number of returning adult salmon and steelhead were based on smolt-adult returns to the IDFG Pahsimeroi hatchery, which is on a tributary of the Salmon further upstream from Panther Creek. For their estimates Reiser (1986) (1) assumed that all habitats degraded by water quality problems were restored and (2) used available habitat under different flow conditions. The Columbia Basin Fish and Wildlife Authority's System Planning model methods used a computer watershed database to estimate all accessible habitats mapped and did not involve field surveys. They also included an estimate of potential re-established chinook and steelhead in the Panther Creek drainage under existing degraded conditions. Both models reportedly considered downstream effects in their capacity estimates. The IDFG maintains a non-sustaining resident rainbow trout sport fishery on Panther Creek through annual "put and take" stocking (IDFG 1993).

Table 4: Estimated Panther Cr. salmonid smolt capacity and expected adult returns

	Chinook		Steelhead	
Estimates	Number Smolts	Returning Adults	Number Smolts	Returning Adults
Based on rearing habitat (limiting factor)				
Minimum	215,539	1,078	36,609	366
Maximum	847,080	4,235	173,746	1,737
Median	531,309	2,657	105,178	1,051
Columbia Basin Fish and Wildlife Authority Model				
If all available habitat were accessible and water quality restored in Panther Creek	471,518	2,170	106, 417	4,963
If all available habitat were accessible, but under existing water quality	42, 769	19	8,201	825

(Reiser 1986; CBFWA 1990; NPPC 1991; Kiefer pers. comm. 1992)

IV. Chemical Contaminants of Concern

The contaminants of primary concern to NOAA are trace elements that have been mobilized by mining and enter the Panther Creek watershed at higher concentrations than would naturally occur in an undisturbed highly mineralized drainage. Antimony, arsenic, cobalt, copper, iron, manganese, nickel and zinc occur in the Big Deer and Blackbird drainages above background levels (Beltman et al. 1993, Wai and Mok 1986). However, further from the mine in the larger downstream waters that could support salmonids (Blackbird, Big Deer, and Panther creeks), only copper, arsenic, and cobalt occur at levels of concern (exceed water quality criteria or at concentrations in sediments predicted to be harmful). This review focuses on the distribution of these three elements.

No studies of contamination at Blackbird Mine have been undertaken as part of the NPL remedial process. However many others have collected data on mineralization, the sources, extent, and fate and effects of the acid mine drainage at Blackbird Mine. The U.S. Geological Survey, Forest Service, Bonneville Power Administration, University of Idaho, State of Idaho, and Noranda have conducted studies at the site. Among the significant studies on contaminants of concern are those by Baldwin et al. (1978), Reiser (1986), and Beltman et al. (1993) which describe sources of contaminants at the site. Wai and Mok (1986) studied the effects of contaminated sediments on water quality and McHugh et al. (1987) described background concentrations of metals in the Blackbird region.

Background concentrations of metals in the Blackbird region

Metal-rich wastes from the Blackbird source areas and downstream contamination need to be evaluated in the context of natural background metals in the area. While dissolved copper concentrations of 0.4 to 4 µg/l are typical in surface waters of the United States away from the immediate influence of discharges (Stephan et al. 1994), ambient metals concentrations in water resulting from natural weathering and leaching of mineralized areas may be above national criteria levels. However, the natural background geochemistry may be obscured by the overprint of the mining activities. Natural background concentrations in mineralized districts need to be characterized to help devise realistic plans for remediation and monitoring. Three methods are generally described for estimating natural background geochemistry of water in mineralized areas that have been mined: examination of historical documents, comparison to natural concentrations in undisturbed, similarly mineralized areas, and predictive theoretical geochemical modeling (Runnells et al. 1992). In the Blackbird mining area, information from two of the three methods is available. Pre-mining disturbance geochemical surveys provide a historical record of metals levels. Natural metals concentrations in undisturbed mineralized areas in the region are also available for comparison.

Blackbird Mine deposit is part of a 50 km-long zone of mineralization with naturally enriched levels of copper, arsenic, and cobalt. Lund et al. (1983) identified areas with copper-cobalt mineralization similar to the Blackbird Mine based on (1) rock units similar to the stratabound deposit at Blackbird mine; (2) similar form of cobalt mineralization along Elkhorn Creek; (3) geochemical anomalies similar to those around Blackbird; (4) the areas occur within the northwest-trending belt of cobalt-copper mineralization (Idaho Cobalt Belt) that extends at least 50 km from Iron Creek through the Garden and Elkhorn Creek drainages to the north side of the Salmon River. Tables 5 to 7 present a summarize metals levels in surface water, stream sediments, and rocks and soils from drainages in the mineralized Idaho cobalt-copper belt areas and non-mineralized areas in the Blackbird vicinity.

Table 5. Background levels of metals reported in surface water in the vicinity of the Blackbird mining area (in µg/l, dissolved metals)

Surface water

Drainages with areas of reported high cobalt-copper mineralization

	Copper		Cobalt		Arsenic		pH		Source
	Min.	Max.	Min.	Max.	Min.	Max.	Min.	Max.	
Bucktail Creek-head (pre-mining disturbance		NR		200		NR		NR	A
Bucktail Creek-mouth (pre-mining disturbance		NR		0.5		NR		NR	A
Little Deer Cr. - seep near head of southern fork *		250		312		NT		4.7	B
Little Deer Creek-Mouth	<1.0	4.4	5.3	10.7		8.4	7.2	7.3	B,C
Indian Creek-Mouth*	1.7	3.5	0.4	7.9		8.6		7.6	B,C
Elkhorn Creek-Mouth		13.0		0.4		1.6		7.2	C
Garden Creek	<1.0	<1.0	<0.1	0.2	2.0	3.2	7.4	7.9	C
Clear Creek	2.3	8.9	0.2	0.4	<1.0	1.3	7.8	7.9	C
Iron Creek *	1.7	40.0	0.2	80.0	1.0	6.3	6.9	7.8	C

Drainages in the Blackbird mining area without areas reported of high cobalt-copper mineralization

	Copper		Cobalt		Arsenic		pH		
	Min.	Max.	Min.	Max.	Min.	Max.	Min.	Max.	
Big Deer Cr. above Blackbird mine influences	<1.0	4.5	<1.0	1.7	<1.6	3.1	7.2	8.2	C,D
South Fork Big Deer Cr. above Blackbird mine influences	<1.0	6.6	0.2	0.4	<1.8	1.2	7.9	8.1	C,D
Musgrove Cr. and springs	2.4	5.6	0.9	2.7	2.6	12.0	4.6	7.4	C
Porphyry Creek	1.5	5.4	0.2	0.3	<1.0	1.1	6.8	7.3	C
Panther Creek upstream of Blackbird	2.7	10	<1.0	<10	0.8	1.1	7.0	8.6	D,E

Sources: A: Canney and Wing 1966; B: Mebane 1994; C: McHugh et. al. 1987; D: Beltman et al. 1993; E: Wai and Mok 1986;

NR Not reported

* Drainage has some mining disturbances

Table 6. Background levels of metals reported in stream sediments in the vicinity of the Blackbird mining area (in mg/kg, dry weight)

Stream Sediments: Drainages with areas of reported high cobalt-copper mineralization

	Copper		Cobalt		Arsenic		Source/Comments
	Min.	Max.	Min.	Max.	Min.	Max.	
Bucktail Creek-head (pre-mining disturbance)	-	10,000		400	NT	NT	A Single sample
Bucktail Creek-mouth (pre-mining disturbance)	-	400		200	NT	NT	A Single sample
Elkhorn Creek	7	100	<10	500	-	<200	n= 23. Median Cu value 15 mg/kg, Co median 10 mg/kg (B,C)
Garden Creek	5	500	5	500	<10	500	n=27 Median Cu value was 30 mg/kg, Co median was 30 (B,C,D)
Little Deer Creek-Head	28	780	238	278	NT	NT	D n=3
Little Deer Creek-Mouth	370	542	163	261	41	52	D, J n=5
Unnamed streams between Little Deer and Bucktail Creeks	20	297	10	436		<200	n=22. NE side of Blackbird Ridge (8069 ridge) (B,D)
Indian Creek	15	500	5	200	<10	200	B,C,D J n=11
Beaver Cr drainage-upper	27	151	10	14	NT	NT	D
Beaver Cr (Mouth)	14	27	11	12	NT	NT	D Below Copper King Mine
Copper Creek-Mouth	-	77	-	8	-	21	E Below Blackpine Mine, n=1
Iron Creek	15	25	10	15	NT	NT	F Authors conclude background Cu is 20 ppm n=2 (above mining activity)

Drainages in the Blackbird mining area without reported high cobalt-copper mineralization

	Copper		Cobalt		Arsenic		Sources
	Min	Max	Min	Max	Min	Max	
Big Deer Cr. above S. Fork (Blackbird mine influences)	5	150	5	50	1	5	C,G, H
South Fork Big Deer Cr. above Bucktail Cr	7	427	12	15		13	D, H (Single As sample)
Big Deer Cr. below S. Fork - pre-mining disturbance		10		10		NT	A
South Fork Big Deer Cr. below Bucktail Cr - pre-mining disturbance		80		100		NT	A
Yellowjacket Creek	5	70	<5	50	<5	5	G Median Cu value in upper Clear and Yellowjacket 20 mg/kg. n = 144
Clear Creek(including samples below mineralized Elkhorn)	<5	70	<5	20	<10	<10	B,C,D,G
Panther Creek upstream of Blackbird	8	67	4	130	7	148	E,I

Sources: A: Canney and Wing1966; B: Evans et al. 1993; C: Cater et al. 1975; D: Bennett 1977; E: This study F: Erdman and Modreski 1984; G: Hopkins et al. 1985; H: Hull 1992; I: Sauter and Wai 1981; J: Mebane 1994. NT- Not tested

Table 7. Background levels of metals reported in undisturbed rocks and soils in the vicinity of the Blackbird mining area (in mg/kg, dry weight)

	Copper		Cobalt		Arsenic		Source and comments
	Min	Max	Min	Max	Min	Max	
Blacktail open pit area prior to mining disturbance	60	2400	10	400	NT	NT	A Grid of 371 samples, median Cu value was 150 ppm, Co 60
Banks of Blackbird Creek-above Meadow Creek	30	700	10	100	NT	NT	A Transect of 66 samples, median Cu value was 100 ppm, Co 20
North side of Blackbird mining area (Inc. Little Deer, Quartz Gulch areas)	4	479	6	273		NT	B
Forest topsoil N. of open pit waste pile	1268	1441	122	142	8	10	C Two samples
Indian Creek	11	541	9	436		NT	B
Elkhorn Creek	5	1500	<5	700	<5	900	D n=9, median Cu 100 ppm, Co 10 ppm, As 10 ppm
Garden Creek	5	1500	5	70	<5	500	D
Lower Panther Cr canyon	5	1500	7	10	<5	500	D
Beaver Cr drainage (Copper King Mine area)	9	130	10	45		NT	B
Salmon River Canyon (near benchmark 3067)	> 20,000		> 2000			830	Single rock sample chipped from a cobalt bloom (D,E)
U.S. soils (average)		30		8		5	F

Sources: A: Canney et al. 1953; B: Bennett 1977; C: Farmer and Richardson 1980; D: Evans et al. 1993, E: Evans pers. comm., F: Lindsay 1979;

NT - Not tested

In addition to geochemical data, extensive outcrops of ferrecrete (literally, iron cement), a metalliferrous material resulting from weathering the sulfide-bearing ore zones, have been reported in upper Meadow Creek. These ancient streambed outcrops indicate that Meadow Creek has naturally had acid rock drainage, presumably also elevating metals levels downstream in Blackbird Creek (Peters 1981).

The most extensive data on background levels is for soils, with about 2000 samples from the 1953 Canney et al. study and 163 from the 1977 Bennett study. Canney et al. reported about two-thirds of the Blackbird site had elevated cobalt and copper soil levels, defined as exceeding 100 and 80 ppm respectively. Median soil copper concentrations were 150 mg/kg in undisturbed forested areas where the open pit was subsequently excavated; median soil copper concentrations were 100 ppm in undisturbed areas along Blackbird Creek (Table 7). The stream sediment chemistry data from the Blackbird region is also extensive - over 760 sample results were reviewed. In streams not reported to have high cobalt-copper deposits, median copper concentrations were 20 ppm; the range was <5 - 427 ppm. Cobalt concentrations ranged from <5-130 ppm; the median concentration was 15 ppm. Few data on arsenic were available; reported concentrations ranged from 1-148 ppm in sediments which where were not known to be contaminated by mining (Table 6). Reliable surface water data were the most limited data set, with data from 54 samples reviewed. In mineralized drainages, most copper concentrations were between 2 and 15 ppb, maximum dissolved copper and cobalt concentrations were 40 and 80 ppb, respectively. In the non-mineralized drainages, copper concentrations were from <1 to 10 ppb and cobalt concentrations were less than 10 ppb (Table 5).

From this review, it appears that streams with anomalously high natural background metals levels are small, first-order streams. Their metals signal is quickly lost or reduced once they join larger second or third-order streams. Median sediment copper concentrations reported from reconnaissance surveys of streams with undisturbed, highly mineralized areas occurring in the drainage were similar to streams without reported mineralized areas - 30 ppm and 15-20 ppm respectively. Meadow, Blackbird, and Bucktail Creeks were naturally mineralized areas before the mining disturbances occurred. Higher, "mineralized background" levels seem appropriate to compare with these streams. The other parts of Big Deer and Panther Creek drainages are generally outside the areas identified in the geochemical surveys as highly mineralized. For example, Clear Creek may be analogous to Big Deer Creek which has a highly mineralized tributary, Bucktail Creek. Clear Creek also has a highly mineralized tributary, Elkhorn Creek, yet stream sediment and surface water values are not significantly higher than the other streams in the region. Metals levels from the "non-mineralized" drainages may be near background values for Big Deer and Panther Creeks.

Source and Pathway Characterization

The four major sources of contaminants of concern to NOAA are (1) contaminated surface deposits resulting from mining activities, (2) contaminated water discharged from the mine openings, (3) contaminated groundwater discharged from seeps and (4) contaminated streambed sediments in source tributaries.

Surface deposits

Contaminated surface deposits at the site include waste rock piles, mill tailings, contaminated floodplain soils, and dredge spoils. Figure 8 shows the location of types of surface contamination at the Blackbird Mine. Limited chemical sampling of the surface wastes was conducted as part of the 1992 source identification study; results are summarized in Table 8:

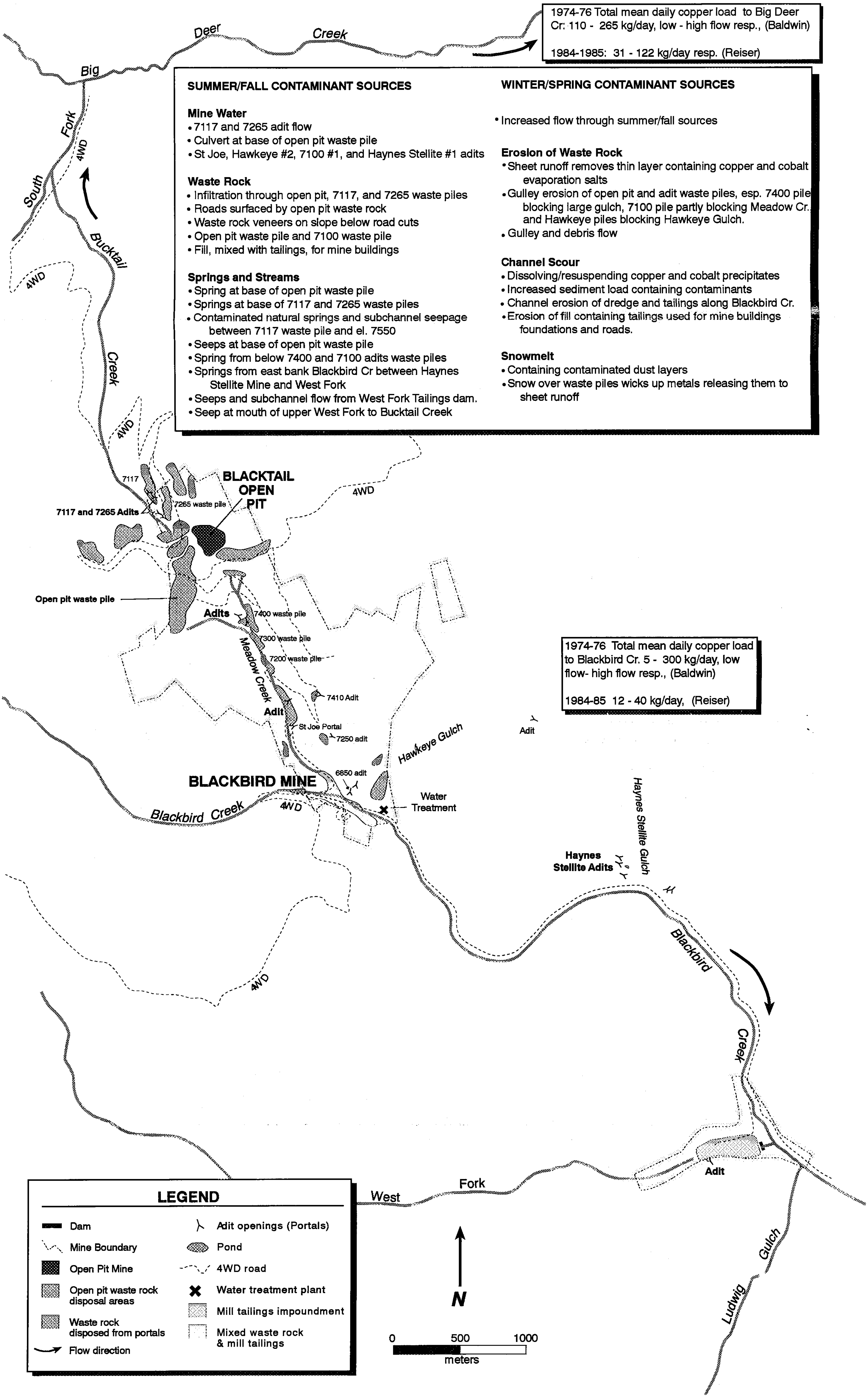

Figure 8. Sources of contaminantion in the Blackbird Mine area (compiled from Reiser (1986), USFS 1981 aerial phots, USGS Blackbird Creek and Gant Mountain 7.5 minute orthophoto and topographic maps).

Table 8. Contaminants levels in Blackbird Mine surface deposits (in mg/kg, dry weight)

Surface Deposits (Including tailings, waste rock and floodplain soils)	Copper		Cobalt		Arsenic		Source and comments:
	Min	Max	Min	Max	Min	Max	
Blacktail open pit	2180	5190	181	310	572	1070	A, B
Mine waste piles	1150	13,100	143	2390	440	5070	A
Efflorescent crusts on waste piles	1930	20,200	1060	2650	471	766	A
Soils and tailings in Blackbird Cr floodplain and along banks	171	1930	26	1060	1880	6720	A
Mill tailings	813	21,400	125	8990	685	4470	A, B

Sources: A: Beltman et al. 1993. B. IDEQ 1992.

Results of this limited sampling show metals levels in disturbed surface deposits in the mine area have higher concentrations than in undisturbed naturally mineralized area. The efflorescent crusts are alteration minerals that are formed from the oxidation of primary sulfides, including eyrithrite, a hydrous cobalt arsenate, and the copper carbonate hydroxide minerals malachite and azurite. All the alteration minerals identified at the site are very soluble in water, especially acidic water. High concentrations of metal and metalloids can be leached to area groundwater and surface water when alteration minerals are dissolved by snowmelt, rainfall, or shallow groundwater (Beltman et al. 1993, Nordstrom 1982).

Groundwater

Groundwater at the Blackbird area occurs in both fracture-controlled bedrock systems and in alluvial (unconsolidated surficial) deposits. Near-surface bedrock groundwater is expressed as seeps, springs, and mine portal discharges. Two major alluvial aquifer systems have been identified at the site: (1) natural alluvial material containing groundwater that is in direct hydraulic contact with streams at the site, and (2) groundwater flow through the large deposits of mine wastes in the Blackbird and Bucktail drainages (Baldwin et al. 1978). Copper and cobalt concentrations reported from groundwater expressed as surface springs and seeps, alluvial groundwater, bedrock groundwater, mine water, and mine adit discharges are compiled and summarized in Table 9. The few data available on arsenic in groundwater suggest that elevated arsenic concentrations occur in alluvial groundwater but less so in bedrock groundwater. Dissolved arsenic levels in two seeps in the Meadow Creek drainage were 17 and 710 µg/l. Concentrations in seven mine adit water samples ranged from <2 to 103 µg/l with all but one sample below 5 µg/l (Beltman et al. 1993).

Table 9. Metals in groundwater at selected sites in µg/l (dissolved metals except as noted)

	Copper		Cobalt		pH		Source
	Min	Max	Min	Max	Min	Max	
Springs and seeps							
Springs along Bucktail Creek	1,250	106,000	1,110	18,800	4.9	6.9	A
Springs along Meadow Creek	176	4420	1,200	6630	3.7	6.5	A
7700 open pit waste pile seep (head of Meadow Cr) 1985	425,000	650,000		NT	3.3	4.3	B (total Cu)
1974-1976	114,000	518,000	25,000	169,000	3.5	4.3	C (app. III-1)
7100 waste pile seep	21,800	200,000	17,100	106,000	2.7	3.1	C (app. III-1)
7400 waste pile seep	11,600	353,000	10,200	112,000	2.7	3.0	C (app. III-1)
Bedrock groundwater, mine water, and mine adit discharges							
Bucktail-Big Deer drainage							
7265 adit (1992)		112,000		66,800		2.7	A
1974-1976	37,600	346,000	63,000	216,000	2.6	3.1	C (App. III-1)
7117 adit (1985)*	220,000	525,000	NT	NT	4.4	4.4	B (total Cu)
1974-1976	12,100	35,900	44,700		2.8	3.2	C (App. III-1)
Blackbird drainage							
Mine workings	<100	46,900	<100	23,300	2.6	6.7	C
Diamond drill holes in mine bedrock	200	1200	1200	17,700	4.5	5.0	C
6850 adit (drains to WTP)	2610	3300	8480	8620		2.9	A,D
Treated 6850 adit discharge		4		41		9.1	A
Haynes-Stellite adit *		3		130		6.8	A
Alluvial groundwater							
Blackbird drainage							
Meadow Cr alluvium	<100	1800	<100	10,400	5.2	6.0	C (App. III-1)
Meadow Cr alluvium beneath mine waste	<100	90,800	<100	75,800	3.9	7.0	C (App. III-1)
Base of West Fork Blackbird Cr. tailings dam	<100	3400	4500	14,600	2.9	3.3	C (App. III-1)
Bucktail-Big Deer drainage							
Open pit waste pile (head of Bucktail Cr.)	425,000	1,070,000	157,000	1,470,000	4.1	4.4	C (App. III-1)
7265 adit waste pile	1,900	9,200	110,000	507,000	3.8	4.4	C (App. III-1)

Sources: A - Beltman et al. 1993; B- Reiser 1986; C - Baldwin et al. 1978; D - IDEQ 1992

*	Dry in Fall 1992
NT:	Not Tested
WTP	Water treatment plant

Sediments in pathway streams

The Big Deer and Blackbird drainages are pathways for contaminants in the source areas to reach anadromous fish habitats in Panther Creek. Several investigators have sampled stream sediments in the Blackbird region as part of geochemical mineral surveys and Blackbird water quality studies. Stream sediments in these pathways are grossly contaminated with metals from the site (Table 10).

Table 10. Contaminants levels in stream sediments draining from the Blackbird mining area (in mg/kg, dry weight)

Sediments							
	Copper		Cobalt		Arsenic		Year/Source
	Min	Max	Min	Max	Min	Max	
Blackbird Cr. drainage							
Blackbird Creek	940	9000	260	1560	630	3790	1980 A
	2330	2595	312	471	1121	2550	1985-1986 B
	4080	7230	413	977	622	1270	1992 C, D
Big Deer Creek drainage							
Bucktail Cr.-head	1600	2700	152	274	232	1050	1980 A
Bucktail Cr.-mouth		19,000		240		1020	1980 A
		15,600		238		698	1992 B
S. Fork Big Deer Cr.	22,000	91,400	330	1100	670	987	1977-1980 A,E
Big Deer Cr.	220	2410	65	104	<50	139	1977-1980 A,E
	286	941	20	72	4	11	1992 C

Sources: A: Sauter and Wai 1981; B: Mok and Wai 1989; C: IDEQ 1992; D: Howell 1992; E: Bennett 1977

Surface water pathways

Surface water chemistry has been studied at the site since the 1960s. Values from the earlier data sets could not be directly compared between sets since methods for sample collection, handling, and analyses either varied or were not reported. Studies which described using standard sample collection, filtration, preservation, and analysis are summarized in Table 11. Results show the same pattern as sediment chemistry in the Panther Creek tributaries. Metals levels in Blackbird and Big Deer Creeks upstream of Blackbird Mine drainages were low, they were greatly increased below the mine inputs, and still greatly elevated at their mouths at Panther Creek.

Metals in snowmelt

The winter snowpack overlying the exposed surface deposits of mine waste and the open pit is a significant agent for concentrating and mobilizing metals contamination. Farmer and Richardson (1980) investigated snowpack, snow meltwater chemistry, streamflow and stream chemistry at the site between January 1975 and July 1976. Their results showed extremely high metals concentrations in the snowpack, acidic snowmelt runoff with high metals, and a distinct spike in stream copper concentrations from the flush of the first melt in late winter/early spring (Table 12). Later during peak spring runoff (when most other Blackbird water chemistry data sets were collected) copper concentrations were much lower.

Table 11. Contaminants in surface water pathways from the Blackbird mining area (in µg/l, dissolved metals)

Surface water pathways to Panther Creek: Tributaries to Panther Creek receiving Blackbird Mine drainage									
	Copper		Cobalt		Arsenic (total)		pH		Source
	Min	Max	Min	Max	Min	Max	Min	Max	
Blackbird drainage									
Blackbird Cr. above Meadow Cr	<1.5	4.4		<1.0		<2	7.5	7.6	B
Meadow Creek	12,900	19,800	4410	14,200	<2	24	3.3	4.7	B·
Blackbird Creek below Meadow Creek	76	2910	330	2770	0.4	4.6	3.1	7.6	A,B
Big Deer drainage									
Big Deer Cr. above S. Fork Big Deer Cr	<1.0	4.5	<1.0	1.7	<1.6	3.1	7.2	8.2	B,D
S. Fork Big Deer Cr. above Bucktail Cr.	<1.0	6.6	0.2	0.4	<1.8	1.2	7.9	8.1	B,D
Bucktail - headwaters	27,000	308,000	34,800	150,000	<2	6.5	3.8	4.4	C,D
Bucktail Cr. - mouth	2380	4620	2080	3540		<2	7.2	7.2	B,D,E
S. Fork Big Deer Cr.	560	980	480	1300	<2	2.2	7.5	8.0	B,D
Big Deer Creek	44	310	41	220	<2	17	7.5	8.0	B,D,E
Sources: A: Wai and Mok1986, B. Beltman et al. 1993; C: Hull et al. 1992 (pH values only); D. McHugh et al. 1987; E. IDEQ 1992.									

Table 12. Metals snowpack, snowdust, and meltwater runoff in source areas (µg/l total metals, except snowdust, mg/kg dry weight, Farmer and Richardson 1980)

	Copper		Cobalt		Arsenic		pH	
	Min.	Max	Min	Max	Min.	Max	Min.	Max
Windblown snowdust scraped off the surface of snowpack, melted, decanted, and dried. Composited to 2 samples	5157	5350	505	900	105	1941	N/A	N/A
Snow	<100	22,100					3.8	7.0
Surface runoff from snow melt (1st snowfall of season-October)	<100	595,000					2.2	4.3
Surface runoff from snow melt (rapid spring melt-May)	<100	315,000					3.9	5.5
Meadow Cr (peak runoff- minimum; early spring melt -maximum	12,000	68,000					3.3	4.6
Blackbird Cr (peak runoff - minimum; early spring melt -maximum	1600	6000					4.2	5.7

Metals loading from surface water pathways

Copper and cobalt loading to Panther Creek were estimated by flow and concentration measurements from 1974-1976, 1980-1981, and 1984-1985 by Baldwin et al. (1978), Davies (1982) and Reiser (1986) with the most comprehensive data collection during 1974-1976 (Table 13). For 1976 only, Baldwin et al. (1978) estimated annual copper and cobalt loading from the mine for the entire year (Table 14). While the loadings reported vary widely between years, the higher loadings are from the Bucktail drainage, in spite of its much smaller discharge. During Baldwin's 1976 annual loading calculations, the loading contribution from the 6850 Portal, which has subsequently been collected and treated, accounted for only 13% of the Blackbird drainage copper loading and 3% of the overall copper from both drainages. Decreasing flows and concentrations in lower Blackbird Creek suggest losses to the alluvium, and thus contaminated alluvial groundwater. In both drainages, concentrations and loading are reported to decrease with distance from the sources, as waterborne metals precipitate out on the streambed sediments. Each of the three studies made mass balance comparisons of metals inputs from discrete sources in the Meadow Creek drainage and outputs below the mine. All found approximately 50% of the loading unaccounted for, suggesting significant diffuse, or non-point, loading.

Table 13. Daily average metals loading (kg/day) reported for the Blackbird and Big Deer Drainages, 1974-1976, 1980-1981, 1984-1985

		Stable (low) Flow	High Flow
Blackbird Drainage Loadings			
Meadow Creek	(1974-1976)	5.4	300
	(1984-1985)	3.2	13
6850 Portal	(1974-1976)	2.7	64
(Subsequent years - see table notes)		<0.1	-
Blackbird Cr. below all contaminant. sources (below confluence of W. Fork, 4 km above mouth)			
	(1980-1981)	11	52
	(1984-1985)	8.9	41
Blackbird Cr. at the mouth	(1984-1985)	6.5	26
Big Deer drainage loadings			
Bucktail Cr. (mouth)	(1974-1976)	108	265
	(1984-1985)	51	-
Big Deer at the mouth	(1984-1985)	31	122

Sources: 1974-1976 - Davies (1982); 1980-1981 and 1984-1985 Reiser (1986).
Notes: Mine water from the lowest opening (6850Portal) was routed through a lime treatment plant starting in December 1980. Later loading measurements did not include treated discharge; this low flow estimate was calculated from Beltman et al. (1993) flow and concentration data.

Table 14. Annual metal loading (tons/year) Blackbird and Big Deer drainages in 1976 (Baldwin et al. 1978)

	Copper	Cobalt	Iron	Discharge (acre-ft)
Blackbird Creek	13.4	7.5	15.2	5800
Bucktail Cr. (Big Deer drainage)	17.5	4.9	Not reported	97

Habitat Exposure Characterization

Panther Creek Surface Water

Much chemical data on Panther Creek surface water has been collected since the late 1960s, including time-series monitoring. As with the water chemistry reported from the pathway tributaries, values from the earlier data sets could not be directly compared between sets since methods for sample collection, handling, and analyses either varied or were not reported. Also, both Reiser and Davies reported in their reviews that some samplings were chosen from sites below Big Deer and Blackbird Creeks where complete mixing had not occurred. This generally skewed metal concentrations below Blackbird Creek too high and below Big Deer Creek too low. Only the data sets from Wai and Mok (1986) described collecting samples over a range of locations and times using standard sample collection, 0.45 µm filtration, preservation, and analysis. These are summarized and compared with the water quality criteria for protection of aquatic life in Table 15. For the four time-series monitoring studies available for review for which values could not be confidently compared between the data sets, the data from stations up and downstream from Blackbird and Big Deer creeks are compared within the sets in Figure 9.

All of the time-series monitoring data sets show clear patterns of consistently elevated copper levels below Blackbird and Big Deer Creeks compared to levels upstream. The copper levels in the reach of Panther Creek just above Big Deer Creek generally decrease with dilution from Napias Creek, a large tributary that almost doubles the volume of Panther Creek. Levels generally increase again below Big Deer Creek. In three of the four monitoring studies, higher concentrations were reported during the high-flow spring/early summer runoff.

Table 15. Metals levels in Panther Creek surface waters in µg/l dissolved metals (1985-1986 spring-summer sampling)

	Copper		Cobalt		Arsenic		pH (B)		Sources
	Min	Max	Min	Max	Min	Max	Min	Max	
Upstream of Blackbird Creek	<10	10	<10	<10	<0.02	0.4	6.6	8.6	A - all metals values
Below Blackbird Creek.	20	160	30	120	0.03	6.2	6.6	8.4	
Below Big Deer Creek	10	60	20	40	1.8	2.6	6.4	8.5	
Ambient water quality criteria (chronic - acute)	6.5 Note: (1)	9.2	5 Note: (2)	144	Note: (3)	190	6.5	9.0	D

Sources: A: Wai and Mok 1986 (all metals data); B: Hull et al. 1992 (pH values only); D: EPA 1986

Notes: (1) Copper criteria is hardness-dependent; value is for a hardness of 50 mg/l as $CaCO_3$. Reported hardness values for Panther Creek range from 10 - 60 mg/l (Davies 1982; Hull et al. 1992; Smith 1993). (2) Cobalt has no national criteria; values are from chronic-acute toxicity for soft water discussion in Section V. (3) Criteria listed are for trivalent arsenic, generally considered more toxic than other inorganic arsenic species, rather than total arsenic species. Listed arsenic values are for total species; most arsenic in Panther Creek was in the pentavalent form; the highest trivalent concentration reported was 2.1 µg/l.

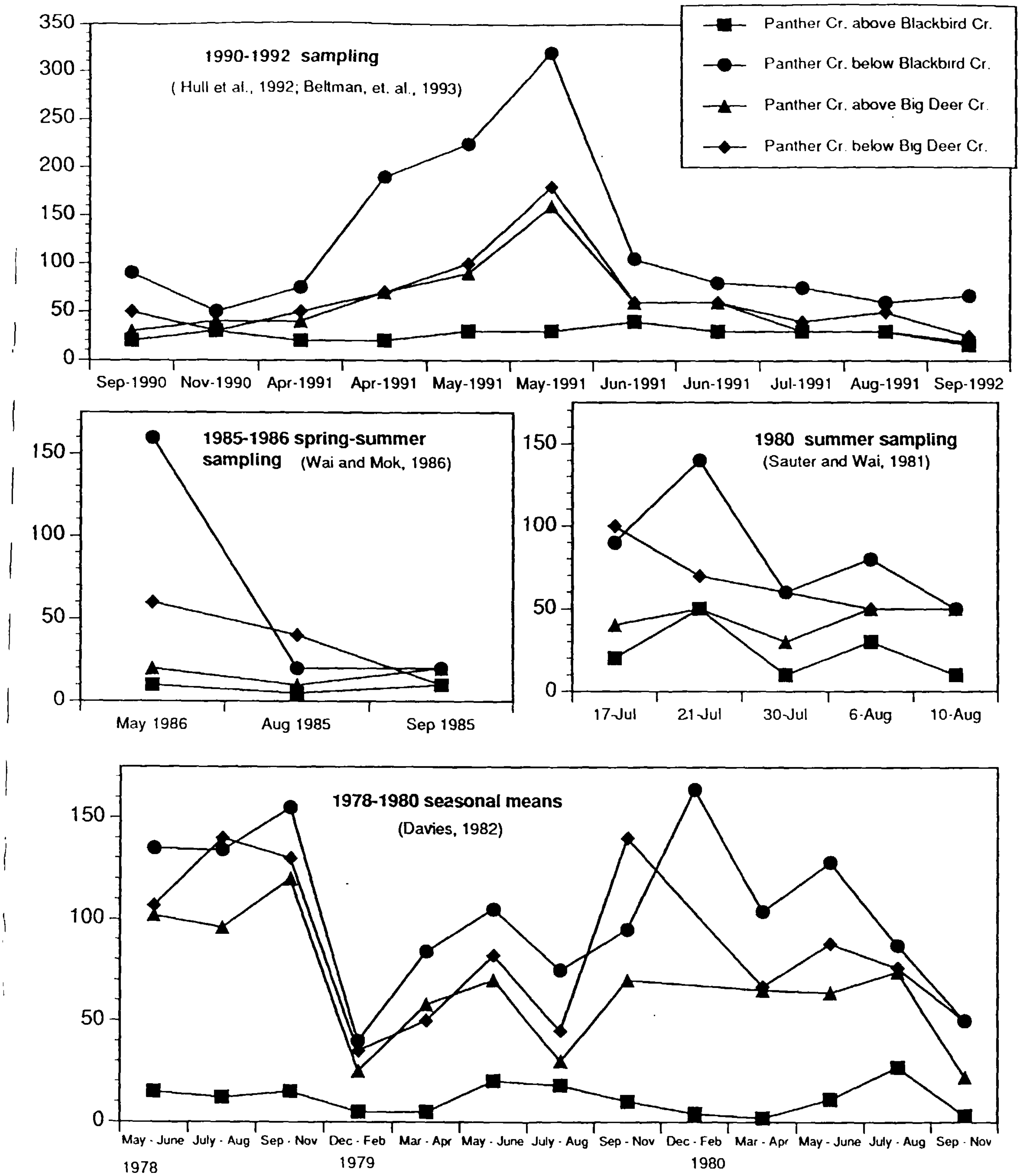

Figure 9. Compilation of Panther Creek copper concentrations above and below the Blackbird Creek and Big Deer Creek confluences. Concentrations in (μg/l). 1990-1992 data are for total copper, all others are for the "dissolved" (filtered) fraction.

Sediments

The objectives of the sediment sampling and analysis portion of the present study were to: (1) reconnoiter metals concentrations in streambed sediments, and (2) conduct a pilot study of sediment bioavailability through toxicity testing. The results were intended to help refine more comprehensive investigations that may be needed to characterize the site and to support cleanup decisions.

The current study examined the distribution of metals in Panther Creek sediments and their acute toxicity to the freshwater crustacean *Hyalella azteca*. A summary of the results of chemical sampling follow; results of the toxicity testing are described in section V. The methods and results of this study are also presented in more detail in Appendix A.

In October 1992, I collected surface sediments from nine Panther Creek stations between river kilometer 45 and river kilometer 8, and one sample from the mouth of Copper Creek, below the Black Pine Mine, a potential contamination source to Panther Creek (Figure 10). Station locations were selected to indicate Panther Creek sediment metals concentrations related to tributaries draining the Blackbird Mine (point sources) and distance and dilution by other tributaries to the Panther Creek drainage. Where possible, stations were sampled as a cross-section transect across the stream (e.g., near the west bank, midstream, and east bank). Depositional areas were located in slow water close to the banks, in relatively slow water mid-stream (e.g., at the tails of pools upstream from rapids, and in fast-water riffles and runs from eddies behind boulders mid-channel). All samples were analyzed for bulk metals concentrations, organic carbon content, and grain size.

Metals concentrations in Panther Creek sediments showed increases which were clearly associated with inflow from Blackbird and Big Deer creeks (Table 16, Figure 11). Copper levels in the sediments below Blackbird Creek were up to 300 times higher than the reference stations above its confluence. Levels generally dropped with distance downstream from Blackbird Creek as Panther Creek is fed by other tributaries. The largest of these, Napias Creek, nearly doubles the volume of Panther Creek. Copper levels just upstream from Big Deer Creek were only elevated 7-10 times the concentrations at the reference stations upstream from Blackbird Creek. Copper levels below the confluence of Big Deer Creek increased by 75 to 120 times over the upstream reference levels. At the lowest station sampled, about 32 river kilometers below Blackbird Creek, copper concentrations remained elevated 14 to 20 times the concentrations at the upstream reference stations. Cobalt and arsenic levels showed similar trends. There was no association between metals levels in Panther Creek and the Copper Creek drainage from the Black Pine Mine.

Controlling for sediment grain size by sampling fine-grained sediments is generally recommended for determining sediment contamination from anthropogenic sources. Significantly higher metal concentrations tend to occur on the clay and silt-sized fractions of sediments rather than on the larger sand and gravel fractions. The variability in grain sizes could hide a metals dispersion pattern from mine effluents or other anthropogenic sources. Methods to correct for grain-size variations to determine trends from a pollution source have also been developed (Horowitz 1991; Håkason 1984). However, the concepts of grain-size correction have mainly been developed in relatively low-energy, fine-grained river, lake or estuarine systems. These grain size corrections may be inappropriate for high-gradient, coarse grained rivers that drain mining areas because the relationship of increasing metal concentration with decreasing particle size is subdued. Larger particles stay in place longer, often in shallow oxygenated areas of a stream and therefore, in these systems, may have more time to accumulate oxide coatings and associated trace metals than smaller particles

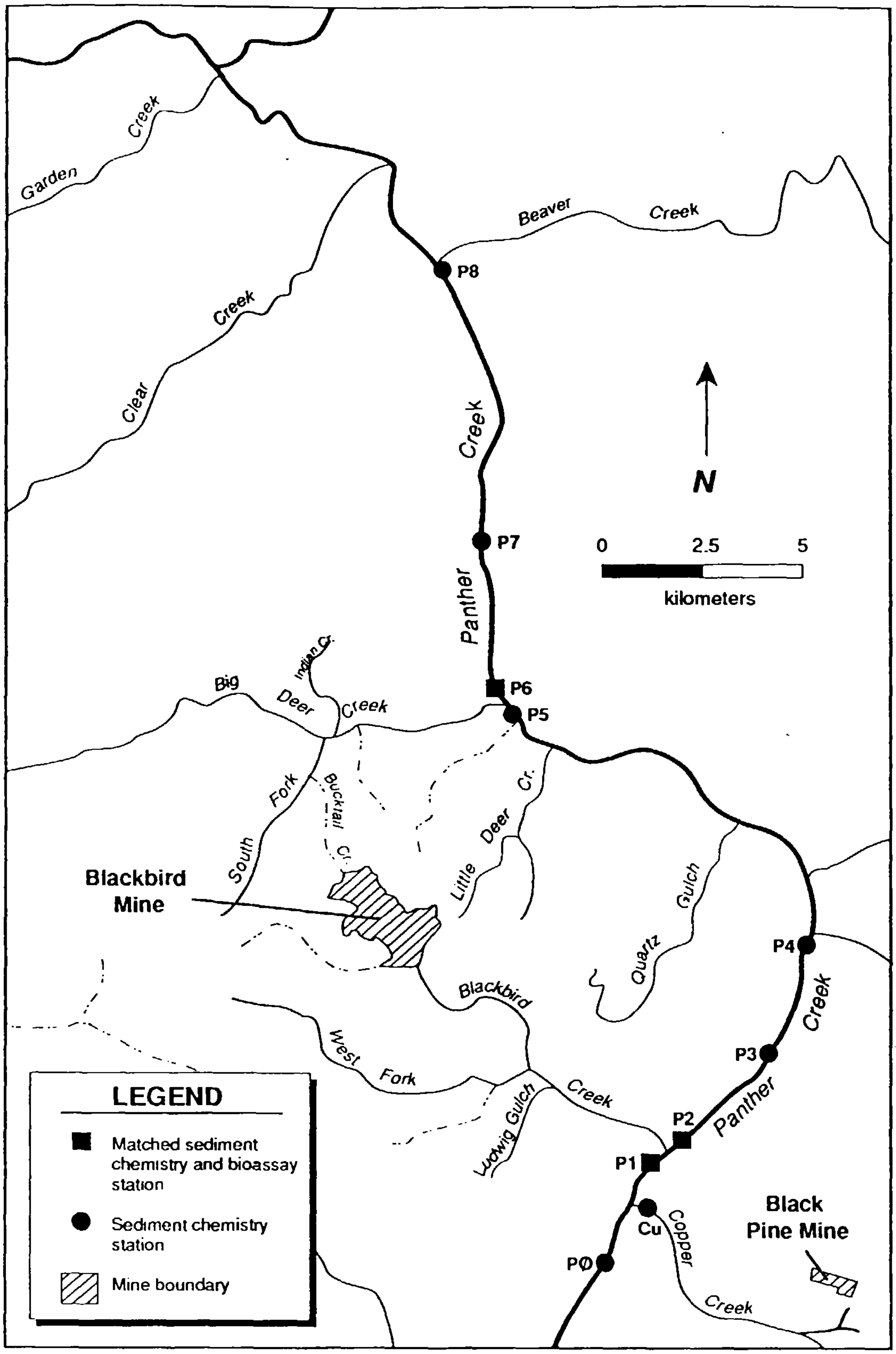

Figure 10. Panther Creek - Sediment sampling stations (this survey).

(Moore et al. 1989). Also, much of the food eaten by salmonid fishes in streams, benthic macroinvertebrates, is produced in the substrate. Invertebrates differ in their ability to thrive in different substrates. Chironomid midges of various species do well in silts and muds, but the larger mayflies (ephemeropterans), stoneflies (plecopterans), and caddisflies (trichopterans) prefer a mixture of coarse sands and gravels. Salmonids build their nests in coarse-grained sandy, gravelly sediments and juveniles are closely associated with coarse-grained substrates in cobble-rubble areas (Bjornn and Reiser 1991). Thus, to investigate whether aquatic insects and fish were exposed to elevated metals levels throughout the streambed environment, coarse-grained sediment samples from faster water mid-channel were included in this study.

Results are grouped in Table 16 by samples from finer-grained sediments that were found in slow water close to banks and more coarse-grained sediments were collected from faster water midstream. Arsenic, cobalt, and copper concentrations in faster water, midstream areas downstream of Blackbird Mine were elevated up to 25 times the highest upstream levels. The trend of increased metals leading from the Blackbird Mine effluents is so extreme that it overwhelms the variability expected from grain size differences: metals levels in fine-grained sediments, coarse-grained sediments, and sediments corrected for grain size differences[5] all show the same trend of very low metals levels upstream of Blackbird, greatly increasing below the Blackbird Creek discharge, dropping with dilution from Napias Creek, and increasing levels again below the Big Deer Creek discharges.

Sediment arsenic and copper concentrations are also compared to two benchmark biological-effects screening ranges in Table 16. The two ranges, the Ontario freshwater guidelines for protecting aquatic sediment quality (Persaud et al. 1993) and the NOAA guidelines for ranking stations sampled in the National Status and Trends program (Long and MacDonald 1992), are both based upon the co-occurance of adverse biological effects and ranges of chemical concentrations associated with them. Both give a lower bound to the effects range, below which adverse effects are seldom observed, and an upper bound, above which adverse effects are usually observed. Most downstream samples exceed the upper bounds of both ranges; no upstream samples exceed them. Additional literature reports on the effects of sediment-sorbed metals is compared to Panther Creek condtions in section V.

Comparison with previous studies Besides the present study, Sauter and Wai (1981) studied the distribution of metals in Panther Creek sediments. Their results showed a similar clear pattern of contamination associated with the confluences of Blackbird and Big Deer Creeks with Panther Creek. Figure 11 compares the 1980 and 1992 distributions and concentrations of arsenic, cobalt, and copper in Panther Creek sediments. There has been no decrease in metals concentrations shown over these twelve years.

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[5] Results of grain-size corrections are presented in Appendix A.

Table 16. Concentrations (mg/kg dry weight) of copper, cobalt, and arsenic in Panther Creek bed sediments compared with benchmark biological effects screening levels (this study, except as noted).

Station (Figure 10)	Copper Near Bank (range)	Copper Mid Stream	Arsenic Near Bank (range)	Arsenic Mid Stream	Cobalt Near Bank (range)	Cobalt Mid Stream
P0 - Above Copper Creek*	16 - -	-	7 - -	-	5 - -	-
P1 - Above Blackbird Creek	8 - 8	13	7 - 10	11	4 - 5	4
P2 - Below Blackbird Creek	2280 - 2890	-	766 - 888	-	436 - 554	-
P3 - Below Cobalt townsite	351 - 2930	149	167 - 344	82	475 - 1150	179
P4 - Above Deep Creek	694 - 1050	208	167 - 193	99	366 - 485	208
P5 - Above Big Deer Creek	94 - 109	65	40 - 56	27	112 - 128	108
P6 - Below Big Deer Creek	750 - 889	386	27 - 31	14	73 - 91	76
P7 - At Fritzer Gulch	1140 - 1200	135	78 - 147	25	314 - 553	63
P8 - Above Beaver Creek*	232 - -	-	25 -	-	106 - -	-
Mouth of Panther*(a)	1030 - -	-	135 -	-	544 - -	-
Lowest/severe effects to freshwater benthic communities (b)	16 110		6 33		16 50	
Effects ranges low - median (c)	34 270		8 70		- -	

* Single samples only for Stations P0, P8 and Cu. All other stations had three samples
(a) Mok and Wai (1989), (b) Persaud et al. (1993), (c) Long and MacDonald (1992)

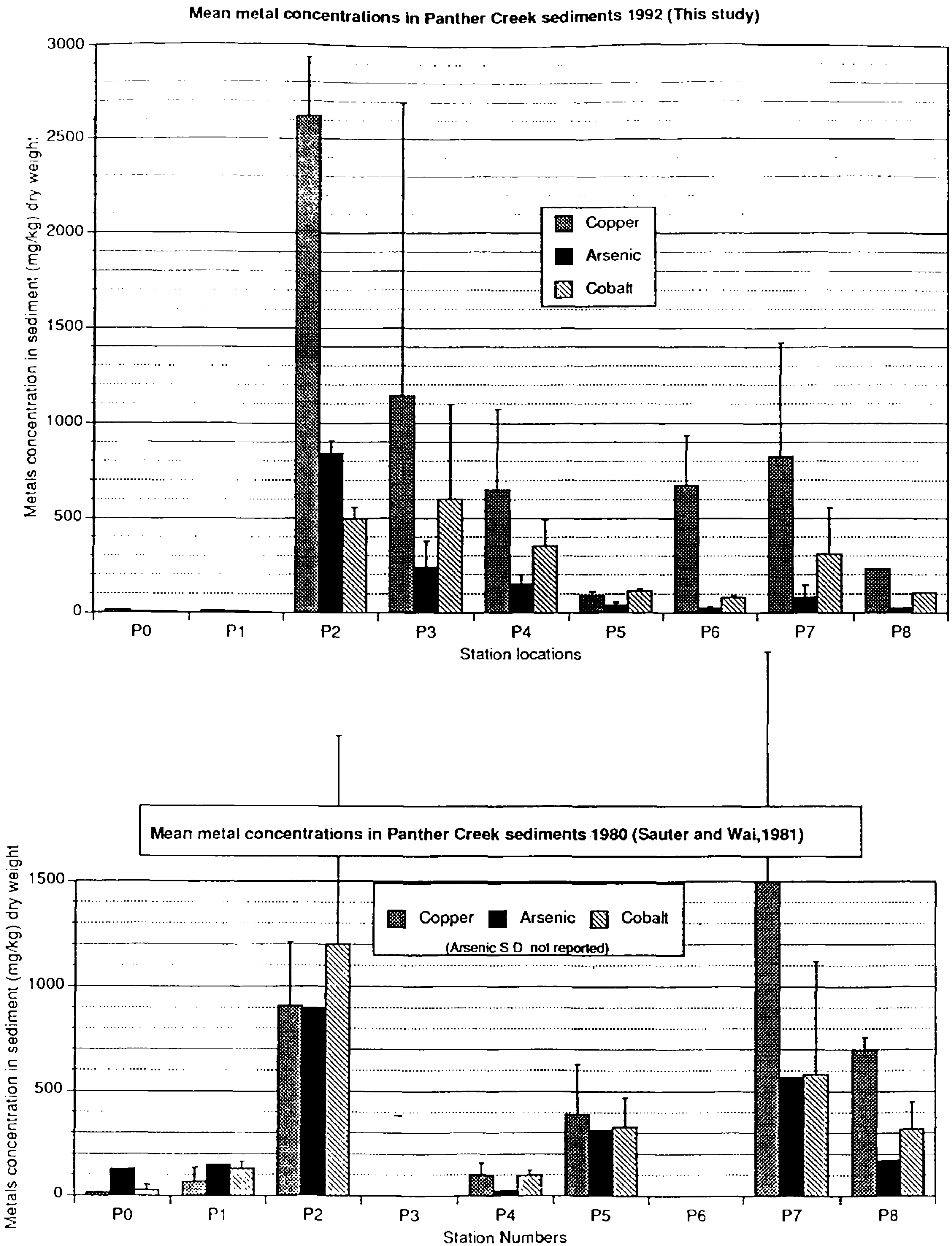

Figure 11. Comparison of 1980 and 1992 distributions of arsenic, cobalt, and copper concentrations in Panther Creek sediments. Error bars show one standard deviation; station locations are from Figure 10).

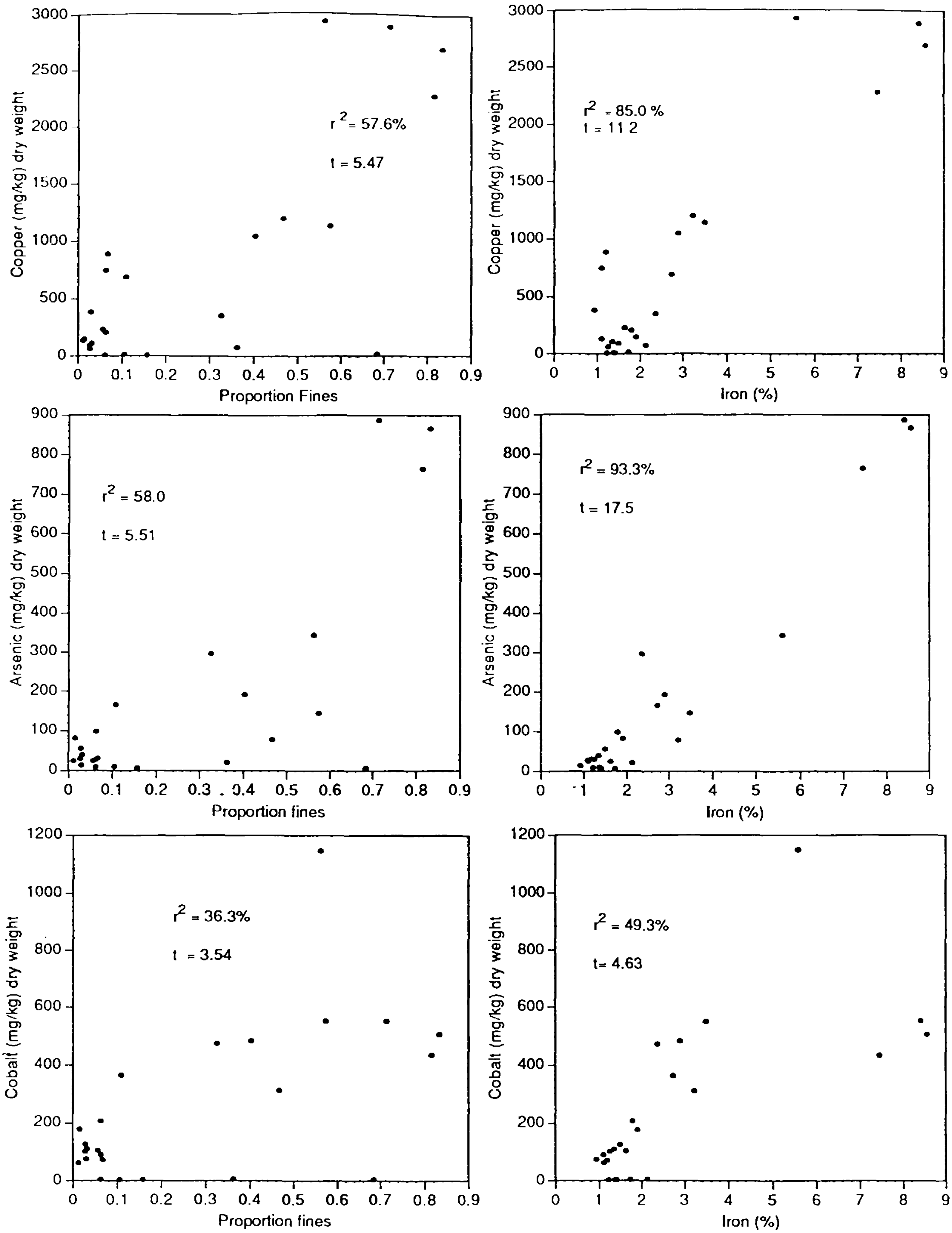

Figure 12. Selected regressions of copper, arsenic, and cobalt versus proportion fines and iron levels for all Panther Creek sediments (critical value of $t_{.001}$ for 22 degrees of freedom is 3.82, $t_{.01}$ is 2.83 (Zar 1984))

Effects of contaminated sediments in source areas on surface water quality

Wai and Mok (1986)[6] and Sauter and Wai (1981) performed leaching tests to determine the mobility of arsenic, cobalt, copper, and iron. They evaluated the short-term release of water-soluble toxic metals and the acid-producing potential from the stream sediments by mixing sediment samples with deionized water and measuring dissolved metal concentrations in the leachate (Table 17).

Table 17. Selected concentrations of water-soluble metals leached from stream sediments in the Blackbird mining area (in µg/l dissolved metals)

Highest equilibrium concentrations in the leaching solution after 10 days						
Sediment location	Cu	Co	As	Fe	pH	Source/Comments
Mine tailings	5000	6000	NT	13,000	3.5	A Initial release of Cu much higher
Lower Bucktail Creek	30,000	60	NT	2500	5.3	A
Blackbird Creek	6500	2000	NT	18000	3.5	A
	2650	12,480	11	5550	4.4	B
Panther Creek below mine influences	50	10	NT	10	6.5	A
	50	310	118	570	6.5	B
Big Deer Cr. above mine effluent	10	10	NT	10	6.5	A
Sources: A: Sauter and Wai 1981; B: Wai and Mok 1986. NT - Not tested						

Dissolved leachate copper concentrations of 50 µg/l following the stirring of Panther Creek sediments indicate that the release of copper and cobalt from contaminated in-place sediments alone, even if complete source control were achieved, could result in concentrations that could be toxic in Panther Creek surface water. Much higher concentrations of copper and cobalt resulted from leaching experiments with the highly contaminated sediments from the Panther Creek tributaries. Wai and Mok further studied the effects of pH on the leaching of metals from sediment. Using a Panther Creek sample with low metals concentrations,[7] they reported that dropping the pH in the overlying water from 6.3.to 3.8 resulted in a 100 fold increase in copper and a 20 fold increase in cobalt released from the sediment. This range of pH shift has been reported for Blackbird Creek (Table 11). This suggests that during spring runoff conditions when pH drops and stream velocities increase, copper and cobalt re-mobilize from the sediments. Wai and Mok reported little release of arsenic under conditions likely to occur in the field. They concluded that the lower solubility of arsenic probably resulted from its strong partitioning behavior with iron oxides and that its mobility was controlled by this partitioning.

Thus leaching is an important mechanism for transporting toxic metals from contaminated sediments to the surface waters of the Blackbird area. In natural conditions, cloudbursts or high velocity, low pH spring runoff from snowmelt may resuspend the stream sediments into the surface water, causing cyclic releases of copper and cobalt.

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[6] Also published in a shorter journal version as Mok and Wai (1989).

[7] As 77, Co 100, and Cu 155 mg/kg; low levels relative to Panther Cr. conditions (compare to Figure 11).

V. Effects on Habitats and Species

Observed effects in the Panther Creek area that have been linked to contaminants released from Blackbird mine include lethality to salmonids in caged fish testing, reduced densities of salmonids below effluent pathways from the mine, bioaccumulation of contaminants in salmonids, severely stressed macroinvertebrate communities, and sediment toxicity. The patterns of observed adverse biological effects in Panther Creek drainage are concordant with the patterns of surface water and sediment copper, cobalt, and arsenic contamination from the Blackbird site. Additionally, other adverse effects that have been observed in numerous other studies with copper contamination include food chain effects, chronic toxicity, and behavior changes that may impede migration. Much less information on the effects of cobalt and arsenic is available. This is discussed separately at the end of this section.

Macroinvertebrates

Macroinvertebrate communities

Benthic macroinvertebrates[8] are an essential component of nutrient and energy cycling in aquatic ecosystems and are the primary food source for salmonids (Platts et al. 1983; Murphy and Meehan 1991). Field surveys of benthic macroinvertebrate communities are often used for ecological assessments of sediment and water quality monitoring. They have several advantages in ecological assessments: Indigenous benthic macroinvertebrates are ecologically important as an intermediate trophic level between microorganisms and fish. They are abundant in most streams; have either limited migration patterns or are sessile, which make them suitable for site specific impacts. Their life spans of several months to a few years allow them to be used as continuous indicators of sediment and water quality by integrating spatial and temporal variation, rather than a snapshot of conditions at one space in time (Platts et al. 1983; MacDonald et al. 1991; La Point and Fairchild 1992). They are closely associated with sediments, bioaccumulate contaminants, and have variation in sensitivities to metals pollution (Clements 1991). Additionally, these aquatic insects are the primary food source for rearing juvenile salmonids (Bjornn and Reiser 1991, Healey 1991).

Several studies of benthic macroinvertebrates have been conducted in Panther Creek between 1967 and 1992. Corley (1967) reported that mayflies (Ephemeropterans), stoneflies (Plecopterans) and caddisflies (Trichopterans) were abundant in Panther Creek stations above Blackbird Creek and absent or scarce below.

Subsequent studies that reported using (1) similar quantitative sample collection methods[9], (2) sampling from similar reaches in Panther Creek[10], and (3) using generally similar levels of taxonomic identification were compiled and evaluated for this report. Six quantitative surveys of benthic macroinvertebrates in Panther Creek were located and reviewed: April, August and November 1981 (Speyer 1982); May 1985 (Mangum 1985); and May and September 1992 (Smith 1993). Results are summarized using several measurements of

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[8] Aquatic insects and other invertebrates large enough to be seen with a naked eye that live principally in or on the bottom.

[9] Sampled riffles using quantitative Surber samplers with three replicate samples per location.

[10] Sample locations are grouped into reaches: upstream of Blackbird Creek (reference); below Blackbird Creek (between Blackbird and Deep creeks); above Big Deer Creek (between Napias and Big Deer creeks); below Big Deer Creek (between Big Deer and Beaver Creeks); and at the mouth of Panther Creek.

benthic community structure commonly associated with adverse effects of metals: abundance of mayflies, number of different mayfly species, overall numbers of species, and relative composition of dominant macroinvertebrate groups. Depressed abundance of mayflies has consistently been shown to be an effect of copper pollution (Sprague et al. 1965; Winner et al. 1980; Clements et al. 1988, 1992). Reductions in the number of different mayfly species has also been associated with metals contamination (Clements 1991; Clements et al. 1992. The relative composition of dominant macroinvertebrate groups in unpolluted Rocky Mountain streams is expected to have a fairly even distribution of chironomid midges, mayflies, stoneflies, and caddisflies with substantial representation of each the generally sensitive orders of mayflies, stoneflies, and caddisflies (Andrews and Minshall 1979; Murphy and Meehan 1991; Barbour et al. 1992). In copper-contaminated streams, chironomid midges consistently dominate the communities (Sprague et al. 1965; Winner et al. 1980; Clements et al. 1988, 1992). Overall reduced species richness has also been consistently reported in studies of effects of copper to benthic macroinvertebrates (Clements 1991).

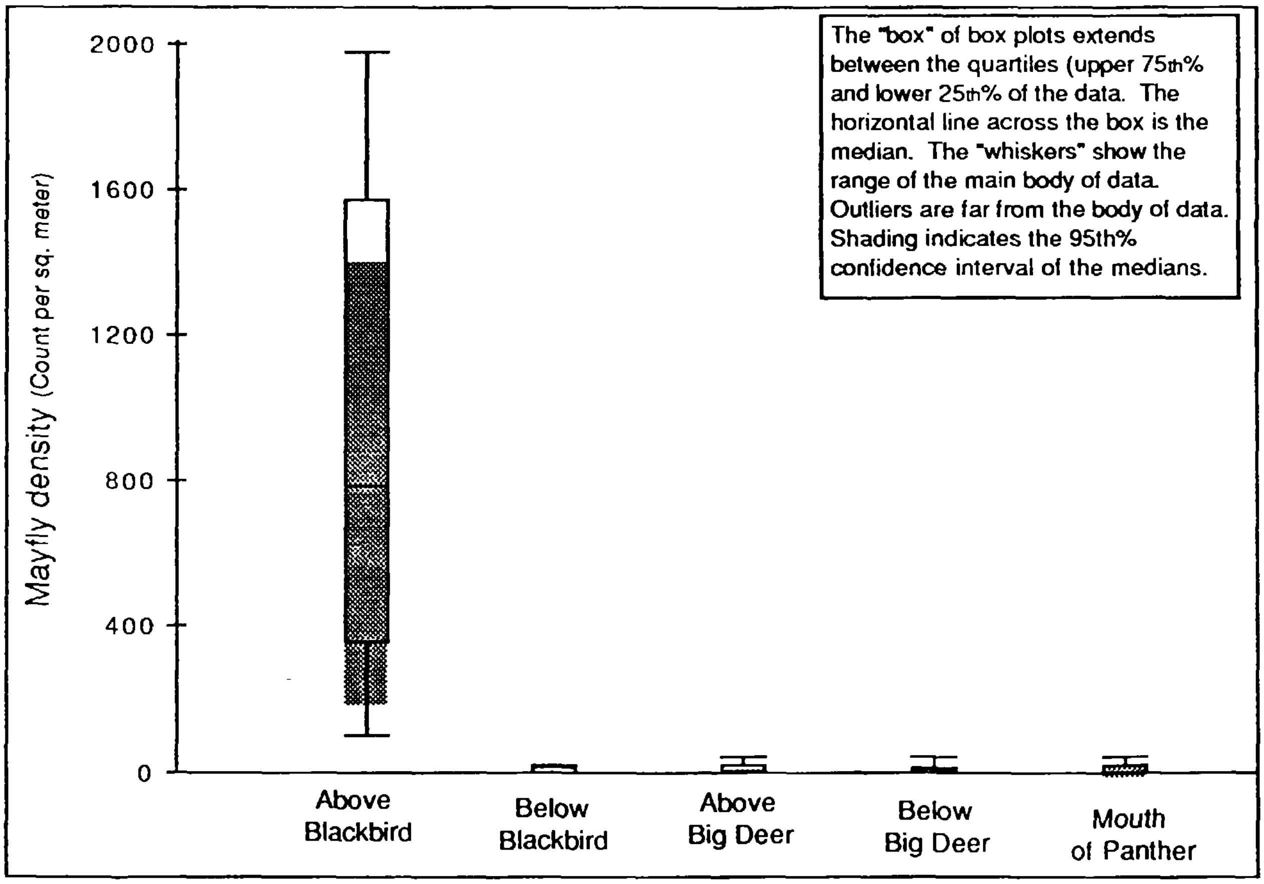
Macroinvertebrate communities in Panther Creek

Figure 13a compares overall densities of mayflies reported from all the Panther Creek surveys above and below the Blackbird mine effluents. Mayflies are abundant above Blackbird Creek and extremely scarce downstream (median density of about 800 mayflies per square meter upstream, zero below). Combining these surveys in this manner shows that, regardless of differences in who conducted the surveys, taxonomic identifications, seasons sampled, and differences in sample locations, mayflies have been largely eliminated downstream of the Blackbird Mine. The box plots in Figure 13a compare the median levels and overall ranges of the mayfly densities reported from Panther Creek surveys. The shaded confidence intervals are constructed so that if two gray boxes fail to overlap, the corresponding medians are statistically different at approximately the 5% significance level (Velleman 1988).

Similarly, Figure 13b shows an abrupt decrease in mayfly species diversity, with overall medians of 5 species reported upstream of Blackbird Creek, one species above Big Deer Creek, and zero species below Blackbird and Big Deer creeks, and at the mouth of Panther Creek. Additionally, when the overall species richness in Panther Creek was compared, all surveys showed a reduction in the overall number of benthic macroinvertebrate taxa at downstream stations compared to the upstream stations.

Figures 13c-h show that in the Panther Creek aquatic macroinvertebrate communities upstream of Blackbird Creek, the relative proportion of mayflies, stoneflies, caddisflies, and chironomids vary greatly with different seasons and surveys, yet all surveys have substantial representation of most groups. Below Blackbird, regardless of seasonal and sampling differences, all sites from all surveys show that chironomid midges dominate the communities with mayflies, stoneflies, and caddisflies rare or completely eliminated. At some times, abundances of all benthic macroinvertebrates were very low, indicating that little prey was available for fish at those times. Figure 14 shows the locations of the shifts in community composition and reduced invertebrate abundances in Panther Creek relative to the Blackbird Mine.

Figure 13a. Overall ranges of mayfly (Ephemeroptera) densities reported from six Panther Creek benthic macroinvertebrate surveys, 1981-1992 (data from Speyer 1982, Mangum 1985, and Smith 1993)



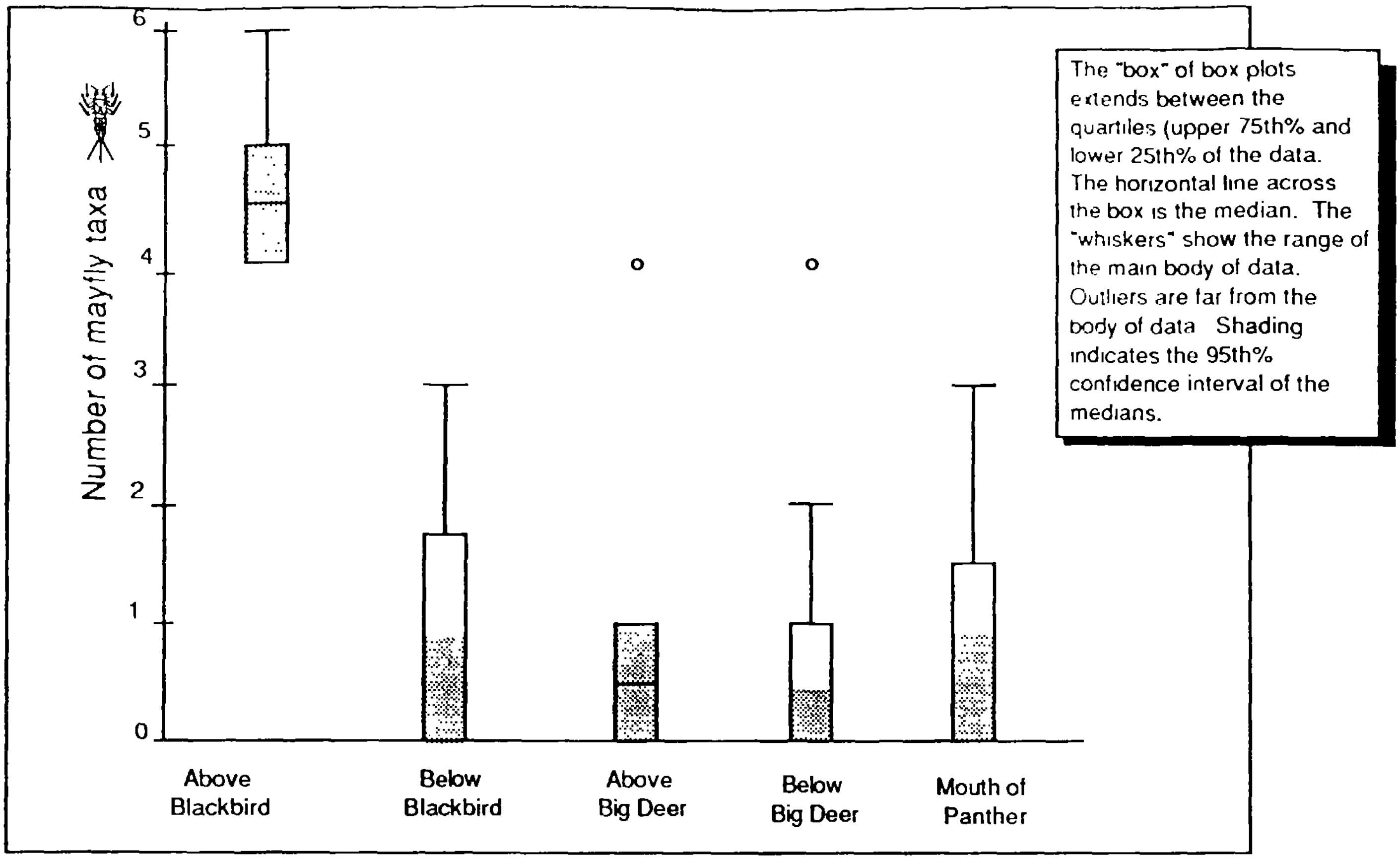


Figure 13b. Overall number of mayfly (Ephemeroptera) species reported from Panther Creek surveys, 1981-1992 (Speyer 1982; Mangum 1985; Smith 1993).

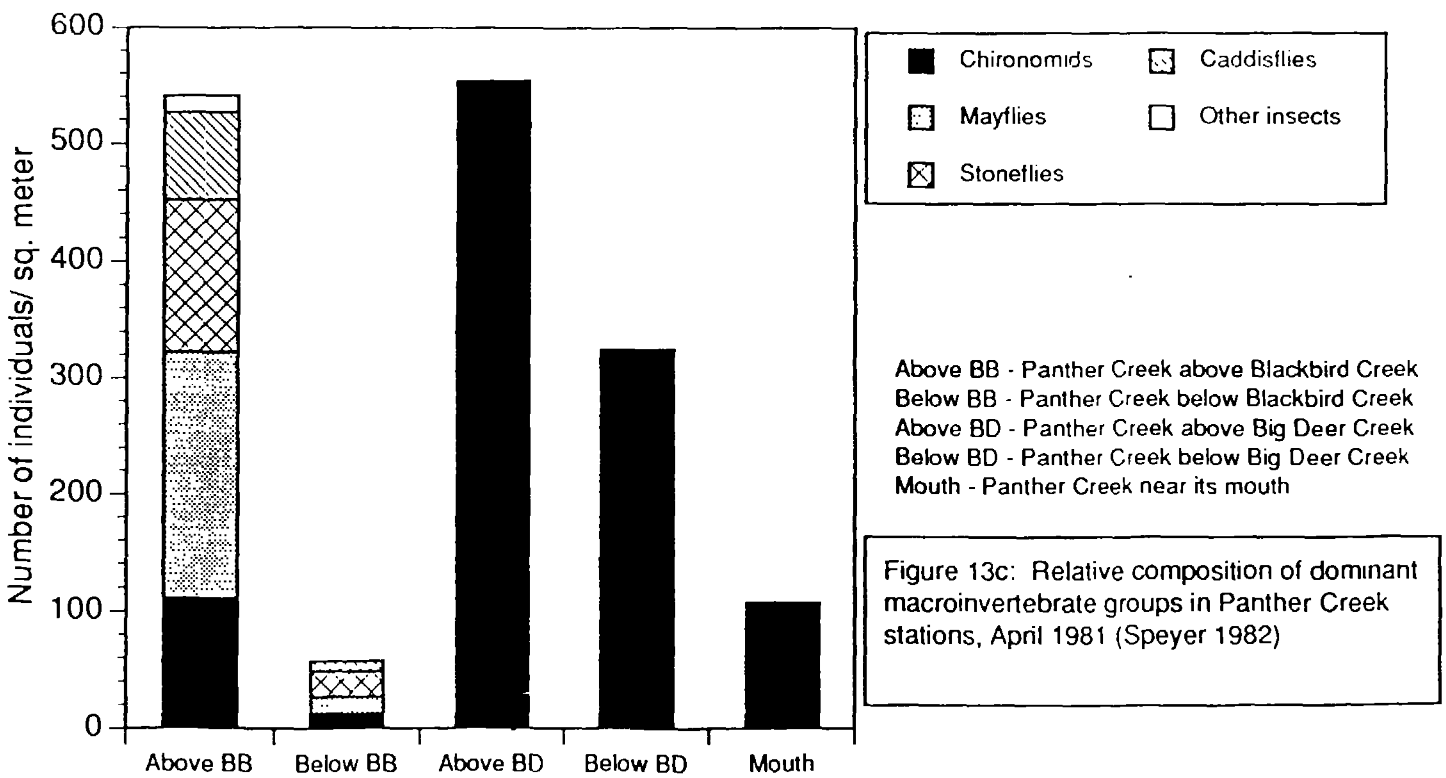


Figure 13c: Relative composition of dominant macroinvertebrate groups in Panther Creek stations, April 1981 (Speyer 1982)

Figure 13 (b-c). (b) Overall mayfly species richness from 1981- 1992 and
(c) composition of aquatic macroinvertebrate communities April 1981.

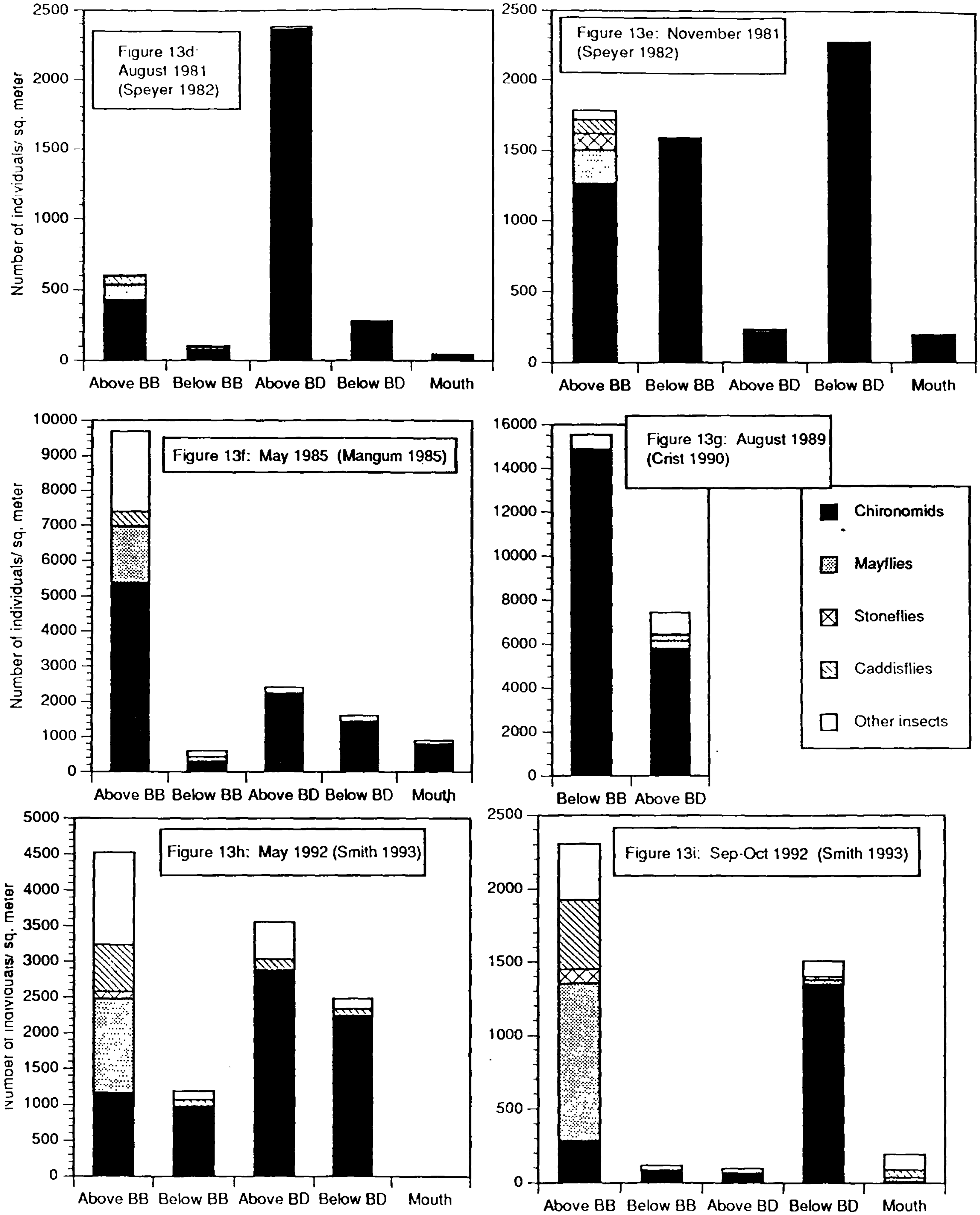


Figure 13 (d-i) Benthic macroinvertebrate community composition at Panther Creek stations sampled between 1981 and 1992.

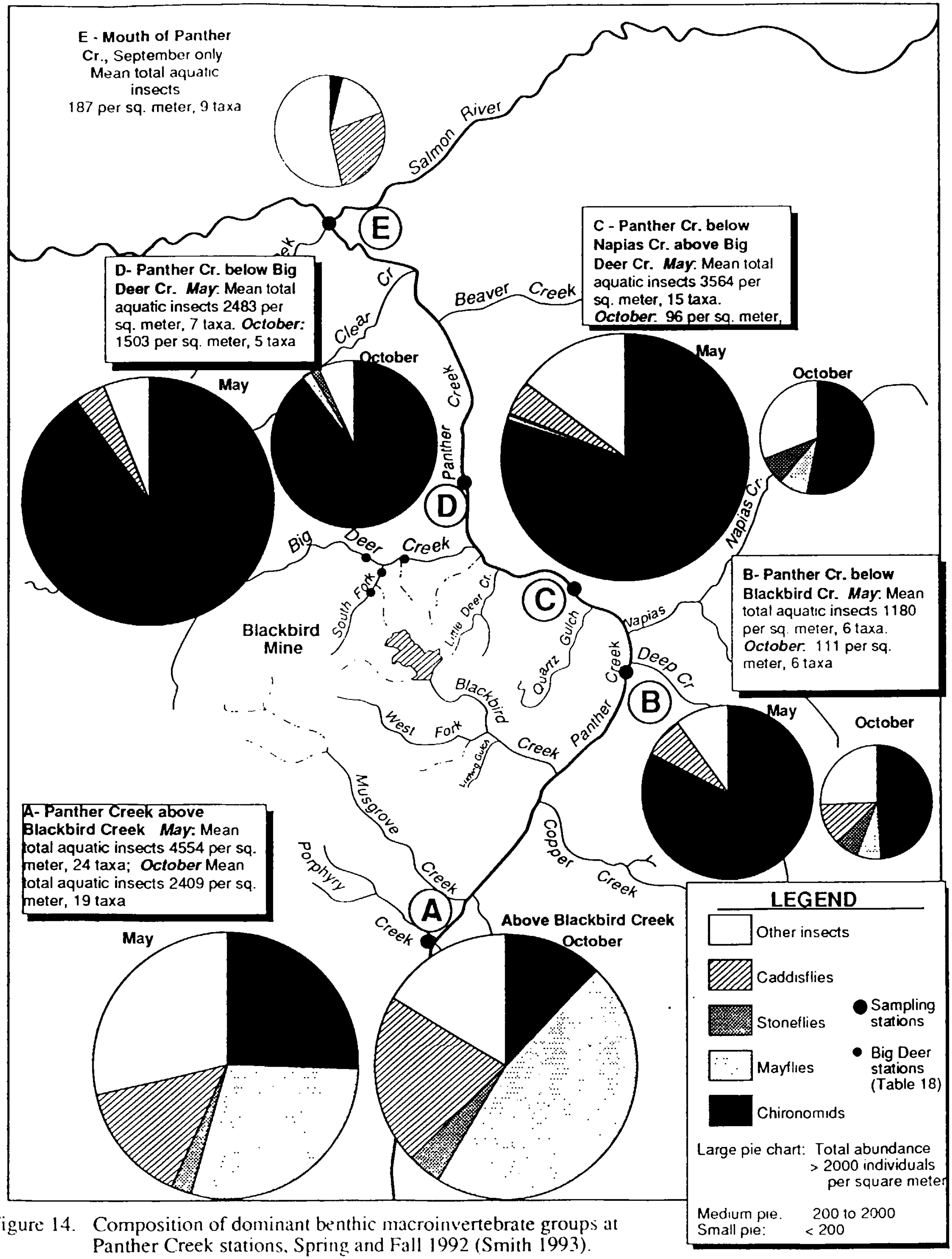


Figure 14. Composition of dominant benthic macroinvertebrate groups at Panther Creek stations, Spring and Fall 1992 (Smith 1993).

Despite seasonal and other differences between surveys, results have consistently shown abundant, diverse populations in Panther Creek above Blackbird runoff and below, depauperate populations dominated by metals pollution tolerant chironomid and simulidae midges. Species thought to be generally sensitive to metals contamination (mayflies, stoneflies, and caddisflies) had greatly diminished numbers or were completely eliminated downstream of Blackbird runoff. Nearly all life has been eliminated in Big Deer Creek below mine runoff (Table 18).

Table 18. 1992 Macroinvertebrate community distribution in Big Deer Creek (Cameron 1993). (Number per square meter; mean of three replicates). Sampling locations are shown on Figure 14.

	Upper Big Deer	Upper S. Fork Big Deer	Lower S. Fork Big Deer	Lower Big Deer
Chironomids	301	337	0	54
Mayflies (Ephemeropterans)	1044	1374	0	0
Stoneflies (Plecopterans)	316	854	0	4
Caddisflies (Trichopterans)	97	2042	0	7
Other	178	387	0	4
Total number of taxa	16	27	0	4
Total macroinvertebrates	1938	4995	0	68

Mangum (1985) concluded that the biomass of benthic macroinvertebrates in Panther Creek downstream of the mine was too low to support fisheries, with downstream stations averaging one-sixth of the upstream biomasses.

Literature on the effects of metals on aquatic insect communities

Published reports on effects of copper contamination on macroinvertebrate communities indicate that copper in water causes shifts in numbers of taxa, overall abundance, and community composition. Winner et al. (1980) compared water chemistry and macroinvertebrate community structure in two streams, one which received copper, chromium, and zinc effluent from a metal plating facility, the other which was experimentally dosed with copper for 30 months to a rather constant intermediate to low (maximum of 120 µg/l) copper dose. In the most heavily stressed sections of both streams, macroinvertebrates, other than tubificid worms and chironomids, were virtually eliminated from rock-rubble, riffle habitats. Midge larvae still comprised 75-86% of the insect communities at the least polluted stations. The correlation coefficient for percentage of chironomids in relation to copper concentrations was +0.93 ($P<0.01$), showing that the percent chironomids in benthic samples was highly correlated with copper concentration in impacted streams. Winner et al. concluded that the macroinvertebrate community gave a "predictable, graded response to heavy metals."

Clements et al. (1988, 1992) reported that exposure to copper significantly reduced both the total number of individuals and number of taxa during each season, with the greatest effects observed in summer. The relative abundance of Ephemeropterans (mayflies) decreased in treated streams during each season. The response of other aquatic insects, including Dipterans (true flies) and Plecopterans (stoneflies), varied between seasons, but these groups were generally less sensitive to copper exposure. The relative abundance of the dominant chironomids collected, Orthocladiini, increased in copper-treated streams in winter and spring. They also noted that the combination of measuring numbers of taxa, overall abundance, abundance and diversity of the sensitive mayflies, and the relative abundance of insensitive Orthoclad chironomids were a reliable indication of heavy metals contamination. Other metrics such as species diversity and functional groups varied more with season than with treatment. Clements et al. (1992) compared benthic communities at copper-zinc stressed stations with copper-dosed outdoor experimental streams. Sensitivity of 13 dominant taxa was measured in outdoor experimental streams by exposing organisms to (25 μg/l) copper for 10 days. Sensitivities, defined as proportional reduction in abundance, in treated streams relative to control ranged from 1.00 for some Ephemeroptera that were completely eliminated to -0.14 for taxa that increased in treated streams (Orthocladiini Chironomids). Clements et al. observed similar shifts in communities in the experimentally copper-dosed streams and in the copper-zinc stressed streams.

In contrast to mayflies, which have consistently been shown to be generally sensitive to metals, caddisflies vary greatly in their tolerance to metals. Nebeker et al. (1984) exposed *Clistornia*, a caddisfly typical of mountain streams in the Pacific Northwest, to copper in soft water in life cycle tests. Copper concentrations of ≥17 μg/l prevented completion of the life cycle, and the no-observed effect level for copper was 8 μg/l. These results indicate that some caddisflies are as sensitive as mayflies to copper contamination. However, Winner et al. (1980) found that some caddisflies are quite tolerant of copper pollution, even increasing in abundance.

These reports from field studies and controlled dosing experiments with copper are consistent with patterns in Panther Creek riffle-run macroinvertebrate community data. Marked reductions in mayfly density, mayfly species richness, overall species richness, and shifts in community composition occur which consistently match the patterns of water and sediment metals contamination originating from the Blackbird area (Figures 9 and 11).

Elimination of a broad base of aquatic macroinvertebrates, an overall reduction in numbers, would leave juvenile salmonids solely dependent on the pollution-tolerant chironomid midges. Seasonal collapses in chironomid abundances after hatchings would be likely to leave the juvenile salmonids with few alternative food sources. These data indicate that severe stress of the Panther Creek aquatic ecosystem with a depauperate food supply for juvenile salmonids is occurring. The observed aquatic community changes are similar to those reported in the literature resulting from metals pollution, and the spatial patterns in the communities match the sediment and water contamination patterns, all strongly implicating the Blackbird runoff as the cause.

Sediment Toxicity Testing

Adverse effects from sediment-sorbed metals contamination are limited by the metals' bioavailability to aquatic organisms. The extent and magnitude of contamination is not necessarily biologically significant if bioavailability is limited. Although sediments may contain relatively high concentrations of toxic heavy metals, this presence may not necessarily cause adverse effects to sediment-dwelling organisms. Bioavailability of contaminants is difficult to predict from chemical concentrations. The factors that determine metals' sorptive behavior, which affects bioavailability, are complex and poorly understood. The only means of measuring bioavailability is by measuring and determining a biological response, such as laboratory bioaccumulation or toxicity testing (Power and Chapman 1992). For this study, bioavailability was measured by exposing the amphipod *Hyalella azteca*, a freshwater crustacean, to sediments collected from Panther Creek in a 10-day acute toxicity test.

Hyalella azteca, Amphipoda, is a small freshwater crustacean that has become routinely used to screen the toxicity of contaminated sediments. It is common throughout lakes and streams in temperate and near-arctic North America. Several characteristics make it desirable as a test organism including a short life cycle (3-4 weeks), widespread and abundant distribution, ecological importance, and a wide tolerance of sediment grain size. *Hyalella azteca* is an epibenthic detritivore that will burrow in the top 1 cm of the sediment surface in search of food (ASTM 1991). Tests were performed according to ASTM (1991) protocols. The endpoints measured were 10 day survival and growth (measured as dry weight).

Testing Methods and Results

Sediment samples from Panther Creek above Blackbird Creek (station P1), below Blackbird Creek (station P2), and below Big Deer Creek (station P6), were tested for acute toxicity to the freshwater crustacean *Hyalella azteca* (Figure 10). Table 19 summarizes the results[11].

The samples from the stations below Blackbird Creek (P2) and Big Deer Creek (P6) had survival rates of 25% and 24%, respectively. 63% survived from the reference sediments collected from above Blackbird Creek (Figure 15). A one-way ANOVA was used to test the hypothesis that *Hyalella* survival was not significantly different when exposed to sediments from different locations in Panther Creek. Finding that survival rates were significantly different, Tukey's multiple comparison test was used to test for differences between locations (Zar 1984). Stations P2 and P6 both had significantly reduced survival ($p < 0.01$) from station P1. There was no difference between the survival rates between stations P2 and P6. Since stations P2 and P6 had reduced survival, statistical tests of the survivors for differences with respect to growth were not made. The square root arcsine data transformation to correct for proportionally distributed survival data did not affect the outcome. The Kruskall-Wallis test statistic and a Tukey-type, non-parametric multiple comparison test gave similar results.

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[11] Appendix A describes the methods and results of this study in more detail.

Table 19 Metals concentrations in sediments (dry weight) matched with *Hyalella* growth and survival.

Location (*Figure 10*)	Arsenic (mg/kg)	Cobalt (mg/kg)	Copper (mg/kg)	Iron (%)	TOC (%)	Mean survival (± S.D.)	Mean dry wt. (mg)
P1	9.5	4.2	8 3	1.21	0.88	5.8 ± 2.2	0.17 ± 0 08
	6.8	4 5	8.3	1.40	0.87	6 4 ± 2.9	0.21 ± 0 07
	10.7	3.8	12.9	1.36	3.78	6.8 ± 1.8	0 18 ± 0 03
Mean						*6.3 ± 2.2*	
P2	766	436	2280	7.45	2.70	3.4 ± 0.9	0.17 ± 0.07
	867	507	2700	8.56	2.71	4.0 ± 2.5	0.18 ± 0.07
	888	554	2890	8.41	2.85	0.0 ± 0.0	0.0 ± 0.0
Mean						*2.5 ± 2.3*	
P6	14	76	386	0.93	0.12	0.0 ± 0.0	0.0 ± 0 0
	27	91	750	1.09	0.28	1.4 ± 1.3	0.07 ± 0.07
	31	74	889	1.20	0.51	5.8 ± 0.8	0.08 ± 0.04
Mean						*2.4 ± 2.7*	
Negative Control	-	-	-	-	-	9.2 ± 1.3	0.15 ± 0.04

Concentrations in dry weight. For survival, a value of 10.0 represents 100% survival (10 amphipods per beaker). Sample mean is the mean of five replicates. Location mean is mean of the three samples.
TOC - Total organic carbon

Invertebrate toxicity related to sediment contamination.

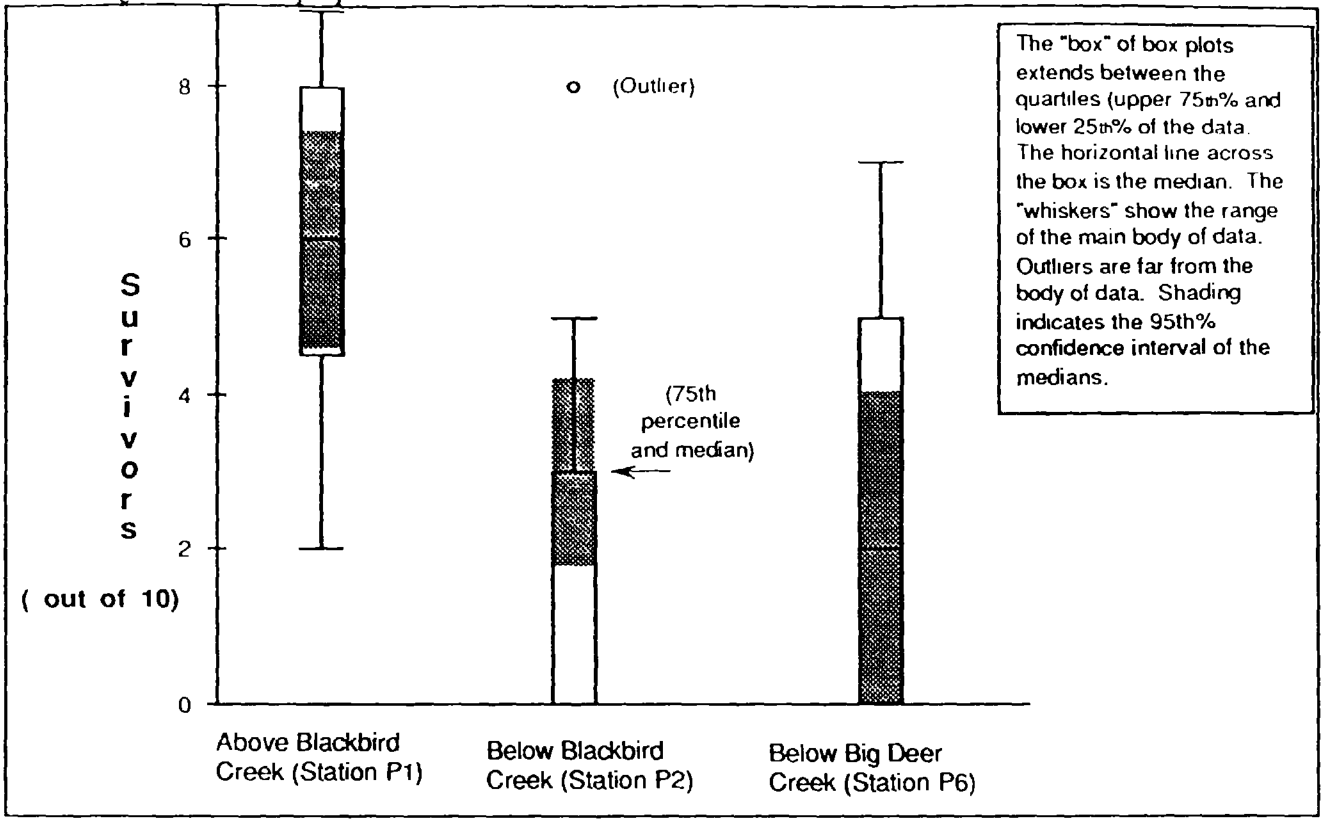
Hyalella survival was correlated to concentrations of copper, cobalt, and arsenic in Panther Creek sediments[12]. Metal-specific influences in the toxicity of Panther Creek sediments were evaluated from differences in correlations with metals in the sediments. Both linear (Pearson's r) and non-parametric (Spearman's rho) were calculated and were similar.

Table 20. Correlation between amphipod toxicity and metal concentrations in bed sediments collected from Panther Creek in October 1992. Correlation coefficients, r, and significance levels are shown. Significance levels : $^{*}p < 0.05$; $^{**}p < 0.01$.

	Copper	Cobalt	Arsenic	Molar sum (Cu + Co)	Molar sum (Cu+Co+As)
Bulk metal (dry weight)	-0.47	-0.48	-0.37	-0.47	-0.45
Normalized to organic carbon	-0.72*	-0.81**	-0.57	-0.83**	-0.78*

[12]Of the metals tested, only copper, cobalt, arsenic, and iron showed trends related to Blackbird Mine drainage.

Figure 15. Amphipod survival tests with Panther Creek sediments.



The results of this study show that high levels of metals are present in Panther Creek sediments below the tributaries draining the Blackbird mining district, and these sediments are acutely toxic to the epibenthic amphipod *Hyalella*. The correlations suggest that *Hyalella* toxicity is related to metals concentrations in the streambed sediments and that organic carbon is a factor significantly associated with metals bioavailability. Of the variables measured in these tests, the strength of association with toxicity was (Co>Cu>As). Normalizing metals concentrations to the amount of organic carbon in the sediment increased the strength of the correlations for each of these metals.

Coarse-grained, mid-channel sediments had much lower metals levels than the fine-grained sediments collected from near the stream banks (Table 16). These sediments also had very low levels of organic carbon, and were well oxygenated due to their more open interstices and location in faster water. Despite their lower bulk metals levels, these sediments were acutely toxic to the *Hyalella*, with no survival in any replicates. This increased mortality can not be entirely accounted for by a possible *Hyalella* intolerance to coarse-grained sediments, since some of the reference sediments also had a high proportion of coarse-grained sediments. These results suggest that, in high gradient, coarse-grained systems that are contaminated with metals, targeting the fine-grained sediments for bioassessment studies may underestimate the exposure of aquatic life to metals contamination.

Various physical and chemical factors have been reported to affect sediment-trace element chemistry and bioavailability, including grain size, organometallic bonding, complexing with iron and manganese oxides, and sulfides (Horowitz 1991). Iron normalization has been reported to affect correlations between sediment lead and arsenic concentrations and bioaccumulation (Cain et al. 1992). Normalizing sediment metal concentrations to iron

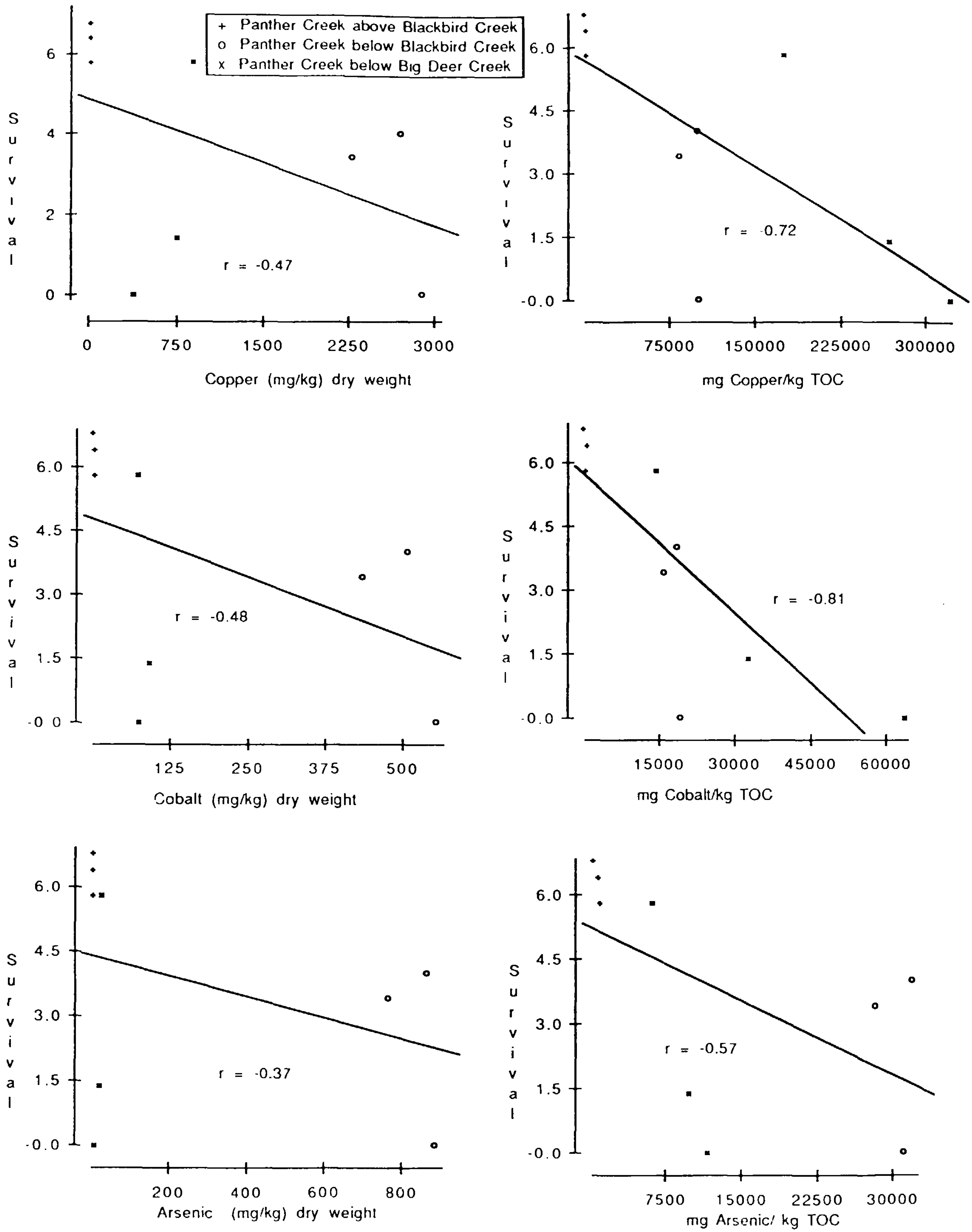


Figure 16. *Hyalella* survival correlated with copper, cobalt, and arsenic concentrations in Panther Cr. sediments. For survival, a value of 10.0 equals 100% survival; value is the mean of five replicates.

concentrations in the sediments (e.g. Horowitz 1991) did not significantly change the correlations between sediment metal concentrations and survival. However, these factors affecting bioavailability of metals are complex and poorly understood. Some studies support porewater metal concentrations as the major controlling factor for copper toxicity and some other sediment-sorbed contaminants. Porewater concentrations appear to be determined by binding phases including sulfides and organic carbon, which regulate partitioning of copper between sediments and porewater (Ankley et al. 1993; Kemble et al. 1993). However, these authors reported that while porewater concentrations of copper and other metals predicted toxicity, porewater concentrations could not be predicted by normalizing to conventionals such as acid volatile sulfides[13]. Vandersypen (1993) reported that porewater copper concentrations did not correlate well with *Hyalella* toxicity and suggested that binding with organics also reduced copper toxicity. Deaver et al. (1993) reported that *Hyalella* response to copper exposure in sediments was related to organic carbon content of the sediments. Increased organic matter content of sediment corresponded with decreased toxicity. A higher organic carbon content sediment (4%) was amended with five times more copper sulfate than a lower organic carbon sediment (0.9%) to achieve the same level of toxicity. Similarly, increased organic carbon content appears to have been related to reduced metals toxicity in the Panther Creek samples tested.

The *H. azteca* ten day-acute survival test has shown low sensitivity in comparative bioassays with a 29-day *Hyalella* test or some other test organisms in metal-contaminated sediments (Ingersoll and Nelson 1990; Kemble et al. 1993; Cairns et al. 1984). Finding significant mortality in a relatively insensitive screening toxicity test suggests that metals in Panther Creek sediments, particularly copper and cobalt, are readily bioavailable, and are toxic to macroinvertebrates. These contaminated sediments are likely to have significant adverse effects on the Panther Creek aquatic ecosystem.

Comparison to sediment toxicity in the literature

Sediment-sorbed copper and arsenic concentrations that have been associated with biological effects in other freshwater studies are compared to the Panther Creek concentrations from this study in Tables 21 and 22. These effects concentrations were developed from several approaches: (1) spiked sediment bioassays where a known quantity of the test chemical is mixed into the sediment, allowed to equilibrate, and tested for toxicity; (2) co-occurrence analyses of benthic community structures with chemical concentrations; (3) an apparent effects threshold (AET), defined as the maximum concentration of a chemical that did not reduce the survival of the particular indicator (e.g. amphipod survival, benthic communities)[14]; and (4) the sediment quality triad, an effects based approach that incorporates measures of sediment chemistry, sediment toxicity, and benthic community structure and has been used to develop numerical criteria. Also, an extensive review of effects associated with sediment-sorbed contaminants was developed to rank stations sampled in the NOAA National Status and Trends (NS&T) program (Long 1992; Long and MacDonald 1992). All listed effects concentrations are from freshwater studies with the exception of the NOAA NS&T review, which included marine and estuarine studies. While these studies, with the exception of the spiked sediment bioassays, do not establish a cause and effect relationship between the contaminant levels and effects, they do provide a useful benchmark of ranges of sediment-sorbed contaminants associated with adverse effects.

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[13] The sulfides that are liberated from a sediment sample to which acid has been added at room temperature, generally the iron monosulfides but not pyrite or other sulfides.

[14] Also called a no effect concentration (NEC) by some authors

Using the NS&T ranking approach, a total of 31 freshwater studies reporting biological effects associated with sediment-sorbed copper were located and ranked. The ranking resulted in a freshwater copper effects range-median (ER-M) of 262 mg/kg (dry weight). Much less information on toxicity of sediment-sorbed arsenic in freshwater and no information on toxicity of sediment-sorbed cobalt was located for this review. Persaud et al. (1993) gives a cobalt sediment quality guideline of 16 and 50 mg/kg, borrowing the values from Ontario's limit for unrestricted disposal of dredged material in lakes. No information on the derivation for these values was available at writing. Most of the concentrations in sediment samples collected from Panther Creek downstream of Blackbird Mine exceed the copper median effects concentration of 287 mg/kg, and the highest effects-concentrations reported for arsenic (70 mg/kg) and cobalt 50 (mg/kg). Concentrations in all sediment samples from Panther Creek upstream of the Blackbird Mine were near or below the lowest concentration associated with effects for all three metals (Tables 21 and 22).

Table 21. Concentrations of arsenic in sediments associated with effects (mg/kg dry weight)

Arsenic	Biological effects
6	Lowest effects level in co-occurrence analyses with benthic communities. Correlated with loss/depression of the most sensitive freshwater benthic species (lower fifth percentile of affected species) (a)
8	NS&T Effects Range-Low (ER-L), the concentration above which adverse effects may begin or are predicted among sensitive species (effects in the lower tenth percentile of studies) (b)
11	Highest level found in Panther Creek sediments above Blackbird mine influences (c)
14	Lowest level found in Panther Creek sediments below Blackbird mine influences (c)
33	Severe effects level. Correlated with loss/depression of the most freshwater benthic species (90th percentile of affected species) (a)
70	NS&T Effects Range-Median (ER-M), the concentration above which effects are frequently or always observed or predicted among most species (b)
888	Highest level found in Panther Creek sediments below Blackbird mine influences (c)

NOEC - No observed effects concentration; LOEC - Lowest observed effects concentration.

References
(a) Jaagumagi 1992; (b) Long and MacDonald 1992; (c) This study.

Table 22. Selected concentrations of copper in sediments associated with effects (in mg copper/kg sediment dry weight)

Copper	Biological effects
16	Highest level found in Panther Creek sediments above Blackbird mine influences (a)
16	Lowest effects level in co-occurrence analyses with benthic communities. Correlated with loss/depression of the most sensitive freshwater benthic species (lowest fifth percentile of affected species) (b)
20	Highly toxic to amphipod *Hyalella azteca*, Waukegan Harbor, Illinois (c)
26	LOEC (reduced growth) in 48-hour spiked sediment bioassays with the freshwater clam *Corbicula fluminea* larvae (d)
34	NS&T Effects Range-Low (ER-L), the concentration above which adverse effects may begin or are predicted among sensitive species (effects in the lower tenth percentile of studies) (e)
36	LC50 for 14-day *Ceriodaphnia dubia* spiked sediment bioassays (f)
37	Benthic macroinvertebrate AET (depression of mayflies and amphipods), Keweenaw Waterway, Michigan (g)
40	LC50 for 14-day *Daphnia magna* spiked sediment bioassays, (f)
65	Lowest level found in any Panther Creek sediment samples below Blackbird mine drainage (a)
86	LOEC (reduced survival) in 10-day spiked sediment bioassays with *Corbicula fluminea* juveniles (d)
110	Severe effects level. Correlated with loss/depression of the most freshwater benthic species (90th percentile of affected species) (b)
162	LC50 for 14 day *Pimepales promelas* (fathead minnow) spiked sediment bioassays (f)
170	LC50 for 48 hr. *D. magna* spiked sediment bioassays (f)
207	NOEC (survival) in 10 d spiked sediment bioassays with amphipod *H. azteca* (d)
216	Reduced growth for 14-day *Chironomus tentans* spiked sediment bioassays (f)
262	LC50 for 10d Hyalella spiked sediment bioassays (f) ***Median of the effects ranges (ER-M) reported from 31 freshwater copper-sorbed sediment studies.***
270	NS&T Effects Range-Median (ER-M), the concentration above which effects are frequently or always observed or predicted among most species (e)

Table 22 (continued). Selected concentrations of copper in sediments associated with effects (in mg copper/kg sediment dry weight)

Copper	Biological Effects
287	Sediment quality triad, Clark Fork River, Montana (h)
325	Amphipod *Hyalella azteca* chronic (reduced growth) AET, Clark Fork River, Montana (h)
347	LOEC (reduced survival) in 10 d spiked sediment bioassays with amphipod H. *azteca* (d)
386	Lowest concentration in sediments that were toxic to the amphipod *H. azteca*, Panther Creek, Idaho in 10-day tests. All sediments with higher concentrations also showed toxicity (a)
480	*Daphnia magna* 48-hour AET, Keweenaw Waterway, Michigan (g)
681	LC50 for 48-hour *Daphnia magna* spiked sediment bioassays, Soap Creek Pond, Oregon (i)
857	LC50 for 10-day *Chironomus tentans* spiked sediment bioassays, Soap Creek Pond, Oregon(i)
878	Non toxic to rainbow trout sac-fry in 21-day exposure (h)
890	Acutely toxic in 10-day tests to the mayfly *Hexagenia* and *Hyalella*, Steilacoom Lake, WA (j)
918	LC50 for 10-day *Hyalella* spiked sediment bioassays, Silver Creek, Washington (k)
1026	LC50 for 14-day *Chironomus tentans* spiked sediment bioassays (f)
1078	LC50 for 10-day *Hyalella* spiked sediment bioassays, Soap Creek Pond, Oregon (i)
2296	LC50 for 10-day *C. tentans* spiked sediment bioassays, Tualatin River, Oregon (i)
2930	Highest level found in Panther Creek sediments below Blackbird mine influences (a)

NOEC - No observed effects concentration; LOEC - Lowest observed effects concentration.

References
(a) This study; (b) Jaagumagi 1992; (c) Ingersoll and Nelson 1990; (d) Lynde et al. 1993; (e) Long and MacDonald 1992; (f) Suedel et al. 1994; (g) Malueng et al. 1984; (h) Kemble et al. 1993; (i) Cairns et al. 1984; (j) Bennett and Cubbage 1992; (k) Vandersypen 1993

Fish

Toxicity testing

The Idaho Department of Fish and Game (IDFG) has conducted in situ (caged fish) studies in Panther Creek using juvenile chinook salmon or rainbow trout. Fish tested in Panther Creek just below the confluences of Blackbird and Big Deer creeks have shown significant mortality relative to the upstream controls in some of the tests. All fish tested in Blackbird and Big Deer creeks at their mouths with Panther Creek died. These tests are compiled and summarized in Table 23.

Table 23. Panther Creek in situ (caged fish) toxicity testing 1967-1985.

	Percent survival					
Test	Panther Cr above Blackbird	Panther Cr below Blackbird	Panther Cr above Big Deer	Panther Cr below Big Deer	Mouth of Blackbird	Mouth of Big Deer
Aug. 1985 (chinook, 216h exposure)[a]	100	100	95	5	0	0
May 1985 (chinook, 216h exposure)[b]	100	95	100	85	0	0
May 1984 (chinook, 144h exposure)[c]	100	60	100	f	NT	NT
Apr. 1977 (rainbow trout, 120h exposure)[a]	100	0	100	80	NT	NT
Jul.- Oct. 1976 (rainbow trout, 120h exposure)[ae]	100	70	100	0	NT	NT
Sep. 1975 (rainbow trout, 120h exposure)[a]	100	5	NT	NT	NT	NT
Aug. 1975 (rainbow trout, 120h exposure)[a]	100	95	NT	NT	NT	NT
Jul. 1972 (rainbow trout, 240h exposure)[a]	98	14	60	0	NT	NT
Summer 1967 (rainbow trout, 72h exposure)[d]	96	0	NT	NT	NT	NT

[a] Reiser (1986); [b]Torf 1985; [c]Petrosky and Holubetz (1985), [d]Corley, (1967)

[e] Panther above and below Blackbird tested in July, above and below Big Deer in October

[f] Cage disappeared during test

n= 20 for all tests except for n=50 in 1972 and 1967

NT - Not tested

Most of the caged fish bioassays did not have any corresponding water chemistry measurements reported[15]. However, assuming that the 1985-1986 range of <10 to 160 µg/l

[15]The only data located were from the May 1985 tests. Reported total copper concentrations (µg/l) above Blackbird were <20; below Blackbird 60-70; above Big Deer 20; below Big Deer 40-60, and 2 km above the mouth of Panther, 100-110 µg/l (75% survival). Two grab samples each cage (Torf 1985).

dissolved copper concentrations in lower Panther Creek is typical (Table 15), some of the caged fish tests had less toxicity than would be expected from the literature on acutely lethal levels of copper to salmonids. Other investigators using in situ caged fish studies in copper contaminated waters have also reported somewhat better survival in in situ caged fish than in laboratory test chambers for similar total concentrations of copper (McKean et al. 1991, Phillips and Spoon 1990). There appear to have been two main factors reported from these studies that may have been relevant to the Panther Creek tests: (1) size of the fish used, and (2) the form of the copper in the streams and inorganic and organic complexing agents that may be present at some times.

<u>Size of fish used</u>. Several investigators reported that larger juvenile and adult salmonids were more resistant to copper lethality than the smaller juveniles tested[16] In these studies copper resistance with the larger fish was up to a factor of nine times greater than that for the smallest "swim-up" larvae (or alevins) which appear to be most vulnerable life stage. In order to avoid escape from the cages, some of the caged fish field studies used fingerling or adult fish instead of the more vulnerable, smaller, swim-up alevins and fry. Very fine mesh that could retain the smallest fish larvae may not allow adequate flow through the cage or may tear out and may not be practical in streams with much flow. Of the toxicity tests reviewed, those that used the smallest, most vulnerable larvae were conducted in laboratory aquaria, not in cages. While no standard terminology for salmonid life stages has been adopted, fry are often described as being between 20-50 mm fork length and fingerlings as between 80-150 mm fork length. The Panther Creek tests described using relatively hardy larger juvenile fish (fingerlings). Finlayson and Wilson (1989) reported that lethality tests with streamside flow-through troughs, rather than in situ cages, using relatively small chinook fry (20-60 mm fork length) gave similar results to laboratory toxicity testing. Phillips and Spoon (1990) conducted in situ caged fish tests initially using both fry and fingerlings. The larger fingerlings were consistently hardier than the fry leading them to drop the fingerlings from the tests. In situ caged fish toxicity testing has been a recommended approach for determining acute lethality from contaminated water (USFWS and UW 1987), but may be a reliable indicator of only the most severe impacts due to logistical limitations carrying out the tests with the smallest, most vulnerable life stages of fish.

<u>Forms and complexing</u> Copper toxicity to salmonids is generally accepted to be caused by ionic copper (the aquo ion $[Cu(H_2O_6]^{2+}$, usually referred to as Cu^{2+}) and ionized hydroxides ($CuOH^+$ $Cu_2(OH)_2^0$ and $Cu_2(OH)_2^{2-}$). Few chemical analyses of water in toxicity testing actually measure species of copper present but report "dissolved" (filtered) copper or "total" (unfiltered), where the so-called dissolved values are assumed to represent the ionic, toxic form of copper. While this may hold as a generality that predictive ability is improved by using filtered samples, some less toxic copper complexes such as copper hydroxides and carbonates sorbed to suspended particulates, and organic ligands may pass through a 0.45 µm filter membranes and be included in the "dissolved" value (Sprague 1985). An examination of the Beltman et al. (1993) filtered and un-filtered analyses of water samples from Blackbird, Big Deer, and Panther creeks showed that, under the low flow conditions sampled (September 1992), a higher proportion of total copper from the Big Deer Creek and Panther Creek below Big Deer was in the dissolved state than were the samples from Blackbird Creek and Panther Creek below Blackbird. While not conclusive, these data may indicate that more sorption to particulates in the Blackbird drainage than Big Deer occurs during low flow conditions. Also, the in situ caged fish tests and salmonid population monitoring (following section) showed greater mortality and fewer fish in Panther Creek

[16] Laurén and McDonald 1986; Chakoumakos et al. 1979; Finlayson and Wilson 1989; Howarth and Sprague 1978; Chapman 1978, 1994; and Chapman and Stevens 1978.

below Big Deer Creek than below Blackbird Creek, even though higher total copper levels have been reported below Blackbird Creek (Figure 9).

McKean et al. (1991) showed with both caged fish studies and tissue analysis of native fish that rainbow trout and coho salmon (*Oncorhynchus kisutch*) exposed to copper concentrations produced metallothionein at levels corresponding with exposure. As exposure increased, increased sub-lethal effects were observed (gill structure damage and increased metallothionein levels). Even when gill histology showed severe damage, all of the salmon survived a 96-hour LC_{50} acute lethality test[17]. McKean et al. (1991) cautioned against placing high reliance on acute lethality testing of rainbow trout as an indicator of safe copper concentrations. They concluded that the tests may miss all but the grossest effects; even with high survival during the length of the test, the fish may incur significant impairment.

[17] LC50 is the median lethal concentration, i.e. the concentration of material in water to which test organisms are exposed which is estimated to be lethal to 50% of the test organisms.

Bioaccumulation of metals in Panther Creek salmonids

Residues of cobalt, copper, iron, zinc, and lead in muscle tissue of salmonids were measured in 1985 to determine potential human health hazards from eating fish from Panther Creek below the Blackbird Mine. Fish muscle tissues were composited from three reaches for analysis: Upper Panther (above Blackbird Cr), middle Panther (below Blackbird but above Big Deer Creek) and lower Panther (below Big Deer Creek). Results show increased residues of cobalt and copper in fish collected in the reaches below Blackbird Creek. (Figure 17).

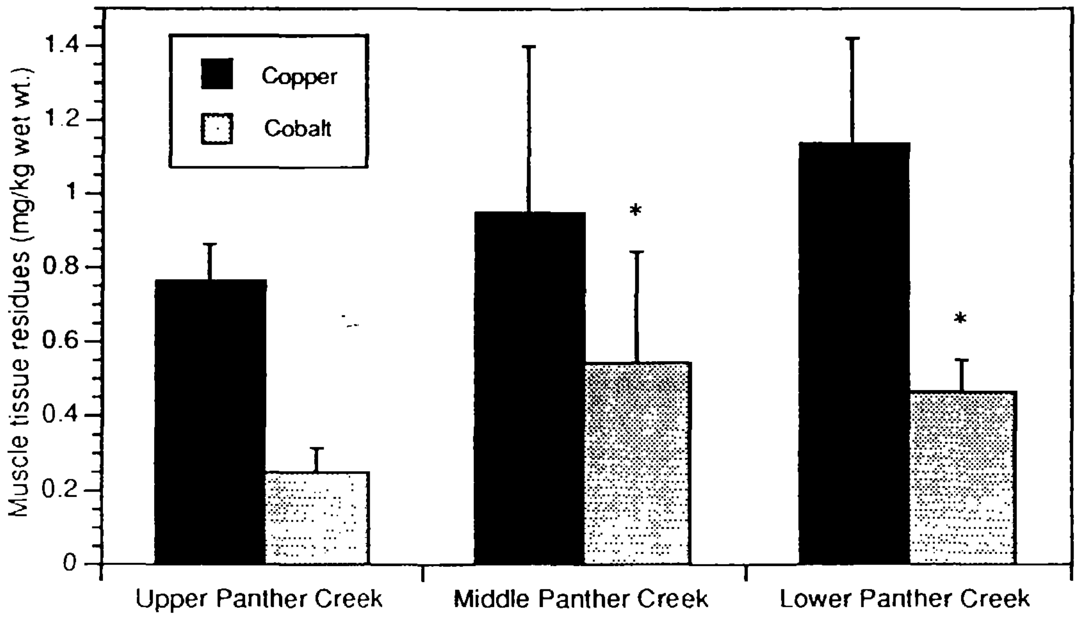


Figure 17. Mean metals residues (mg/kg wet weight) in muscle tissue from Panther Creek fish. "*" indicates significantly higher than upper Panther Creek at ($p < 0.05$) by Kruskall-Wallis non-parametric multiple comparison test. Error bars show one standard deviation (Reiser 1986)

Copper accumulates most strongly in fish livers, and to a lesser extent, kidneys and gills, but muscle tissues do not appear to be involved in copper accumulation. Copper levels in muscle tissue of laboratory exposed fish have been similar to those of controls (Sorensen 1991). Panther Creek data show a similar pattern; copper residues below Blackbird are not as elevated as highly as are cobalt residues. The fact that increased copper residues were found at all in a part of the fish were copper does not readily accumulate suggests significant exposure to copper for these fish.

Population Studies

Salmonid population surveys

Blackbird mine effluents are associated with depressed salmonid populations in middle and lower Panther Creek. The first fish population surveys in Panther Creek were reported by Corley (1967). Corley electrofished four 140 m stream sections of Panther Creek, two above Blackbird Creek, one just below Blackbird Creek, and one between Deep and Napias creeks. He found no fish below Blackbird Creek, nine fish (two species) at the lowest site, compared with 24 and 66 fish (three and six species), at the upstream sites. Further electrofishing in July 1980, found rainbow trout, brook trout, mountain whitefish, and sculpins upstream from Blackbird Creek. No fish of any species were found in the section between Blackbird and Big Deer Creeks. Rainbow trout, which are stocked annually, were the only species collected from lower Panther Creek just above its mixing zone with the Salmon River (Sgro et al. 1981).

Since 1984, the Idaho Department of Fish and Game has annually monitored the abundances of juvenile steelhead and chinook in rearing habitats in 166 stream sections in the Salmon and Clearwater drainages, including five in Panther Creek (Rich et al. 1992). Fish counts have been made by direct observation by snorkeling with observers moving slowly upstream spaced to observe the entire stream width, yet not to herd fish upstream. Snorkel counts have been used since streams in the study area are clear, have low conductivity, and the unreliability of electrofishing in large streams. Comparisons of snorkel counts and electrofishing/removal estimates in these types of streams support using snorkeling surveys under these conditions (Petrosky and Holubetz 1987; Hankin and Reeves 1988; Hillman et al. 1992).

The five reaches are each about 100 m long and include at least one riffle-pool complex (Petrosky and Holubetz 1985). Direct upstream-downstream comparisons are difficult since the established monitoring sections are not normalized for habitat variables. Morphological and physical habitat features, including valley and channel type, substrate, and microhabitat features (e.g. fast/slow water, pool size, overhanging vegetation, bank type) that provide feeding and refugia for juvenile fish also affect abundances. For example, Hankin and Reeves (1988) reported that in undisturbed streams, higher parr abundances are usually found in lower, larger reaches of streams than in their upper reaches. The reverse has been reported for Panther Creek. Results of the annual IDFG population surveys are compiled in the following section:

Chinook salmon

Chinook salmon have been rare throughout Panther Creek from 1984-1992 (Table 24). All occurrences appear to be related to stocking. Highest abundances were recorded in upper Panther Creek in 1987 following the release of over 3000 adult chinook near the mouth of Panther Creek in summer 1986 (Table 3). No juvenile chinook were observed in the section below Big Deer Creek even though this section's "Type A" pool-cascade-boulder-habitat provides an abundance of juvenile rearing habitat (Table 1; Reiser 1986, p. B2-16). Sightings of spawning adults and juveniles in vicinity of Clear Creek are presumably the progeny of this release or from eggs planted near Clear Creek in 1987.

Table 24. Annual densities of juvenile chinook in established monitoring sections, Panther Creek. (Petrosky and Holubetz 1985;Petrosky and Everson 1988; IDFG 1992).

Chinook salmon - Age 0 and 1 (fish/100 m^2)					
Survey	Above Blackbird, section #1 (by Cabin Cr)	Above Blackbird, section #2 (by Moyer Cr)	Below Blackbird Cr	Below Big Deer Cr	Below Clear Cr
1984	0	0	0.1	0	0
1985	-	-	0	0	0
1986	-	0	0	0	0
1987	0	56.2	3.2	0	0.1
1988	0 3	12.5	0.5	0	0.3
1989	0.0	0	0	0	0
1990	0	0	0	0	0.1
1991	0	0	0	0	0.1
1992	0	0	0	0	0.1

Rainbow and steelhead trout

Rainbow/steelhead trout[18] densities are highly variable and appear to be maintained above sustainable levels by stocking. The IDFG annually stocks Panther Creek with catchable size rainbow trout for recreational fishing. This "put-and-take" fishery depends on hatchery reared fish with no expectation of long term survival (IDFG 1993). Despite this confounding stocking, IDFG population surveys for rainbow/steelhead show a trend of somewhat lower densities of rainbow/steelhead trout downstream of Blackbird Creek relative to upstream areas. Extremely low densities of fish have been observed in the section just below Big Deer Creek, although the "Type A" pool-cascade-boulder habitat should support high densities of juvenile trout (Table 1). Numbers of rainbow/steelhead increased somewhat near the mouth of Panther Creek below Clear Creek. Figure 18a shows the average densities from 1984-1992. The box plots in Figure 18c illustrate the high variability of some rainbow/steelhead counts (except for the consistently depressed section below Big Deer Creek) suggesting that some surveys counted recently released fish.

Resident salmonids

Brook trout, bull trout, cutthroat trout, and mountain whitefish populations in Panther Creek are not supplemented by ongoing stocking. Without the confounding influence of stocking, resident salmonid counts show a consistent pattern of depressed numbers in Panther Creek in relation to Blackbird mine effluents (Figure 18b). Compared to upstream sections, resident salmonids are less abundant at all locations surveyed below Blackbird Creek and extremely scarce in the section below Big Deer Creek, despite the suitable pool-cascade-boulder habitat there.

[18]Juvenile rainbow and steelhead trout are indistinguishable in visual surveys and are recorded as rainbow/steelhead.

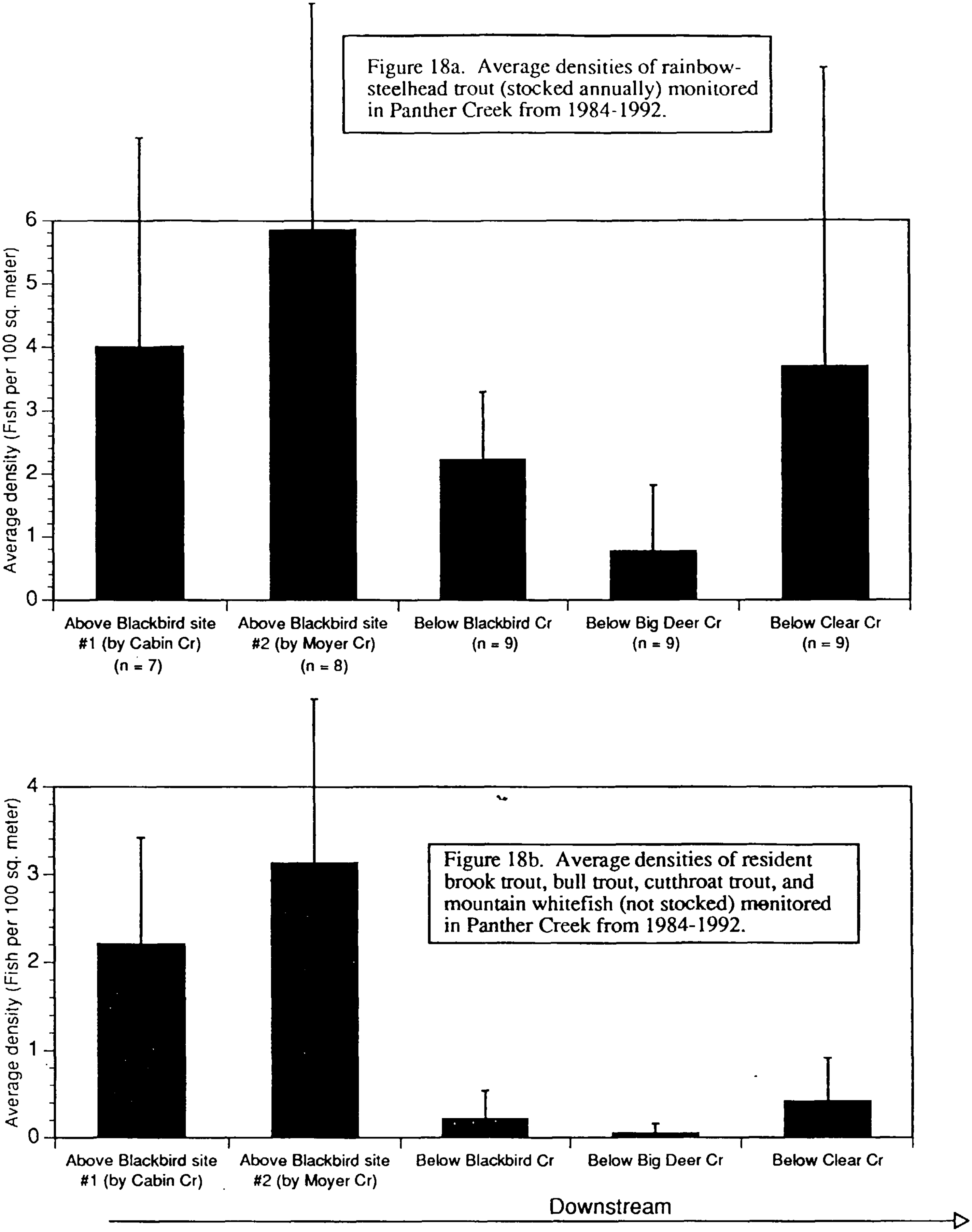
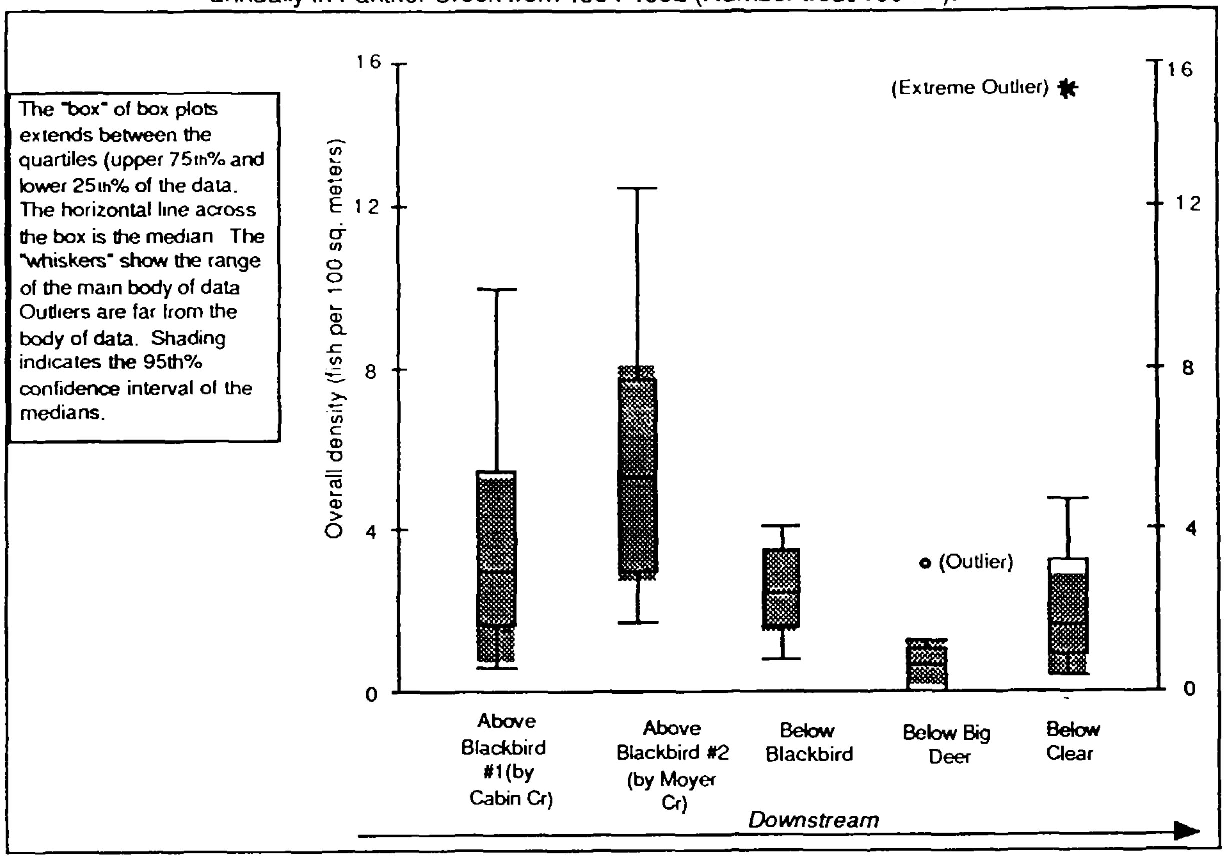


Figure 18a. Average densities of rainbow-steelhead trout (stocked annually) monitored in Panther Creek from 1984-1992.

Figure 18b. Average densities of resident brook trout, bull trout, cutthroat trout, and mountain whitefish (not stocked) monitored in Panther Creek from 1984-1992.

Figure 18. Average densities of rainbow-steelhead trout (stocked annually) and resident salmonids (not stocked) monitored annually in Panther Creek from 1984-1992. Error bars show standard deviation.

Figure 18c Comparisons of overall densities of juvenile steelhead-rainbow trout monitored annually in Panther Creek from 1984-1992 (Number trout/100 m²).



Big Deer Creek.

Although Big Deer Creek has excellent physical habitat for trout rearing and spawning (Reiser 1986), resident trout have been eliminated from Big Deer Creek below the confluence of the South Fork of Big Deer Creek (Blackbird effluent). Physical habitats and fish were inventoried by Forest Service researchers in August 1991 (snorkel counts). Nearby streams of similar size with similar high quality trout habitat have abundant trout populations. The results of summer 1991 snorkeling surveys among different Panther Creek tributaries are given in Table 25. Big Deer was not surveyed above the confluence of the contaminated South Fork (Price pers. comm. 1992).

Table 25. Trout densities in Panther Creek tributaries 1991(all habitat types)

	Moyer Cr.	Deep Cr.	Napias Cr.	Lower Big Deer Cr.	Clear Cr.
# Trout /100 m² (all species)	5.8	7.0	9.7	0	17.1
Source: U.S. Forest Service 1992					

In July 1992, two 100-meter sections of Big Deer Creek above and below the South Fork effluents were also electrofished (USFS 1993). No fish were found below the confluence of the South Fork. Above the confluence, rainbow trout were the only fish species identified with an overall density in that section of 26 fish/100m^2. The population was apparently made up of smaller than average fish with abundances comparable to or somewhat higher than in other Panther Creek tributaries sampled or throughout the Salmon River basin (Rich et al. 1992; USFS 1993). This may be attributed the smaller size of fish in the upper Big Deer population. This in turn may be related to the trout's restriction to the smaller, upstream habitats away from rearing habitats (larger pool complexes) in the lower reaches of Big Deer Creek.

Expected salmonid densities

Based on results of their monitoring both pristine and degraded salmonid habitats and on salmonid habitat requirements, Scully et al. (1990) estimated rearing potentials (carrying capacity) for steelhead and chinook parr. Mean steelhead densities from 89 streams in the Salmon and Clearwater drainages that were estimated to be near carrying capacity ranged from 10.0 - 24.4 per square meter for poor to excellent habitats, respectively. Panther Creek monitoring sections show a trend of reduced juvenile steelhead and chinook densities relative to both these other streams and upstream sections of Panther Creek. Even with repeated rainbow-steelhead stockings above and below Panther Creek, surveyed reaches below Big Deer Creek always show severely diminished abundances, and below Blackbird Creek, somewhat reduced abundances.

Literature on copper toxicity to fish

Lethality

The lethality of waterborne copper to salmonids has been extensively studied. **All** of the studies reviewed reported increased lethality to salmonids at copper concentrations that have been reported from Panther Creek downstream of the mine under hardness, alkalinity, and pH conditions similar to those that have been reported from Panther Creek[19]. Copper concentrations that resulted in lethality and other adverse effects in several studies are compared to 1985-1986 Panther Creek copper concentrations in Figure 19[20]. Most studies reported concentrations acutely lethal to 50% of the fish, referred to as LC_{50} values, between 15 - 40 μg/l for exposure times between 4 to 8 days.

Of particular interest for Panther Creek conditions are Chapman's (1982) early life stage testing with chinook salmon in low-alkalinity, low-hardness water. The chinook embryo-larvae were exposed to copper from just before fertilization until about 30 days after swim up, for a total exposure time of 120 days. The results are summarized in Table 26. Reduced growth occurred at all concentrations, giving an effects threshold of ≤ 7.4 μg/l. At 20 μg/l, 94% of the chinook died compared to 9% in the controls. These 120 day duration test results are more representative of the extended exposures of several months that juvenile salmon would encounter rearing in Panther Creek than are the acute (4 to 6 day) toxicity thresholds. Long-term, relatively low-level exposures that appear safe in acute tests could result in high mortality and place fish at competitive disadvantages due to reduced growth. Results of these early life stage tests also illustrate seasonal differences in vulnerabilities of salmonids to copper toxicity. Brief exposure of swim-up steelhead fry to copper concentrations between 7 and 20 μg/l resulted in a pulse of mortality. Exposure at fertilization, eyed egg stage, or hatch did not result in a mortality phase. Significance in nature could be that a brief exposure of copper during swim-up from snowmelt or a thunderstorm could result in pulse of mortality (Chapman 1994). Co-occurrence of chinook fry emergence in late spring and higher copper concentration in Panther Creek in late spring increases the likelihood of the occurrence of conditions lethal to early life-stages of salmonids (Figures 5, 6, and 9).

Table 26. Chinook salmon 120-day early life stage testing with copper (Chapman 1982).

Copper (μg/l)	1.2 (Control)	7.4	9.4	11.7	15.5	20.2
Mortality (%)	8.95	10.00	9.55	20.30	48.50*	94.15*
Mean length (mm)	47.9	43.0*	40.6*	40.2*	36.5*	-
Mean wet weight (g)	.990	.624*	.493*	.518*	.341*	-

*Significantly different from control, p= .05, Dunnetts Procedure. Total hardness (mg/l $CaCO_3$) 25.4 ± 3.9, alkalinity (mg/l $CaCO_3$) 23.9 ± 2.3, pH 7.32 ± 0.07.

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[19] McKean et al. 1991, Cuisimano et al. 1986; Laurén and McDonald 1986; Seim et al. 1984; Finlayson and Verrue 1982; McKim 1985; Chapman 1982, 1978; Chapman and Stevens 1978, Miller and MacKay 1980; Chakoumakos et al. 1979, Howarth and Sprague 1978, Schreck and Lorz 1978, Stevens 1977; Lorz and McPherson 1976, Hazel and Meith 1970 and Sprague and Ramsay 1965.

[20] Data from Wai and Mok (1986). At writing, this is the only available Panther Creek study documenting reliably quantified time-series water chemistry data.

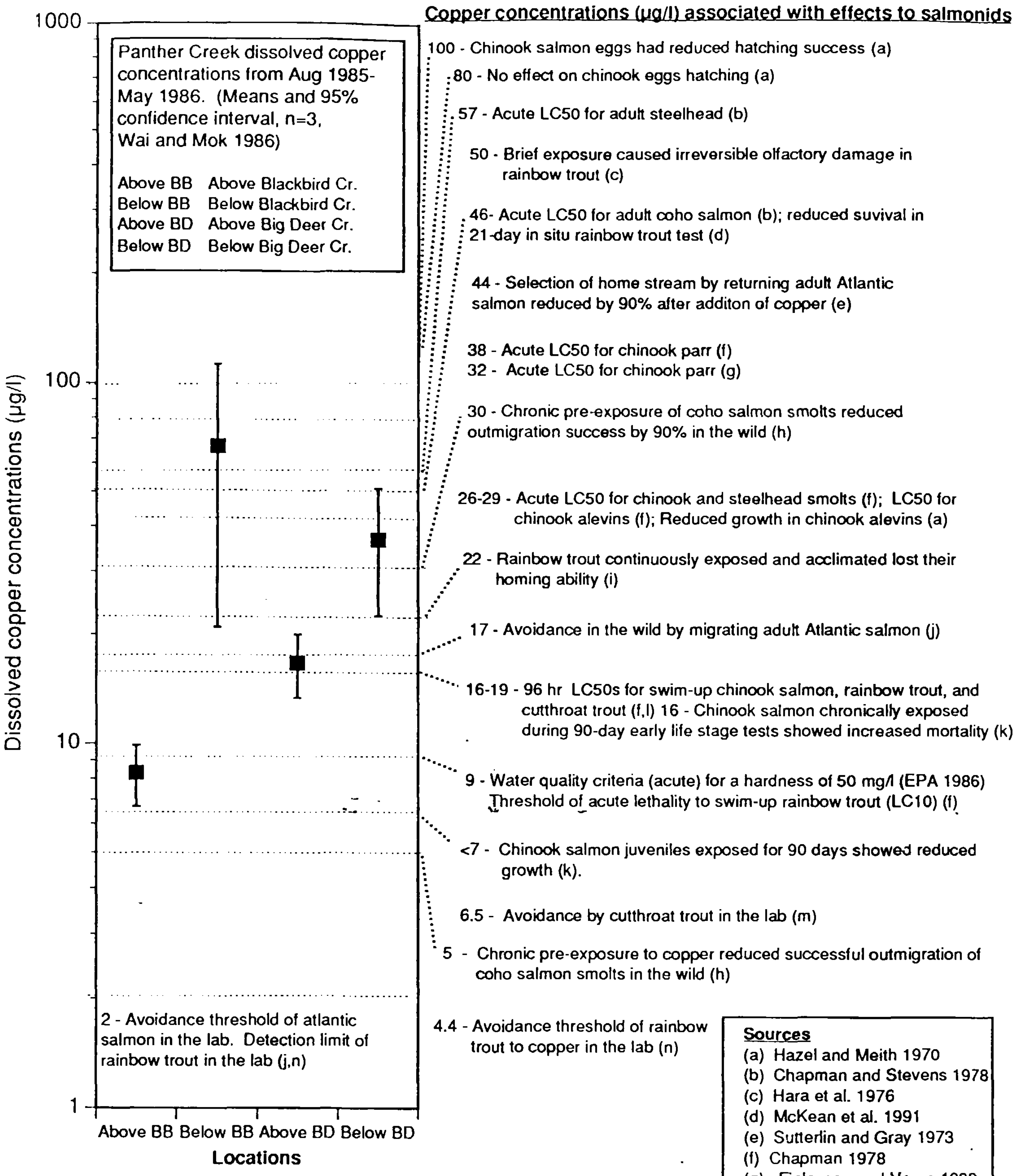

Figure 19. Panther Creek copper concentrations compared with the effects of dissolved copper to salmonids reported under conditions of hardness, alkalinity, and pH similar to Panther Creek conditions (pH 6.5 - 8.5, hardness 20-50 mg/l, and alkalinity 10-50 mg/l, both as as $CaCO_3$).

Sources
(a) Hazel and Meith 1970
(b) Chapman and Stevens 1978
(c) Hara et al. 1976
(d) McKean et al. 1991
(e) Sutterlin and Gray 1973
(f) Chapman 1978
(g) Finlayson and Verue 1982
(h) Lorz and McPherson 1976
(i) Saucier et al. 1991
(j) Sprague et al. 1965
(k) Chapman 1982
(l) Chakoumakos et al. 1979
(m) Woodward 1993
(n) Giattini et al. 1982

Effects of cyclic or episodic copper exposures on salmonid toxicity

Environmental doses of copper in the Panther Creek would vary cyclically during day-night temperature changes during spring runoff or in pulses during summer thunderstorms. Seim et al. (1984) studied the chronic relationships between continuous and intermittent copper exposures and the survival growth and development of steelhead trout embryos and larvae. They found a two-fold increase in chronic toxicity (impaired growth) when steelhead eggs and fry were intermittently rather than continuously exposed to copper at the same mean concentration. Similar to spring snow-fed stream flow patterns, copper was introduced daily for a single 4.5-hour period in which concentrations quickly increased and decreased four to five times the mean copper concentrations. Steelhead embryos exposed in this intermittent manner hatched later than control groups or groups continuously exposed to higher concentrations of copper. Intermittent exposure of copper reduced growth of steelhead fry more than continuous exposure (EC 50 of 27 µg/l vs. 46 µg/l, respectively, hardness 120 mg/l, alkalinity 126 mg/l $CaCO_3$, pH 7.4-7.9).

Bergman (1993a) simulated pulses of metals that result from runoff from mill tailings in a river floodplain during summer thunderstorms. Concentrations of metals (copper, zinc, lead, and cadmium) in concentrations measured during rainstorms were introduced in an 8-hour pulse to aquaria with rainbow trout juvenile and fry. Survival was significantly reduced for both juveniles and fry after these brief pulse exposures (fry were more sensitive). Mortality was delayed. In one test all rainbow trout survived during the 8 hour pulse but <u>all</u> died during the following 96 hours. Delayed mortality following a pulse from a rainstorm could significantly reduce survival of fry in the wild and thus lower recruitment of sub-adults into the resident trout populations. Also, because rainstorm pulses are associated with turbid water and because trout fry are very small, fish kills involving trout fry could have occurred and gone unnoticed in Panther Creek. While most toxicity tests are conducted under constant test concentrations, these two studies indicate that variable exposures encountered in nature may be more severe than estimated in many experiments.

Acclimation

Several studies have shown that salmonids may acquire a tolerance to metals exposure: salmonids may be able to tolerate low levels of metals without succumbing. However the costs of this acclimation include reduced growth and reduced swimming capacity due to reduced ability to digest food and thus lower conversion efficiency of food to energy. The longer-term consequence of this acquired toleration was starvation, increased susceptibility to disease and predation, with overall reduced survival potential at the population level (McKim 1985). Dixon and Sprague (1981) found that after fish were pre-exposed to a sublethal copper level, the fish could <u>either</u> become more tolerant or more weakened. Their study of the copper response showed that there was a greater increase in tolerance with higher pre-exposure concentrations and also with time of pre-exposure over the three-week test period. Acclimation approximately doubled acute copper tolerance. When returned to clean water, trout lost their increased tolerance. Acclimation stimulated production of a liver protein, probably a metallothionein, the apparent defense mechanism. Acclimated fish were able to maintain their body burden of copper at a steady level, with no increase during subsequent exposures that should have been quickly lethal. There was an apparent threshold for triggering this defense mechanism at one-fifth the acute toxicity threshold concentration: above that there was reduced growth and increased tolerance, and below there was no effect on growth. At lower levels of exposure (one-tenth the lethal level) fish did not have a defense mechanism and were more sensitive to subsequent lethal levels. Miller et al. (1993) subjected rainbow trout to an acute lethality test after pre-exposing them to dietary and waterborne copper for six weeks. Waterborne pre-exposure significantly increased the tolerance of trout to waterborne copper.

In rainbow trout, the gill has been shown to be the primary target for waterborne acute copper toxicity, as opposed to a food chain exposure route. At lower concentrations where the gills can withstand elevated copper, the liver appears to become the primary organ for accumulating and detoxifying copper, apparently resulting from increased synthesis of the specific metal-binding protein metallothionein (Laurén and McDonald 1987). Cobalt also binds metallothionein, giving the potential for additive effects when the individual concentrations of trace metals are below levels expected to be lethal to aquatic life (Leland and Kuwabara 1985). McKean et al. (1991) showed with both caged fish studies and tissue analysis of native fish that rainbow trout and coho salmon exposed to copper concentrations produced metallothionein at levels corresponding to exposure. As exposure increased, increased sub-lethal effects were observed,that is, gill structure damage and increased metallothionein production. Even when gill histology showed severe damage, all the salmon survived a 96-h LC_{50} acute lethality test. These results caution against solely relying on acute lethality testing of salmonids: even with high survival (e.g. 95% below Blackbird Creek in May 1984) it would be prudent to consider other variables as well to interprete conditions accurately.

In studies of wild brown trout (*Salmo trutta*) acclimated to elevated metals in the Clark Fork River, Montana, and naive hatchery trout, the acclimated trout were more resistant to acute metals elevations as result of genetic and physiological adaptations. Copper generally becomes toxic by disrupting the osmoregulatory balance; the metabolic demands from detoxifying metals to maintain homeostatic balance injure trout populations by diverting resources from normal growth process. Over the long run, these genetic and physiological changes have associated costs that may ultimately reduce survival in the wild (Bergman 1993b).

These studies provide clues to the persistence of resident salmonids in all reaches of Panther Creek, albeit at reduced numbers, downstream of Blackbird Creek.

Behavior

Even briefly elevated copper levels entering the river from Panther Creek could potentially interfere with the migration and thus survival of anadromous populations in the drainage. Field and laboratory experiments with salmonids have shown avoidance of low concentrations of copper, disruption of downstream migration by juvenile salmon, loss of homing ability after chronic low level copper exposure, loss of avoidance response to even acutely lethal concentrations of copper follow long term habituation to low level copper exposure.

Saucier et al. (1991) examined the impact of a long-term sublethal copper exposure on the olfactory discrimination performance in rainbow trout. When controls were given a choice between their own rearing water against either well water or non-specific water, they significantly preferred their own rearing water, whereas both copper-exposed groups showed no preference. Their results demonstrate that a long-term sublethal exposure to copper, as it commonly occurs in the "natural" condition, may result in olfactory dysfunction with potential impacts on fish survival and reproduction. Chapman (1994) reported that long-term sublethal copper exposures had impaired the avoidance performance of salmonids. Steelhead trout, acclimated to low copper levels by surviving about three-months early life stage toxicity testing, subsequently failed to avoid much higher, acutely lethal concentrations. Following about three-months continuous exposure to 9 µg/l copper from fertilization to about 1-month after swim up, the copper-acclimated fish and control fish with no previous copper exposure were exposed to a range of copper concentrations from 10 to 80 µg/l in avoidance-preference testing[21]. The acclimated trout failed to avoid even the highest copper concentrations while most of the unexposed fish avoided all concentrations.

Studies have shown that salmonids can detect and avoid copper at low concentrations when tested with concentration gradients. (Giattina et al. 1982) reported that the avoidance threshold of rainbow trout to copper was 4.4 -6.4 µg/l when tested in steep and shallow concentration gradients. At exposure to extremely high copper levels, trout showed diminished avoidance to acutely lethal concentrations. Avoidance thresholds of 2 µg/l copper have been reported for Atlantic salmon, concentrations that are less than one-tenth of the incipient lethal concentrations (Saunders and Sprague 1967). Cutthroat trout have been shown to have an avoidance threshold of 6.5 µg/l copper in the laboratory (Woodward 1993). Further, rainbow trout avoided low concentrations of copper but were apparently intoxicated and sometimes attracted to very high concentrations (Giattina et al. 1982).

Salmonid preference/avoidance behavior with copper and other metals have been related to narcotic effects or dysfunction in the chemoreceptors. Numerous studies have shown that low levels of metals in freshwater cause olfactory and chemoreceptor impairment in salmonids and other fish. For example, Hara et al. (1976) examined the effects of copper on the olfactory response in rainbow trout. Threshold copper concentration of 8 µg/l caused a minimal depression of bulbar response of rainbow trout to olfactory stimulant. Irreversible damage to the olfactory receptor occurred at brief exposure to 50 µg/l copper. Further physiological and behavioral studies by Rehnberg and Shreck (1986) showed that copper exposure reduced coho salmon's ability to detect natural odors.

Field studies have reported that copper impairs both upstream spawning migration of salmon and downstream out migration of juveniles. Avoidance of copper in the wild has been demonstrated by: delay in upstream passage of Atlantic salmon while moving past the

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[21] The tests used the counterflow avoidance-preference test chambers described in Giattina et al. (1982)

copper contaminated reaches of the river to their upstream spawning grounds; unnatural downstream movement by adults away from the spawning grounds; and by increased straying from their contaminated home stream into uncontaminated tributaries. Avoidance thresholds in the wild of 0.35 to 0.43 toxic units were higher than laboratory avoidance thresholds (0.05 toxic units) perhaps because the laboratory tests used juvenile fish rather than more motivated spawning adults or perhaps of motivation to move upstream, which was lacking in laboratory tests. For this study 1.0 toxic units was defined as an incipient lethal level, ILL (analogous to a 24 hr LC_{50}), of 48 μg/l in soft water[22] (Saunders and Sprague 1967; Sprague et al. 1965). Studies of home-water selection with returning adult salmon showed that addition of 44 μg/l copper to their home-water reduced the selection of their home stream by 90% (Sutterlin and Gray 1973).

These results may pertain to natural conditions in Panther Creek. In 1986, 3,383 adult chinook salmon were released at the mouth of Panther Creek (Table 3). At least some fish must have passed through the Blackbird Mine effluent since the following year juvenile chinook were observed in Panther Creek above Blackbird Creek (Table 23). Water quality was not monitored; however, results from fall 1985, 1991, and 1992 (Figure 9) suggest that the fish would have passed through dissolved copper concentrations of about 20 μg/l. Releases of similar levels of copper from a mine drainage into a salmon spawning river resulted in 10% to 22% repulsion of ascending salmon during four consecutive years compared to 1% to 2% swimming back downstream prior to mining (Sprague et al. 1965)

Sublethal copper exposure has been shown to interfere with the downstream migration to the ocean of yearling coho salmon. Lorz and McPherson (1976, 1977); and Lorz et al. (1978) evaluated the effects of copper exposure on salmon smolts' downstream migration success in a series of 14 field experiments. Lorz and McPherson (1976, 1977) exposed yearling coho salmon for six to 165 days to copper concentrations varying from 0 - 30 μg/l[23]. They then marked and released the fish during the normal coho migration period and monitored downstream migration success. The fish were released simultaneously, allowing for evaluation of both copper exposure concentrations and exposure duration on migration success. All exposures resulted in reduction of migration compared with unexposed control fish. Migration success decreased with both increasing copper concentrations and increased exposure time for each respective concentration (Figure 20). Exposure to 30 μg/l copper for as little as 72 hours caused a considerable reduction in migration compared to control fish. These concentrations were one-tenth to one-third the 96-hour LC_{50} for the same stock of juvenile coho salmon in soft water. Lorz et al. (1978) further tested downstream migration with yearling coho salmon previously exposed to copper, cadmium, copper-cadmium mixtures, zinc, and copper-zinc mixtures. Copper concentrations in all tests were held at 10 μg/l. In all cases, the copper exposed fish again had poorer migratory success than did controls. The other metals did not show the dose-dependent result found for copper. These studies suggest that exposure to copper concentrations at levels found in Panther Creek below the Blackbird Mine may impair downstream migration, and thus survival, of salmon smolts.

Avoidance of chemical contaminants by migratory salmon that home by chemoreception has been demonstrated to impede migratory pathways, reducing the numbers that spawn. From 1980-1982, sub-lethal levels of a contaminant (fluoride) from an aluminum mill at the John Day Dam on the Columbia River were associated with a significant delay in salmon passage

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[22] Hardness was about 20 mg/l $CaCO_3$

[23] Nominal concentrations: measured concentrations were within ± 10% nominal concentrations. Background concentration of copper in well water was reported to be < 2.0 μg/l, pH 6.8-7.5, alkalinity 72-75, and hardness 89-99 mg/l, both as $CaCO_3$.

and decreased survival. Salmon took an average of 36 hours to pass up the fish ladder at the Bonneville and McNary dams compared to 157 hours delay at the John Day Dam, Greater than 50% mortality occurred between the Bonneville and McNary dams (above and below the John Day dam), compared to about 2% mortality associated with the other dams. Damker and Dey (1989) introduced similar levels of the contaminant in streamside test-flumes alongside a salmon spawning stream (Big Beef Creek, Washington). Significant numbers of adult chinook salmon failed to move out of their holding area and continue upstream; those that did move upstream chose the non-contaminant side of the flume. By adjusting the dose, Damker and Dey predicted a threshold detection limit for avoidance by salmon. The mill subsequently reduced its release of the contaminant to below these experimental threshold levels that did not show a response in the streamside tests. Afterwards, fish passage delays and salmon mortality between the dams decreased to 28 hours and <5% (Damkaer and Dey 1989). This study suggests that the delay due to avoidance of a chemical affected the spawning success of migrating adult salmonids. These results are also consistent with the field studies of salmon migration in copper-contaminanted streams and from laboratory avoidance/preference testing. Experimental avoidance/preference testing thus appears to be relevant to fish behavior in nature.

Hartwell et al. (1987b) compared the avoidance behavior of schools of fish exposed to metals in an artificial stream supplied with raw river water and in a natural stream. Fish exposed continuously for three months to a blend of four metals in river water were tested during summer in the artificial and natural streams. Control (unexposed) fish avoided concentrations of metals in the artificial streams at about one-half the concentrations that they avoided in the natural stream. Exposed fish did not avoid metals concentrations 42 to 40 times higher than concentrations that the unexposed fish avoided in the artificial and natural streams, respectively. These comparisons indicate that laboratory avoidance is a relevant indicator of field behavior and may be somewhat conservative (by a factor of about two in these tests).

The literature review of both laboratory and field studies clearly indicates that salmonid behavior is adversely affected by waterborne metals, including copper. These responses are supported by field studies of homing adults and outmigrating juveniles. A contaminant (fluoride) that elicited similar avoidance behavior in streamside chambers to copper avoidance in laboratory test chambers was linked with substantial mortality of salmon by delaying their migration. These behavior modifications resulting from sublethal exposures may not directly lead to the death of the individual but may nonetheless cause disturbances of major ecological significance (Rand 1985).

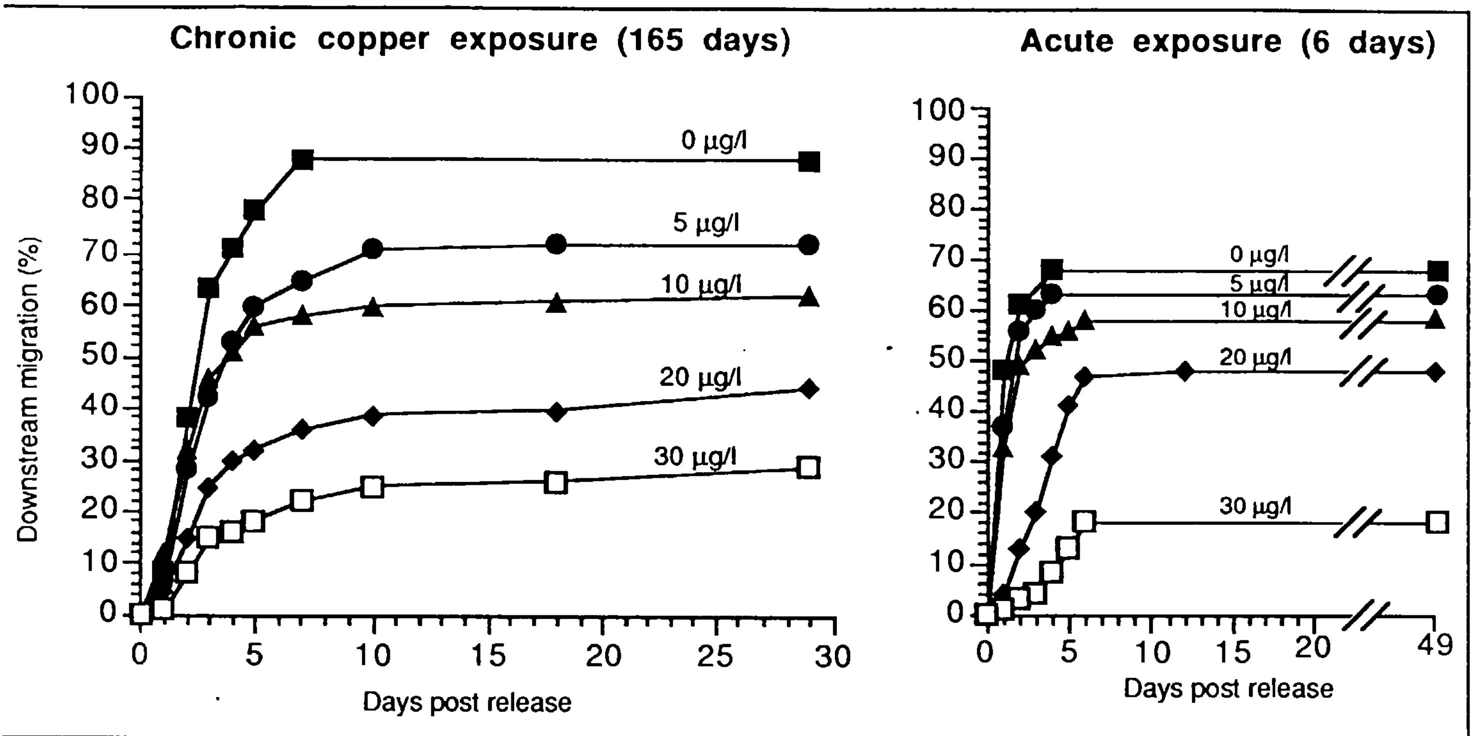

Figure 20. Reduction in percentage of yearling salmon migrating downstream following copper exposure in fresh water(Lorz and McPherson 1977).

Food chain effects

Metals are available to benthic organisms and fish through uptake of dissolved forms across the gill and assimilation of particulate material through the food chain. Benthic invertebrates are important food sources for fish and are essential in the aquatic ecosystem for breaking down refractory organic material into more readily used forms for trophic energy transfer and for nutrient cycling (ASTM 1991). Detrital and filter feed invertebrates may accumulate more metals by ingestion than from water (Kiffney and Clements 1993). Fish that feed at higher trophic levels may be chronically exposed to metals through the food chain as well as water and sediment. Food chain uptake and waterborne exposure to metal should both be considered when evaluating the risk of metals contamination to organisms in Panther Creek.

Effects of chronic exposure of rainbow trout to combined waterborne and natural dietary exposure to metals were studied in the Clark Fork River, Montana. Woodward et al. (1994) exposed newly hatched trout alevins to 12 different combinations of metal-contaminated water and diet. Benthic invertebrates with relatively high levels of arsenic, copper, cadmium, lead, and zinc were collected from the Clark Fork River and fed to rainbow trout for 90 days in waters simulating different ambient metals concentrations from the river. They found that metals in the food chain were more important in reducing survival and growth of early life stage fish than were metals in the water. Metals residues in the dietary-exposed trout were similar to concentrations found in Clark Fork fishes. Concentrations used in the tests ranged from no metals to twice the EPA water quality criteria (AWQC). The fish were fed diets of field-collected forage fish and invertebrates from the Clark Fork River and from an uncontaminated reference river. Concentrations of arsenic, copper and lead were 10 to 20 times higher in a diet of invertebrates from the Clark Fork River than from the reference diet. Survival and growth of rainbow trout were reduced by exposure to the metal-contaminated invertebrate diet. Metals at twice the water quality criteria had no effect on survival or growth of trout independent of the effect of diet. Concentrations of metals in the metal-contaminated invertebrate diet collected from riffles of the Clark Fork River were similar to concentrations of metals in amphipods exposed in the laboratory to sediment from the river. Although the sediments were not toxic to the amphipods, bioaccumulation of metals in the amphipods occurred at levels similar to those that in the invertebrate diet of trout that caused reduced growth and survival (Kemble et al. 1993). Thus, similar effects on trout may be inferred for Panther Creek where some sediment stations did have high toxicity, indicating that metals are at least as readily bioavailable to invertebrates as in the Clark Fork.

Although biomagnification was not observed and bioconcentration factors were small in the Clark Fork study, the amount of metals transferred by food was high enough to attain biologically harmful concentrations in fish. Once in the gut of fish, heavy metals are absorbed into the gut tissue where they are distributed to the liver, kidney, and muscle. The contaminated invertebrate diet and/or the water exposure to twice the AWQC also resulted in scale loss of adult fish. The energetic costs of scale regeneration and potential enhancement of disease susceptibility may be detrimental to fish in the wild (Farag and Bergman 1993). Reduced growth of fish receiving dietary metals could be due to the energy required for binding the metals to proteins in the livers where they are excreted through the bile or through decreased assimilation efficiency in the gut (Miller et al. 1993; Woodward et al. 1994). Because growth rate directly relates to fecundity in teleost fish, it is important for reproductive success and is one indicator of population fitness (Bergman 1993). Miller et al. (1993) and Mount et al. (1993) also studied of dietary effects of copper to rainbow trout by adding inorganic copper to fish food and by brief exposures of brine shrimp post-hatch to copper-spiked brine solution. These studies had higher threshold effects concentrations than did the Clark Fork River study using field collected invertebrates (435 vs. 684 or 828 mg/kg copper, dry weight respectively). These results suggest that the field invertebrates may metabolize the metals into a protein-bound form that is more bioavailable to predators than the laboratory food-spiking experiments. Thus the laboratory exposures with inorganic metals may underestimate bioavailability in the wild.

Additional studies of metals contamination in streams where copper concentrations in water were low, but remained elevated in sediments, macrophytes, or benthic invertebrates, include the Arkansas River, Colorado and Iron Creek, Idaho. Kiffney and Clements (1993) found that metals concentrations in *aufwuchs* (biotic and abiotic material accumulating on submerged surfaces) are bioavailable and represent a significant source of metals to benthic macroinvertebrates, even though ambient water concentrations were relatively low. Sampling sediments and aquatic mosses in Iron Creek, Idaho (within the Blackbird study area), Erdman and Modreski (1984) found that copper and cobalt were more highly concentrated in the mosses than in sediments in some areas; macroinvertebrates live in and ingest these metal-enriched mosses.

In sum, studies on effects of metals in the food-chain for salmonids have demonstrated increased mortality and decreased growth both in laboratory and field studies (Miller et al. 1993, Woodward et al. 1994). The copper-and other metals-contaminated Clark Fork River system is relevant in many ways to the copper-and other metals-contaminated Panther Creek system. The Clark Fork research has demonstrated a consistent pattern of metal accumulation in tissues, degeneration of digestive cells (likely leading to reduced growth), cellular damage and synthesis of enzymes required to detoxify/excrete metals (incurring a metabolic cost, that may reduce growth and long term survivability). These effects were both observed in laboratory-exposed and free-ranging fish, as was cell damage (including digestive system degeneration, lipid peroxidation and liver abnormalities). These indicators are diagnostic of copper poisoning in fish. (Bergman 1993c; Farag and Bergman 1993).

Other adverse effects

Other adverse effects of copper to salmonids reported in the literature include weakened immune function and disease resistance, increased susceptibility to stress, liver damage, reduced growth, impaired swimming ability, and weakened eggshells.

Stevens (1977) reported that pre-exposure to sublethal levels of copper interfered with the immune response and reduced the disease resistance in yearling coho salmon. Stevens vaccinated salmon with the bacterial pathogen *Vibrio anguillarum* prior to copper exposure to investigate the effects of copper upon survival and the immune response of juvenile coho salmon. Following copper exposure the survivors were challenged under natural conditions to *V. anguillarum*, the causative agent of vibriosis in fish. Vibriosis is a disease commonly found in wild and captive fish from marine environments and has caused kills of coho and chinook salmon. The fish were exposed to constant concentrations of copper for about one month at levels that covered the range from no effect to causing 100% mortality. The antibody titer level against *V. anguillarum* was significantly reduced in fish exposed to this concentration of copper when compared to that developed in control fish. The survivors of the copper bioassays were then exposed in saltwater holding ponds for an additional 24 days to the *V. anguillarum* pathogen. The unvaccinated, non-copper exposed control fish had 100% mortality and the vaccinated, non-copper exposed fish had the lowest mortality. The vaccinated, copper-exposed fish had increasing mortality corresponding to the lower antibody titers levels which in turn corresponded to the increasing copper exposure levels. Thus, copper exposure may reduce the fish's immune function and disease resistance.

Schreck and Lorz (1978) studied the effects of copper exposure to stress resistance in coho salmon. Fish that were pre-exposed and acclimated to sublethal copper concentrations and control fish were subjected to severe handling and confinement stress. Copper-exposed fish survived this stress for a median of 12-15 hours while the control fish had no mortality at 36 hours. Schreck and Lorz concluded that exposure to copper placed a sublethal stress on the fish which made them more vulnerable to secondary stress including disease, pursuit by predators, adaptation to saltwater, or handling.

Chronic, low-level copper exposure to brook trout eggs has resulted in weakened chorions (eggshells) and embryo deformities. After hatching, poor yolk utilization and reduced growth were demonstrated. These overall weakened conditions may reduce survival chances in the wild (McKim 1985). Copper accumulation in the liver of rainbow trout caused degeneration of liver hepatocytes, which resulted in reduced ability to metabolize food, reduced growth, or eventual death (Leland and Kuwabara 1985). Waiwood and Beamish (1978b), Chapman (1982), Seim et al. (1984), McKim (1985), and Woodward (1994) have also observed reduced growth of salmonids to in response to chronic copper exposures. Waiwood and Beamish (1978a) reported that rainbow trout exposed to sublethal copper levels had reduced swimming performance and reduced oxygen consumption apparently due to gill damage and decreased efficiency of gas exchange.

In sum, in addition to direct mortality, copper stress has been shown to be a "loading" stressor by increasing the cost of maintenance to fish and a "limiting" stressor, limiting oxygen consumption and food metabolism. Reduced growth may result in increased susceptibility to predation and impaired swimming ability may result in an impaired escape reaction and prey hunting, with a possible consequence of reduced survival at population level.

Toxicity of cobalt and copper-cobalt mixtures to aquatic life

Cobalt is generally rare in natural aquatic systems, and much less information has been published on cobalt toxicity to aquatic life than copper. EPA presently does not have an ambient water quality criterion for cobalt due to the rarity of cobalt's occurrence and to a lack of toxicological data. Elevated concentrations of cobalt do occur in association with copper-nickel ores in northeastern Minnesota; the Minnesota Pollution Control Agency has established cobalt criteria for the protection of aquatic life (Kimball, personal communication 1993). The available literature on cobalt toxicity indicates that it most likely contributes to the degraded water quality of Panther Creek. *Daphnia magna* was the most sensitive species reported for cobalt toxicity followed by rainbow trout and fathead minnows (Table 27). The large differences in chronic and acute toxic levels of cobalt reported in several of the studies suggest that cobalt toxicity is slower-acting than metals such as copper. Short-duration tests thus may underestimate long-term risks of cobalt to aquatic life.

Lind et al. (1978) tested the acute toxicity of copper-cobalt mixtures to *Daphnia pulicaria* and fathead minnows (48-hour and 96-hour LC_{50}'s, respectively). The joint toxicity of copper and cobalt appeared to be slightly slightly less than additive for fathead minnows and approximately additive to *D. pulicaria*. These studies indicate that either copper or cobalt could cause injury to aquatic life at concentrations reported in Panther Creek (Table 15) and that mixed together this risk is increased.

Toxicity of arsenic to aquatic life

The available information suggests that arsenic does not play a significant role in impacts to aquatic life in Panther Creek. Although greatly elevated in sediments downstream from Blackbird, arsenic has not been reported in surface water at concentrations expected to cause toxicity to most aquatic life. For example, Birge et al. (1980) reported that the threshold of toxicity of total arsenic to rainbow trout was 42 µg/l in 28-day embryo larval tests. The national chronic criterion for trivalent arsenic is 190 µg/l. The highest concentration of total arsenic reported in Panther Creek was 6.2 µg/l, almost all of which was the less toxic pentavalent form. The highest Panther Creek concentrations of the more toxic trivalent arsenic reported were 2.1 µg/l (Table 15; Mok and Wai 1989). These low levels of arsenic in surface waters, nearly perfect correlation with iron in Panther Creek sediments (Figure 12), lack of correlation to amphipod toxicity (Figure 16), and insolubility in leaching tests (Table 17) suggest that arsenic is complexed with iron hydroxides and has low bioavailability under ambient conditions in Panther Creek. Similarly, Cain et al. (1992) reported low bioavailability and Smith et al. (1992b) reported little partitioning of arsenic from sediments to surface waters in streams with elevated arsenic concentrations in sediments.

Table 27. Effects of cobalt concentrations on aquatic life under various conditions

Species	Cobalt (µg/l)	Hardness (mg/l)	Test endpoint
MN aquatic life criterion (chronic)	5	N/A	Criterion based on protecting *Daphnia magna* (g)
Daphnia magna	5.1	Hard (220)	MATC. NOEC for 28-day survival, molting, and reproduction endpoints(a)
Daphnia magna	9.3	Hard	LOEC — reduced reproduction, 28-day exposure (a)
Daphnia magna	10	Soft (45)	Reproductive impairment (16% reduction), 21 day exposure (b)
D. magna	21	Soft (45)	LC50 — 21-day exposure (b)
Daphnia magna	27	Hard(220)	LC50 —28-day exposure (a)
Oncorhynchus mykiss (Rainbow trout)	38	Mod-hard (96)	LC1, concentration lethal to 1% of test population — considered the toxicity threshold by the authors. Embryo-larval toxicity tests. Treatment maintained continuously from fertilization through 4 days posthatching, giving an 28-day exposure period (c)
Pimephales promelas (Fathead minnow)	112	Soft (48)	Chronic toxicity to embryos and larvae (significantly lower growth & survival than lowest treatment α0.05) (d)
O. mykiss	120	Mod-hard (96)	LC10, concentration lethal to 10% of test population (c)
Panther Cr. upper level	120	50	Highest level reported from studies with adequate data quality for quantitative comparisons (f)
7 species	144	Soft (25-50)	Proposed cobalt soft water acute criteria (1/2 acute LC50 for 95th percentile of species tested in soft water) (e)
P. promelas	290	Hard(220)	MATC from 28-day exposure to embryo-larvae (a)
MN aquatic life criterion (acute)	436	N/A	Criteria based on EPA national procedures, direct aquatic life toxicity, using 14 species (g). 1/2 acute LC50 for 95th percentile of species tested
O. mykiss	490	Mod-hard (96)	LC50, Median lethal concentration of test population. (c)
P. promelas	531	Soft (48)	Acutely toxic (96-hour LC50) to 8-wk old minnows (d)
P. promelas	1232	Soft (50)	NOEC in 7-day exposure to larvae (< 24 h old) (e)
P. promelas	1245	Soft (48)	Acutely toxic (96-hour LC50) to 5-15 day old minnows (e)
D. pulicaria	2025	Soft (50)	48-hour LC50 (d)

MATC-Maximum acceptable threshold concentration. NOEC- No observed effects concentration. LOEC - Lowest observed effects concentration. (a) Kimball 1978; (b) Biesinger and Christensen 1972; (c) Birge et al. 1980; (d) Lind et al. 1978; (e) Diamond et al. 1992;.(f) Table 15; (g) Minnesota Pollution Control Agency 1990

VI. Review

Before large scale operation of the Blackbird Mine began in 1945, chinook salmon and steelhead trout were numerous in the Panther Creek drainage. By the early 1960s these anadromous fish populations were eliminated by releases of toxic materials from the Blackbird Mine. Should they return, anadromous fish in the Panther Creek system would continue to be at risk during most stages of their life history as the result of metals contamination in their habitats (Figure 21).

Acid mine drainage and copper loading of up to 500 kg/day are the major water quality problems. Erosion and leaching from the old mine waste dumps and water seeping from the underground mine workings of the inactive Blackbird Mine are the principal sources of contaminants reaching the river. During high water, sediments from the waste rock piles enter Panther Creek tributaries, contaminating and silting in the streambed gravels, making the habitat less suitable for aquatic insects and for fish spawning (Reiser 1986; Chapman et al. 1991). Metals concentrations in the surface waters and sediments of Panther Creek show a pattern of elevated levels below the streams draining the Blackbird Mine. These elevated levels correspond with the measured impacts to the aquatic biota. Intermittent monitoring of copper levels in water over the last 20 years shows little improvement. Comparison of metals levels in sediments over a 12-year interval shows no decline. Copper and cobalt levels in surface water and stream sediments, downstream from the Blackbird Mine area are much higher and are elevated over a larger area than in undisturbed areas with similar natural copper-cobalt mineralization.

A number of studies have indicated severe stress to the aquatic environment of Panther Creek, lower Big Deer Creek, and lower Blackbird Creek associated with releases from the Blackbird Mine:

- Chinook spawning runs began declining in the 1950s, long before major downstream dam effects, and were completely eliminated by 1962.
- Few anadromous fish return to Panther Creek relative to returns to other streams in the region in spite of re-stocking efforts.
- Caged fish studies downstream of Blackbird Creek have shown reduced survival of salmonids in short-term exposures with larger juvenile fish, which are not the most sensitive life stage of these fish.
- There are reduced populations of salmonids in affected reaches relative to upstream despite repeated stocking efforts.
- Benthic macroinvertebrates communities, the main food supply for salmonids and an important indicator of stream impairment, are severely stressed in affected reaches. Overall populations, with the exception of pollution-tolerant chironomids, are considerably reduced, and sensitive species of mayflies, caddisflies, and stoneflies, which are principal components of the diet of salmonids, are mostly absent.
- Laboratory toxicity testing of Panther Creek sediments with the amphipod *Hyalella azteca* showed reduced survival at stations from the downstream reaches relative to the upstream station.
- Bioaccumulation of metals in salmonids has been shown by increased concentrations in fish collected below the Blackbird mine.

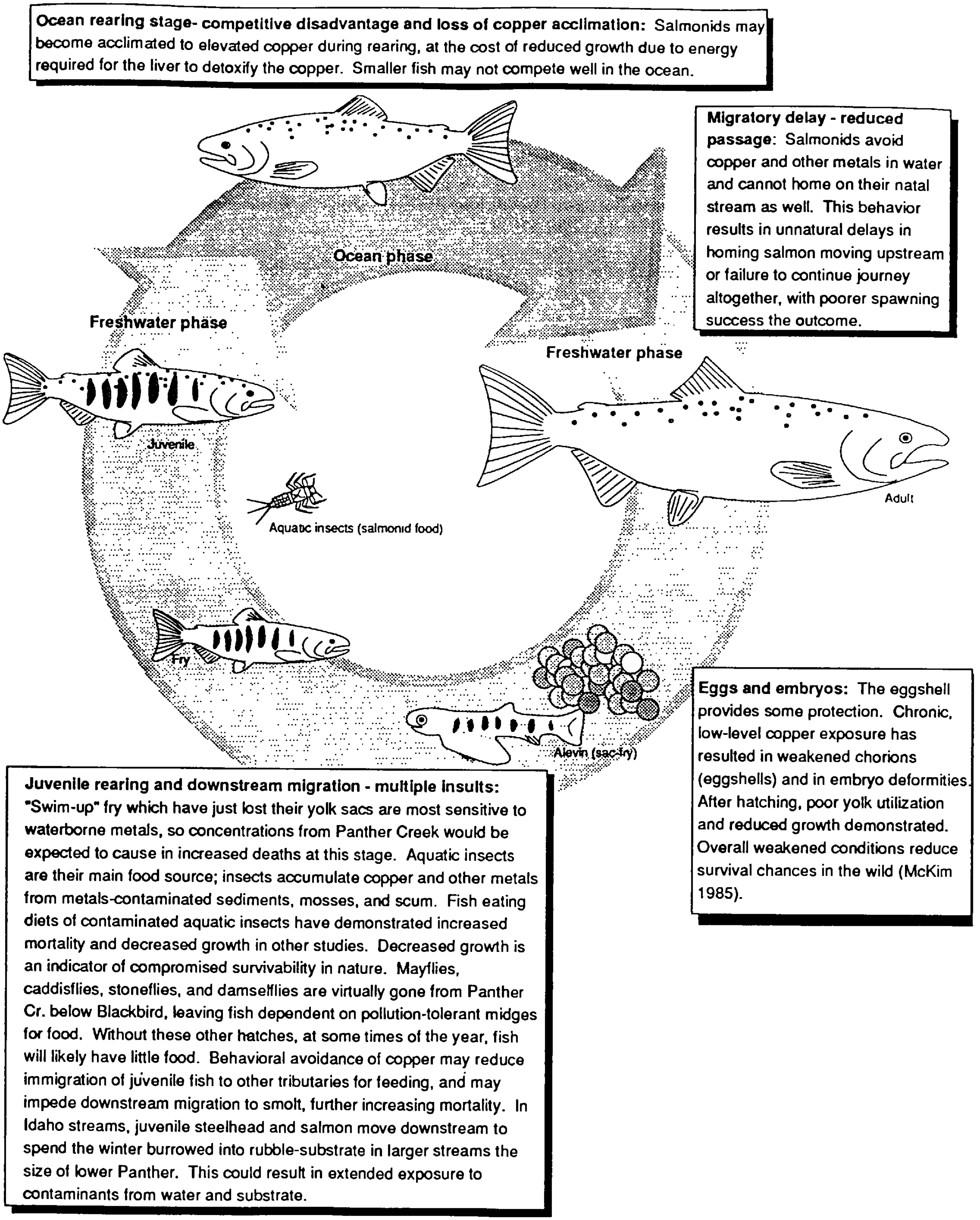

Figure 21. Risk to salmonids at different life stages in the Panther Creek system resulting from contaminants released from the Blackbird Mine (compiled from section V).

These studies clearly implicate copper and cobalt releases from the Blackbird Mine as the primary cause of the impacts to aquatic resources in the Panther Creek drainage. Additionally, studies from other areas with metals contamination strongly suggest that (1) copper contamination impedes homing and downstream migratory behavior in anadromous salmonids, and (2) metals in the food chain are an important factor in reducing the survival and growth of early-life-stage salmonids.

Limited improvement in Panther Creek water quality apparently has occurred as a possible result of water treatment of some mine discharges. Small numbers of resident trout and other salmonids are found below Blackbird Creek, and a few steelhead and chinook have returned to its lower, most diluted, reaches. However, benthic communities remain affected, and the resident fish appear to be present in reduced abundances. Continuing copper releases into Panther Creek from the site are likely to adversely affect rearing habitat and upstream and downstream migration of these fish.

Of the ten essential features of critical habitats for threatened and endangered Snake River salmon species[24], four are degraded due to Blackbird Mine discharges: (1) substrate quality, (2) water quality, (3) food, and (4) migration conditions. These features must be restored for Panther Creek drainage to once again be productive for these species.

[24] Essential features are adequate: (1) substrate; (2) water quality; (3) water quantity; (4) water temperature; (5) water velocity; (6) cover/shelter; (7) food; (8) riparian vegetation; (9) space and (10) migration conditions (Section III).

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Appendix A: Sediment sampling and bioassays, Panther Creek, Idaho

C.A. Mebane
NOAA Coastal Resources Coordination Branch
Hazardous Materials Response and Assessment Division
7600 Sand Point Way NE
Seattle, WA 98115

Abstract

Panther Creek, Idaho, receives acid mine drainage from the Blackbird mining district. A survey was made of the distribution of metals-contaminated sediments in Panther Creek and their toxicity to aquatic invertebrates . Sediment samples were collected near suspected point sources of contamination and were tested for particle size, organic carbon, bulk metals concentrations, and acute toxicity to the aquatic invertebrate *Hyalella azteca* (Amphipoda). Results showed a pattern of copper, arsenic, and cobalt enrichment that was clearly associated with the tributaries draining the mine area, Blackbird and Big Deer Creeks. Copper concentrations in sediments downstream of the tributaries were up to 300 times concentrations in sediments upstream. At the lowest station sampled, 32 km below the confluence of Blackbird Creek, the uppermost tributary, copper concentrations were still elevated over 20 times the upstream concentrations. Streambed sediment samples collected from below Blackbird and from below Big Deer Creeks were significantly toxic in controlled laboratory tests with an aquatic invertebrate ($P < .01$) compared to sediments from upstream from both creeks.

Introduction

The objectives of this present study were to (1) conduct a reconnaissance study of metals levels in streambed sediments, and (2) conduct a pilot study of bioavailability of sediment-sorbed contaminants through toxicity testing. The results were intended to help refine more comprehensive investigations that may be needed to base cleanup decisions. Trace metals have been released by hard rock mining disturbances at the headwaters of two tributaries to Panther Creek, Blackbird Creek and Big Deer Creek (Platts et al. 1979). Streambed sediments may accumulate metals from the surface waters of these tributaries by the settling of suspended particulates in quiet depositional areas of the stream and by precipitation of dissolved metals out of solution onto the bottom substrates. In addition to becoming a sink for trace metals, streambed sediments may continue to be a source of contamination to the overlying waters due to resuspension and desorption of sediment-sorbed metals back into the water column (Mok and Wai 1989). In addition to their role as a sink and source of contaminants to the overlying water, if they are bioavailable in place, contaminated sediments can have direct and indirect adverse effects on aquatic ecosystems. Contaminated bottom sediments may have direct toxic effects on bottom dwelling fauna, such as aquatic invertebrates. Contaminants may also be passed up the food chain, for example as bottom feeding aquatic invertebrates accumulate them before being eaten by trout or other predatory fish.

Adverse effects are limited by the metals bioavailability to organisms. The extent and magnitude of contamination is not necessarily biologically significant if bioavailability is limited. Although sediments may contain relatively high concentrations of toxic compounds, this presence may not cause adverse effects to sediment dwelling organisms. Bioavailability of contaminants is difficult to predict from chemical concentrations. The factors determining contaminants sorptive behavior, which affects bioavailability, are a

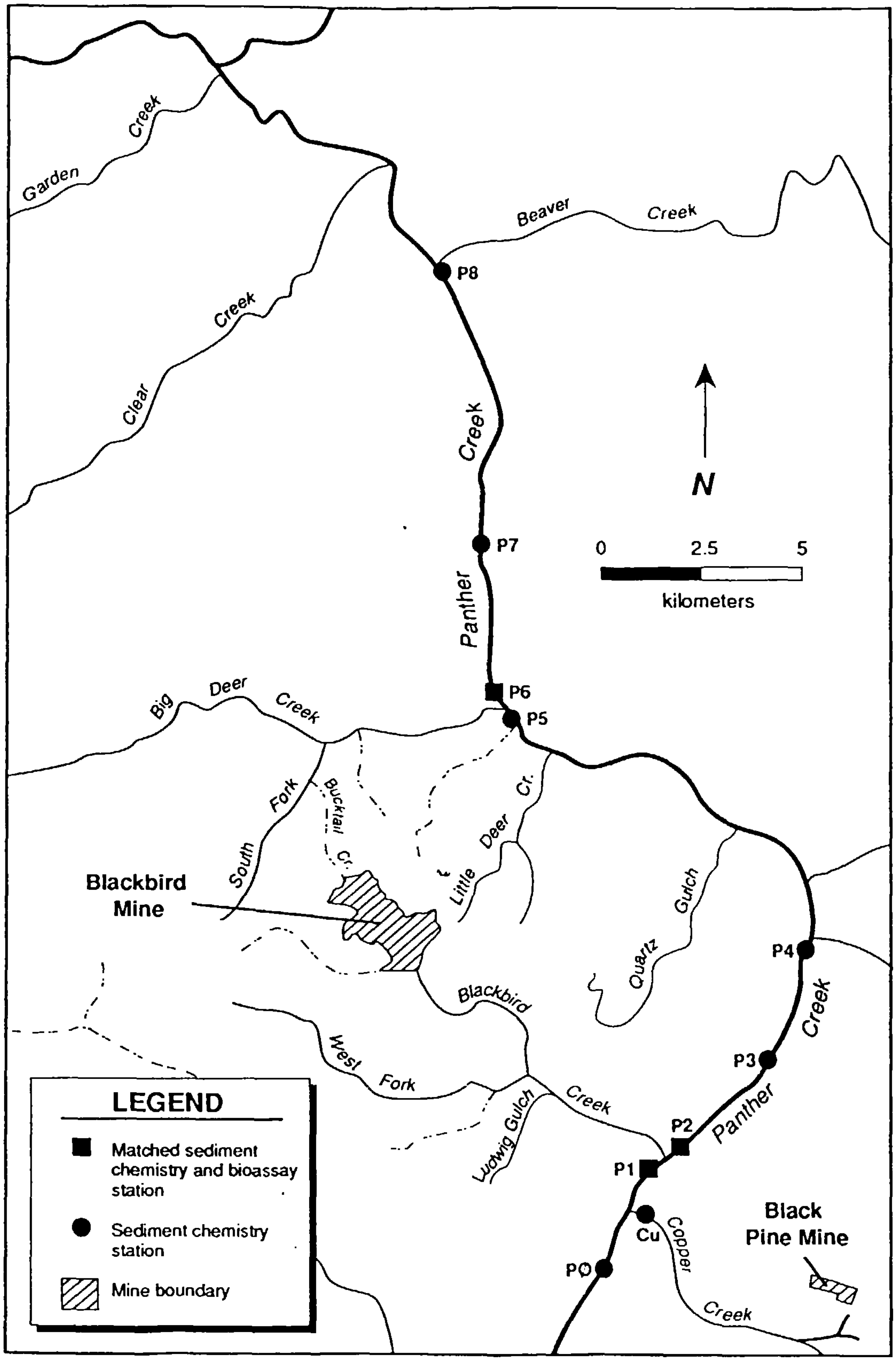

Figure A1. Panther Creek - Sediment sampling stations.

complex and poorly understood process. The only means of measuring bioavailability is by measuring and determining a biological response, such as laboratory bioaccumulation or toxicity testing (Power and Chapman 1992).

For this study, bioavailability was measured by exposing the amphipod *Hyalella azteca*, a freshwater crustacean, to sediments collected from Panther Creek in a 10 day acute toxicity test.

Methods

Summary of Sampling Stations

Attempts were made to select stations likely to represent a gradient of metals concentrations. Stations were selected by bracketing known contaminated tributaries to Panther Creek, and by a desire to provide some geographic coverage of lower Panther Creek. Sediment samples were collected from Panther Creek October 15-17, 1992. Nine Panther Creek stations between river mile 25 and river mile 5 were sampled and screened for bulk metals concentrations (Figure A1). One sample was collected from the mouth of Copper Creek, below the Blackpine Mine, a potential contamination source to Panther Creek. Three of these stations were tested for acute toxicity to an aquatic invertebrate. Table 1 and Figure 1 give approximate sampling station locations. Detailed station descriptions are in the attached chemical data report.

Table 1: Sediment sampling Stations

Station	Location	Samples per station	Toxicity Tests
P0	Panther Creek above Copper Creek	1	No
P1	Panther Creek between the confluences of Copper and Blackbird Creeks	3	Yes
P2	Panther Creek about 100m below the confluence with Blackbird Creek	3	Yes
P3	Panther Creek below the Cobalt town site	3	No
P4	Panther Creek just above the confluence of Deep Creek	3	No
P5	Panther Creek just above the confluence of Big Deer Creek	3	No
P6	Panther Creek just below the confluence of Big Deer Creek	3	Yes
P7	Panther Creek near Fritzer Gulch	3	No
P8	Panther Creek above confluence with Beaver Creek	1	No
Cu	Copper Creek at the mouth (drains the Blackpine Mine)	1	No

Sample Collection, Handling, and Storage

Stations' locations were selected to indicate Panther Creek sediment metals concentrations related to tributaries draining the Blackbird Mine (point sources) and distance and dilution by other tributaries to the Panther Creek drainage. Station locations were recorded by GPS (Global Positioning System), where not obstructed by terrain, or else by map, compass, altimeter and pacing distance from a prominent landmark. The GPS receiver, a Magellan 1000, was set to North American Datum 1927.

Sediment collection, storage, characterization, and manipulation were generally consistent with ASTM (1990). However, ASTM (1990) does not address streambed sampling from wadable streams. Instead, for this survey, sample collection methods were derived based from recommendations from Horowitz (1991), Burton (1992) and MacKnight (1991).

Where possible, stations were sampled as a cross-section transect across the stream (e.g., near the west bank, midstream, and east bank). Depositional areas were located in slow water close to the banks, in relatively slow water mid-stream (e.g., at the tails of pools upstream from rapids, and in fast water riffles and runs from eddies behind boulders mid-channel). At station P2 the bottom type (exposed bedrock, boulders and cobbles) prevented collecting sediments from a transect; there sediments were collected only from shallow areas near the bank.

Sediment samples were collected by hand from about the upper 5 cm of sediment with a stainless steel spoon and placed in a stainless steel bowl. Samples were stirred until the sediment color and texture was uniform, stones and sticks removed, and split into sample jars for toxicity and chemistry testing. A total of 2.5 liters sediment were collected for samples that would be tested for both toxicity and chemical/physical characteristics; 0.5 liters sediment were collected for samples that would be screened for physical, organic carbon, and metals concentrations only. The sampling spoon and mixing bowl were thoroughly rinsed with site water before use and between samples. Sample collection, handling, and rinsing was done so not to disturb the next sampling location.

Samples were stored and transported in 0.5 or 1.0 liter clean glass jars with Teflon lined lids. Glassware was obtained from All-World Scientific and Chemical Supply, Lynnwood, Washington. Samples were stored in the dark on ice in a cooler until shipped to the EVS Environment Consultants, Ltd., laboratories, North Vancouver, British Columbia, Canada, on October 20 by overnight air express. EVS delivered the sample splits designated for physical and chemical analyses to Analytical Services Laboratory (ASL), Vancouver on October 26. Metals analyses of the sediments were conducted by ASL on October 28. Organic carbon and particle size analyses were conducted on November 13 and December 1, 1992.

Details of laboratory methods, materials, results, and quality control procedures are included in the appended laboratory reports.

Results - Metals Enrichment

Metals concentrations in Panther Creek sediments showed increases clearly associated with inflow from Blackbird and Big Deer Creeks. Copper levels in the sediments below Blackbird Creek were up to 300 times higher than the reference stations above its confluence. Levels generally dropped with distance downstream from Blackbird Creek as Panther Creek is fed by other tributaries. The largest of which, Napias Creek, nearly doubles the volume of Panther Creek. Copper levels just upstream from Big Deer Creek were only elevated 7-10 times the concentrations at the reference stations upstream from Blackbird Creek. Copper levels below the confluence of Big Deer Creek increased by 75 to 120 times over the upstream reference levels. At the lowest station sampled, about 20 river miles below Blackbird Creek, copper levels remained enriched over twice the level upstream from Big Deer creek and 14 - 20 times the levels at the upstream reference stations. Cobalt and arsenic levels showed similar trends. There was no association found between metals levels in Panther Creek and drainage from the Blackpine Mine (Copper Creek). Table 2 and Figure 2 summarize findings. Where streambed sediments were available, samples were collected from a cross section of the stream. Results are grouped by samples collected from slow water close to banks and samples collected from pockets of sediments located midstream. Detailed results are appended in the laboratory reports.

Sediment Toxicity Testing

The sediment samples from Panther Creek above Blackbird Creek (station P1), below Blackbird Creek (station P2), and below Big Deer Creek (station P6) were tested for toxicity to the aquatic invertebrate *Hyalella azteca*. *H. azteca,* Amphipoda, is a small freshwater crustacean which has become routinely used to screen the toxicity of contaminated sediments. It is common throughout lakes and streams in North America. It has several characteristics making it desirable as a test organism including a short life cycle (3-4 weeks), widespread and abundant distribution, ecological importance, and a wide tolerance of sediment grain size. *H. azteca* is an epibenthic detritivore and will burrow in the top 1 cm of the sediment surface in search of food (ASTM 1991). *H. azteca* is considered a moderately sensitive indicator of sediment toxicity. Ten day toxicity tests were performed according to ASTM (1991). The endpoints measured were survival and growth (measured as dry weight). The negative control sediment was a clean silica aquarium sand. The samples from the stations below Blackbird and Big Deer Creeks had reduced survival compared to the reference sediments from the station above Blackbird Creek (25% and 24% versus 63% survival respectively). The differences were statistically significant at P< .01. For all composited stations, there were no significant differences with respect to growth. Table 3 summarizes the toxicity testing results. The appended laboratory report describes the toxicity testing program in more detail.

Discussion

Trace metals tend to concentrate on/in finer grained sediments, that is the clay and silt fractions, rather than on the coarser sands and gravels. The inclusion of coarser-grained sediments to a metals-rich finer grained material could be viewed as a dilution process. This inclusion of coarser dilutants can readily hide a significant metals dispersion pattern from mine effluents or other anthropogenic sources. Thus, some investigators recommend collecting only the fine grained fraction of sediments which pass through a 63 μm sieve (Håkason 1984; Horowitz 1991). However, since an objective of this study was to sample a transect of width of the stream where possible, this survey did not discriminate against including coarse grained sediments in the samples. Samples collected from the faster mid-stream waters were predominantly coarse grained sediments. In order to estimate whether this inclusion of coarse grained sediments diluted the dispersion pattern of metals in Panther Creek sediments, the percentages of the fine grained sediments were determined from separate aliquots of the bulk samples and the chemical data was normalized to them. Both the distribution of bulk sediment copper concentrations and sediment concentrations which were corrected for grain size differences are shown in Figure 2. Results of this comparison show that the influx of metals from Blackbird and Big Deer Creeks overwhelm the variability expected from the different binding capacities of fine and coarse grained sediments. Both distributions show very low copper in sediments collected from above all Blackbird mine effluents, a marked increase below Blackbird Creek, some attenuation further below Blackbird Creek as clean tributaries enter and dilute Panther Creek, and a second marked increase below Big Deer Creek. Comparing the pattern of cobalt and arsenic from bulk sediment concentrations with fines normalized concentrations, a similar pattern of very low metals above Blackbird Creek with a large increase below Blackbird exists. However, the pattern indicates that the great majority of cobalt and arsenic enrichment comes from the Blackbird Creek drainage, with little additional contribution from the Big Deer drainage.

Table 2. Concentrations of selected metals in sediment samples collected from Panther Creek in mg/kg (ppm) dry weight.

Station ID#	Copper		Arsenic		Cobalt	
	Near Bank (range)	Mid Stream	Near Bank (range)	Mid Stream	Near Bank (range)	Mid Stream
P0*	16 - -	-	7 - -	-	5 - -	-
P1	8 - 8	13	7 - 10	11	4 - 5	4
P2	2280 - 2890	-	766 - 888	-	436 - 554	-
P3	351 - 2930	149	167 - 344	82	475 - 1150	179
P4	694 - 1050	208	167 - 193	99	366 - 485	208
P5	94 - 109	65	40 - 56	27	112 - 128	108
P6	750 - 889	386	27 - 31	14	73 - 91	76
P7	1140 - 1200	135	78 - 147	25	314 - 553	63
P8*	232 - -	-	25 -	-	106 - -	-
Cu*	77 - -	-	21 -	-	8 - -	-

Notes: *Single samples were collected for Stations P0, P8 and Cu. All other stations had three samples.

Table 3. Metals concentrations in the sediments (dry weight) matched with *Hyalella* growth and survival

Station	Arsenic (mg/kg)	Cobalt (mg/kg)	Copper (mg/kg)	Iron (%)	TOC (%)	Mean survival (out of 10)	Mean dry wt. (mg)
P1	9.5	4.2	8.3	1.21	0.88	5.8 ± 2.2	0.17 ± 0.08
	6.8	4.5	8.3	1.40	0.87	6.4 ± 2.9	0.21 ± 0.07
	10.7	3.8	12.9	1.36	3.78	6.8 ± 1.8	0.18 ± 0.03
P2	766	436	2280	7.45	2.70	3.4 ± 0.9	0.17 ± 0.07
	867	507	2700	8.56	2.71	4.0 ± 2.5	0.18 ± 0.07
	888	554	2890	8.41	2.85	0.0 ± 0.0	0.0 ± 0.0
P6	14	76	386	0.93	0.12	0.0 ± 0.0	0.0 ± 0.0
	27	91	750	1.09	0.28	1.4 ± 1.3	0.07 ± 0.07
	31	74	889	1.20	0.51	5.8 ± 0.8	0.08 ± 0.04
Control	-	-	-	-	-	9.2 ± 1.3	0.15 ± 0.04

Concentrations in dry weight. For survival, a value of 10.0 represents 100% survival (10 amphipods per beaker). Mean of 5 replicates.

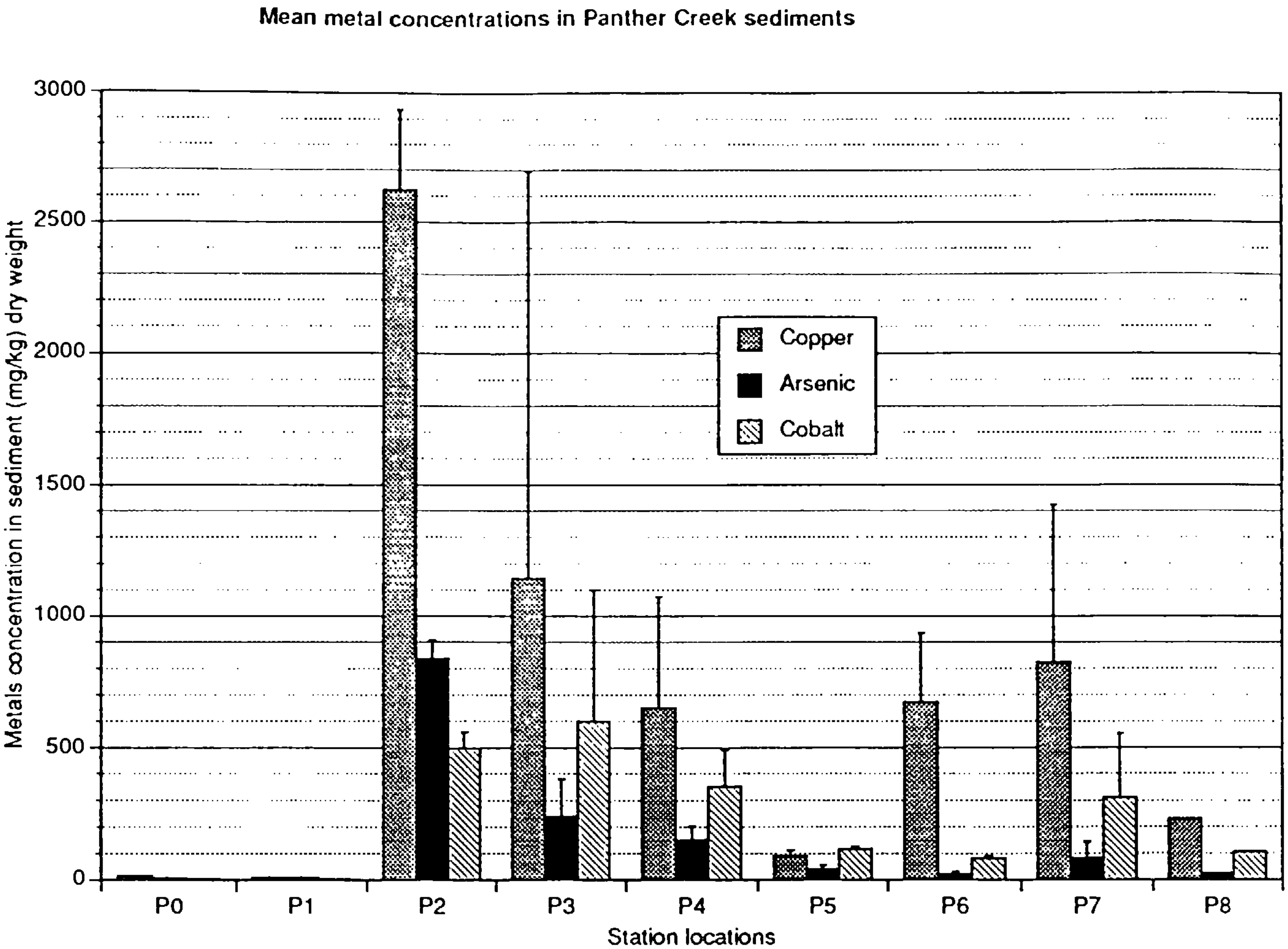

Figure A2. Concentrations of arsenic, cobalt, and copper in Panther Creek sediments. Error bars show one standard deviation; station locations are from Figure A1).

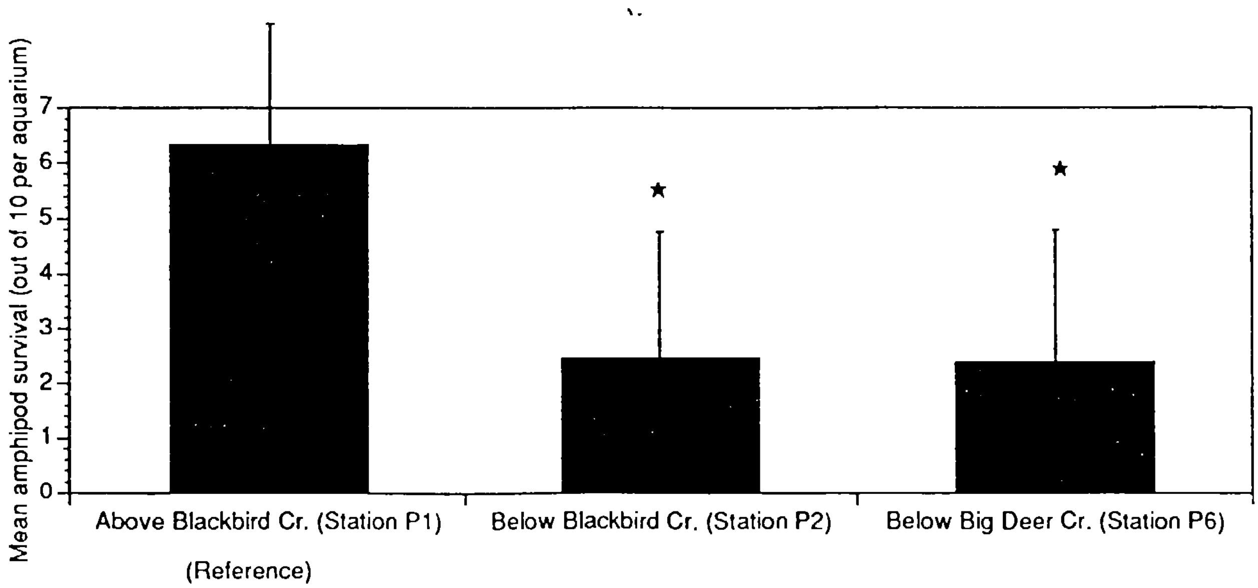

Figure A3. Mean survival of Hyalella azteca after a 10 day laboratory exposure to Panther Creek sediments. Error bars show one standard deviation; "★" indicates statistically significantly different from reference at $p < 0.01$.

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Attachments

A. Chemical laboratory report
B. Bioassay laboratory report - sediment toxicity testing program

analytical
service
laboratories
ltd.

ASL

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CHEMICAL ANALYSIS REPORT

Date: November 19, 1992

ASL File No. 6815C

Report On: Blackbird Mines PNRS
Soil Analysis

Report To: **E.V.S. Consultants Ltd.**
Suite 200
2517 East Lake Avenue
Seattle, WA 98102 USA

Attention: **Mr. Bob Stuart**

Received: October 26, 1992

ASL ANALYTICAL SERVICE LABORATORIES LTD.
per:

Joyce Chow, B.Sc.
Project Chemist

Frederick Chen, B.Sc.
Project Chemist

CELEBRATING 10 YEARS OF EXCELLENCE

Station Cu

Station Location: Mouth of Copper Creek					
Station Description: ≈3m above upstream from mouth of Panther Creek.					
taking sharp bend to the West, beneath talus slope.					
	Coordinates				
	Position Source		Landmark USGS Blackbird 7.5" topo		
Sample description (single sample):					
Sample collected from 1st small pool above confluence (beneath undercut bank) beneath thick brush.					
Sample Numbers	Cu				
Date Sampled	92 10 16				
Time	10:45				
Physical Tests					
Moisture %	72.9				
Total Metals	(mg/kg - dry weight)				
Arsenic T-As	20.8				
Cadmium T-Cd	<2.0				
Cobalt T-Co	7.5				
Copper T-Cu	76.7				
Iron T-Fe %	2.13				
Lead T-Pb	17				
Manganese T-Mn	315				
Mercury T-Hg	0.029				
Nickel T-Ni	15.3				
Silver T-Ag	<2.0				
Zinc T-Zn	85.1				
Organic Parameters					
Total Organic Carbon C %	2.95				
Particle Size					
Gravel (>2.00mm) %	11.4				
Sand (2.00mm - 0.063mm) %	52.4				
Silt (0.063mm - 4um) %	24.4				
Clay (<4um) %	11.8				
Toxicity Tests	Not tested				

Station P0

Station Location: Panther Creek above confluence Copper Creek					
Station Description: ≈0.2 km above Cobalt Ranger Station. Pool 50m upstream of Panther Creek taking sharp bend to the West, beneath talus slope.					
	Coordinates	45-03.77 N	114-16.42 W		
	Position Source		Landmark USGS Blackbird 7.5" topo		
Sample description (single sample):					
Sample taken from pool formed by large fallen tree at bottom of gravel bar, East bank.					
Sample Numbers	P0				
Date Sampled	92 10 17				
Time	14:45				
Physical Tests					
Moisture %	57.7				
Total Metals	(mg/kg - dry weight)				
Arsenic T-As	6.76				
Cadmium T-Cd	<2.0				
Cobalt T-Co	5.4				
Copper T-Cu	16.3				
Iron T-Fe %	1.72				
Lead T-Pb	11				
Manganese T-Mn	358				
Mercury T-Hg	0.027				
Nickel T-Ni	11				
Silver T-Ag	<2.0				
Zinc T-Zn	57.2				
Organic Parameters					
Total Organic Carbon C %	3.71				
Particle Size					
Gravel (>2.00mm) %	0.2				
Sand (2.00mm - 0.063mm) %	31.4				
Silt (0.063mm - 4um) %	52.1				
Clay (<4um) %	16.3				
Toxicity Tests	Not tested				

Station P1

Station Location: Panther Creek between confluence Copper and Blackbird Creeks				
Station Description: Beaver pond about 0.1 mile below Cobalt R.S.				
	Coordinates:	45-04.22 N	114-16.08 W	
	Position Source:	GPS		
Stream cross section - sample descriptions:				
	a: Near W bank at mid-section of beaver pond			
	b: Near W bank at head of beaver pond slackwater			
	c: Near E bank at mid-section of beaver pond			
Sample Numbers	P1a	P1b	P1c	PC1c Dup.
Date Sampled	92 10 17	92 10 17	92 10 17	92 10 17
Time	12:30	12:45	13:15	13:15
Physical Tests				
Moisture %	25.7	45.4	51.7	-
Total Metals	(mg/kg - dry weight)			
Arsenic T-As	9.49	6.76	10.7	10.5
Cadmium T-Cd	<2.0	<2.0	<2.0	<2.0
Cobalt T-Co	4.2	4.5	3.8	4.9
Copper T-Cu	8.3	8.3	12.9	13.1
Iron T-Fe %	1.21	1.4	1.36	1.37
Lead T-Pb	<10	<10	<10	<10
Manganese T-Mn	205	205	173	181
Mercury T-Hg	0.013	0.013	0.016	0.019
Nickel T-Ni	6.6	7.5	7.9	10.1
Silver T-Ag	<2.0	<2.0	<2.0	<2.0
Zinc T-Zn	28.9	35.7	41	41.8
Organic Parameters				
Total Organic Carbon C %	0.88	0.87	3.78	-
Particle Size				
Gravel (>2.00mm) %	33.2	5.3	3.2	-
Sand (2.00mm - 0.063mm) %	60.6	79.1	86.3	-
Silt (0.063mm - 4um) %	4.6	9.4	6.5	-
Clay (<4um) %	1.6	6.2	4	-
Toxicity Tests	(Five replicate tests per sample)			
Mean survival (out of 10)	5.8	6.4	6.8	
Standard deviation (+/-)	2.2	2.9	1.8	
Individual mean dry weight (mg)	0.17	0.21	0.18	
Standard deviation (+/-)	0.08	0.07	0.03	

Station P2

Station Location: Panther Creek below confluence Blackbird Creek				
Station Description: ≈ 100m below mouth of Blackbird Cr, by power pole across from bedrock				
outcrop.				
	Coordinates	45-04.67 N	114-15.63 W	
	Position Source	USGS Blackbird Cr		
Stream cross section - sample descriptions:				
	All three samples from lateral scour pool near west bank.			
	No sediments available midchannel/east bank for a cross section			
Sample Numbers	PC2a	PC2b	PC2c	
Date Sampled	92 10 15	92 10 15	92 10 15	
Time	15:30	16:15	17:00	
Physical Tests				
Moisture %	64.9	67	67.5	
Total Metals	(mg/kg - dry weight)			
Arsenic T-As	766	867	888	
Cadmium T-Cd	<2.0	2.4	<2.0	
Cobalt T-Co	436	507	554	
Copper T-Cu	2280	2700	2890	
Iron T-Fe %	7.45	8.56	8.41	
Lead T-Pb	<10	<10	<10	
Manganese T-Mn	284	309	316	
Mercury T-Hg	0.031	0.03	0.033	
Nickel T-Ni	45.7	51.5	53.7	
Silver T-Ag	<2.0	<2.0	<2.0	
Zinc T-Zn	63.3	65.7	70.6	
Organic Parameters				
Total Organic Carbon C %	2.7	2.71	2.85	
Particle Size				
Gravel (>2.00mm) %	0.9	0.4	0.9	
Sand (2.00mm - 0.063mm) %	17.6	16.4	27.8	
Silt (0.063mm - 4um) %	32.4	67.4	57.3	
Clay (<4um) %	49.1	15.8	14	
Toxicity Tests	(Five replicate tests per sample)			
Mean survival (out of 10)	3.4	4	0	
Standard deviation (+/-)	0.9	2.5	0	
Individual mean dry weight (mg)	0.17	0.18	0	
Standard deviation (+/-)	0.03	0.07	0	

Station P3

Station Location: Panther Creek below Cobalt townsite				
Station Description: Transect at old bridge abutment				
	Coordinates		45-06.04 N	114-13.72 W
	Position Source		Landmark - USGS Cobalt	
Stream cross section - sample descriptions:				
	a: Near E bank at pool behind E abutment			
	b: Mid-channel between abutments			
	c: Near W bank at pool behind W abutment			
Sample Numbers	PC3a	PC3b	PC3b Dup.	PC3c
Date Sampled	92 10 17	92 10 17	92 10 17	92 10 17
Time	11:00	11:20	11:20	11:45
Physical Tests				
Moisture %	85.2	26.6	-	42.9
Total Metals	(mg/kg - dry weight)			
Arsenic T-As	344	82.1	75.5	297
Cadmium T-Cd	3.5	<2.0	<2.0	<2.0
Cobalt T-Co	1150	179	177	475
Copper T-Cu	2930	149	141	351
Iron T-Fe %	5.58	1.81	1.99	2.36
Lead T-Pb	14	<10	<10	<10
Manganese T-Mn	811	316	327	208
Mercury T-Hg	0.043	0.01	0.011	0.018
Nickel T-Ni	86.6	15.1	16	39.4
Silver T-Ag	<2.0	<2.0	<2.0	<2.0
Zinc T-Zn	140	29.9	29.5	48.6
Organic Parameters				
Total Organic Carbon C %	7.36	0.16	-	1.45
Particle Size				
Gravel (>2.00mm) %	8.5	47	-	4.7
Sand (2.00mm - 0.063mm) %	35.4	51.6	-	62.8
Silt (0.063mm - 4um) %	21.9	0.4	-	23.9
Clay (<4um) %	34.2	1	-	8.6
Toxicity Tests	Not tested			

Station P4

Station Location: Panther Creek above Deep Creek confluence				
Station Description: Transect 60m below P.Cr. bridge across from campground.				
	Coordinates		45-07.55 N	114-12.41 W
	Position Source		GPS	
Stream cross section - sample descriptions:				
	a: Near West bank			
	b: Mid-channel behind a boulder			
	c: Near East bank			
Sample Numbers	PC4a	PC4b	PC4c	
Date Sampled	92 10 17	92 10 17	92 10 17	
Time	09:45	10:00	10:30	
Physical Tests				
Moisture %	60.6	23.9	52.8	
Total Metals				
Arsenic T-As	193	98.6	167	
Cadmium T-Cd	<2.0	<2.0	<2.0	
Cobalt T-Co	485	208	366	
Copper T-Cu	1050	208	694	
Iron T-Fe %	2.89	1.78	2.73	
Lead T-Pb	<10	<10	<10	
Manganese T-Mn	349	252	325	
Mercury T-Hg	0.025	0.007	0.017	
Nickel T-Ni	39	18.2	32.3	
Silver T-Ag	<2.0	<2.0	<2.0	
Zinc T-Zn	67	30.8	53.2	
Organic Parameters				
Total Organic Carbon C %	2.37	0.47	2.02	
Particle Size				
Gravel (>2.00mm) %	8.6	27.7	43	
Sand (2.00mm - 0.063mm) %	51.2	66	46.3	
Silt (0.063mm - 4um) %	26.2	4.1	6.9	
Clay (<4um) %	14	2.2	3.9	
Toxicity Tests	Not tested			

Station P5

Station Location: Panther Creek above Big Deer Creek.					
Station Description: About 50 m upstream from confluence Big Deer Creek with Panther Creek.					
	Coordinates	45-010.60 N	114-18.80 W		
	Position Source		GPS		
Stream cross section - sample descriptions:					
	a: Behind boulder mid-channel				
	b: Eddy along West bank				
	c: Near East bank				
Sample Numbers	PC5a	PC5a	PC5b	PC5c	
		Dup.			
Date Sampled	92 10 16	92 10 16	92 10 16	92 10 16	
Time	15:00	15:00	15:30	16:00	
Physical Tests					
Moisture %	24.8	-	24.5	32.7	
Total Metals	(mg/kg- dry weight)				
Arsenic T-As	27	32.7	55.7	40	
Cadmium T-Cd	<2.0	<2.0	<2.0	<2.0	
Cobalt T-Co	108	98.7	128	112	
Copper T-Cu	65.3	61.4	94.3	109	
Iron T-Fe	1.24	1.24	1.48	1.34	
Lead T-Pb	<10	<10	<10	<10	
Manganese T-Mn	234	216	276	172	
Mercury T-Hg	0.008	0.008	0.007	0.01	
Nickel T-Ni	9.4	8.4	10.5	10.7	
Silver T-Ag	<2.0	<2.0	<2.0	<2.0	
Zinc T-Zn	19.2	18.1	24.7	25.1	
Organic Parameters					
Total Organic Carbon C %	0.53	-	0.53	0.33	
Particle Size					
Gravel (>2.00mm) %	40.2	-	40.2	41.4	
Sand (2.00mm - 0.063mm) %	57.2	-	57.2	55.6	
Silt (0.063mm - 4um) %	1.5	-	1.5	1.6	
Clay (<4um) %	1.2	-	1.2	1.5	
Toxicity Tests	Not tested				

Station P6

Station Location: Panther Creek below confluence Big Deer Creek					
Station Description: Below and alongside East side of channel confluence pool below mouth of Big Deer.					
	Coordinates	45-04.22 N	114-16.08 W		
	Position Source:		GPS		
Stream cross section - sample descriptions:					
	a: Sand from behind rocks directly beneath suspended cable, near				
	center channel ≈ 70m below mouth Big Deer				
	b: Sandbank near East bank Panther Creek near tail of pool				
	c: Sandbank near East bank Panther Creek near head of pool				
Sample Numbers	PC6a	PC6b	PC6c	PC6c (Dup.)	
Date Sampled	92 10 16	92 10 16	92 10 16	92 10 16	
Time	16:30	17:00	17:30	17:30	
Physical Tests					
Moisture %	21.5	31.1	32.8	-	
Total Metals	(mg/kg - dry weight)				
Arsenic T-As	14	27.4	31.2	30	
Cadmium T-Cd	<2.0	<2.0	<2.0	<2.0	
Cobalt T-Co	76.3	91.3	73.2	75.8	
Copper T-Cu	386	750	882	896	
Iron T-Fe %	0.926	1.09	1.14	1.2	
Lead T-Pb	<10	<10	<10	<10	
Manganese T-Mn	192	160	127	138	
Mercury T-Hg	<0.005	<0.005	<0.005	<0.005	
Nickel T-Ni	5.4	6.6	7.3	7.5	
Silver T-Ag	<2.0	<2.0	<2.0	<2.0	
Zinc T-Zn	17.5	24.7	28.4	29.8	
Organic Parameters					
Total Organic Carbon C %	0.12	0.28	0.51	-	
Particle Size					
Gravel (>2.00mm) %	18.7	6.8	4.1	-	
Sand (2.00mm - 0.063mm) %	78.4	87	89.2	-	
Silt (0.063mm - 4um) %	2.2	4.5	5.5	-	
Clay (<4um) %	0.7	1.8	1.2	-	
Toxicity Tests	(Five replicate tests per sample)				
Mean survival (out of 10)	0	1.4	5.8	-	
Standard deviation (+/-)	0	1.3	0.8	-	
Individual mean dry weight (mg)	0	0.007	0.008	-	
Standard deviation (+/-)	0	0.07	0.04	-	

Station P7

Station Location: Panther Creek above Fritzer Gulch					
Station Description: Adjacent to Frizter Flat at USFS baseline temp/benthos/WQ conventionals station.					
Two chinook salmon redds at station					
	Coordinates	45-11.98 N	114-18.98 W		
	Position Source		GPS		
Stream cross section - sample descriptions:					
	a: Near east bank immediately adjacent to USFS monitoring statior				
	b: Near West bank				
	c: Behind boulder mid-stream ≈ 7m below salmon redds				
Sample Numbers	PC7a	PC7b	PC7c		
Date Sampled	92 10 16	92 10 16	92 10 16		
Time	13:30	13:45	14:15		
Physical Tests					
Moisture %	63.7	80	21.4		
Total Metals	(mg/kg -dry wight)				
Arsenic T-As	78.2	147	25		
Cadmium T-Cd	<2.0	<2.0	<2.0		
Cobalt T-Co	314	553	62.9		
Copper T-Cu	1200	1140	135		
Iron T-Fe %	3.21	3.48	1.1		
Lead T-Pb	11	13	<10		
Manganese T-Mn	282	712	175		
Mercury T-Hg	0.04	0.073	0.007		
Nickel T-Ni	29.3	35.2	6.2		
Silver T-Ag	<2.0	<2.0	<2.0		
Zinc T-Zn	68.4	78.1	16.5		
Organic Parameters					
Total Organic Carbon C %	2.46	5.54	0.39		
Particle Size					
Gravel (>2.00mm) %	8.5	11.1	63.7		
Sand (2.00mm - 0.063mm) %	44.8	31.6	35.3		
Silt (0.063mm - 4um) %	29.9	35.9	0.4		
Clay (<4um) %	16.7	21.4	0.7		
Toxicity Tests	Not tested				

Station P8

Station Location: Panther Creek above confluence Beaver Creek				
Station Description: 30m below bridge over Panther Creek				
	Coordinates	45-16.42 N	114-20.01 W	
	Position Source		GPS	
Sample description (single sample):				
About 30m below bridge over Panther Creek, sand bar near east bank.				
Sample Numbers	PC8			
Date Sampled	92 10 16			
Time	12:30			
Physical Tests				
Moisture %	28.2			
Total Metals	(mg/kg - dry weight)			
Arsenic T-As	24.6			
Cadmium T-Cd	<2.0			
Cobalt T-Co	106			
Copper T-Cu	232			
Iron T-Fe %	1.62			
Lead T-Pb	<10			
Manganese T-Mn	151			
Mercury T-Hg	0.016			
Nickel T-Ni	10.8			
Silver T-Ag	<2.0			
Zinc T-Zn	27.9			
Organic Parameters				
Total Organic Carbon C %	0.31			
Particle Size				
Gravel (>2.00mm) %	28.1			
Sand (2.00mm - 0.063mm) %	66.3			
Silt (0.063mm - 4um) %	3.5			
Clay (<4um) %	2.1			
Toxicity Tests	Not tested			

Samples were analyzed by methods acceptable to the appropriate regulatory agency. Outlines of the methodologies utilized are as follows:

Moisture

This analysis is carried out gravimetrically by drying the sample to constant weight at 103 C.

Metals in Sediment/Soil

These analyses are carried out using procedures that are consistent with the requirements of the appropriate regulatory agencies and adapted from U.S. EPA Method 3050 (Publ. # SW-846, 3rd ed., Washington, DC 20460). The procedures involve a digestion using a combination of nitric and hydrochloric acids. The resulting extract is bulked to volume with deionized/distilled water. The digested portion is then analysed by a variety of instrumental techniques, which may include specific atomic absorption spectrophotometric techniques (AAS) and/or atomic emission spectrophotometry (ICP), to obtain the required detection limit for each element. Specific details are available upon request.

PLEASE NOTE (When the following elements are reported):

Aluminum, barium, calcium, chromium, iron, magnesium, manganese, molybdenum and vanadium are often associated with the silicate matrix of the sediment. Because of this, the recoveries of these elements may be low using the specified digestion. From an environmental standpoint, this is not usually of concern since the "available" metals are typically the fraction of interest.

Total Organic Carbon in Sediment/Soil

This analysis is carried out in accordance with U.S. EPA Method 9060A (Publ. # SW-846 3rd ed., Washington, DC 20460). The procedure involves a carbonate analysis (Leco gasometer) and a total carbon analysis (Leco induction furnace). The difference in carbon values is reported as Total Organic Carbon.

Sediment/Soil Particle Size Distribution

This analysis is carried out using a method adapted for Fisheries and Environment Canada, Ottawa, described in Walton 1978. The procedure involves oven-drying prior to using standard sieves for the sand and silt fractions and the pipette method for the clay fraction.

APPENDIX 1

QUALITY ASSURANCE / QUALITY CONTROL

An extensive quality assurance/quality control program is routinely incorporated with the sample analysis. This program includes the analysis of quality control samples to define precision and accuracy, and to demonstrate contamination control for the type of samples and parameters under investigation. Quality control samples may include method blanks, sample duplicates, standard reference materials (SRM), and analyte or surrogate spikes. For this project, the following quality control analyses were carried out:

- Method blanks (i.e. digestion blanks, extraction blanks, etc.);
- Reagent blanks (ie. calibration blanks);
- Reagent spikes (ie. calibration standards);
- Matrix spike (spike of standard into sample);
- Sample duplicates (n = 4);
- MESS-1, Marine Sediment SRM (National Research Council of Canada), certified for metals;
- BCSS-1, Marine Sediment SRM (National Research Council of Canada), certified for metals;
- BEST-1, Marine Sediment SRM (National Research Council of Canada), certified for mercury;
- PACS-1, Marine Sediment SRM (National Research Council of Canada), certified for metals.

There were no anomalies or analytical problems involving instrumentation or sample handling. The sample duplicate data is reported in the results tables. The quality control data demonstrates that precision, accuracy, and contamination control met acceptance criteria for all parameters analyzed.

RESULTS OF ANALYSIS - QA Data

File No. 6815C

		Method Blank 1	Method Blank 2	BCSS-1	BCSS-1 Certified Range
		92 10 27	92 10 27	92 10 27	
Total Metals					
Arsenic	T-As	<0.05	<0.05	9.72	11.1 ± 1.4
Cadmium	T-Cd	<2.0	<2.0	<2.0	0.25 ± 0.04
Cobalt	T-Co	<2.0	<2.0	11.8	11.4 ± 2.1
Copper	T-Cu	<1.0	<1.0	16.8	18.5 ± 2.7
Iron	T-Fe	<0.005	<0.005	3.20	-
Lead	T-Pb	<10	<10	21	22.7 ± 3.4
Manganese	T-Mn	<1.0	<1.0	208	229 ± 15
Mercury	T-Hg	<0.005	<0.005	-	-
Nickel	T-Ni	<2.0	<2.0	57.2	55.3 ± 3.6
Silver	T-Ag	<2.0	<2.0	<2.0	-
Zinc	T-Zn	<1.0	<1.0	116	119 ± 12

Remarks regarding the analyses appear at the beginning of this report.
Results are expressed as milligrams per dry kilogram except where noted.
< = Less than the detection limit indicated.
Low recoveries are expected due to silicate matrix association.

RESULTS OF ANALYSIS - QA Data File No. 6815C

		MESS-1 92 10 27	MESS-1 Certfied Range	PACS-1 92 10 27	PACS-1 Certified Range
Total Metals					
Arsenic	T-As	9.48	10.6 ± 1.2	208	211 ± 11
Cadmium	T-Cd	<2.0	0.59 ± 0.10	<2.0	2.38 ± 0.20
Cobalt	T-Co	12.4	10.8 ± 1.9	18.6	17.5 ± 1.1
Copper	T-Cu	24.1	25.1 ± 3.8	462	452 ± 16
Iron	T-Fe' %	2.85	-	4.71	-
Lead	T-Pb	29	34.0 ± 6.1	404	404 ± 20
Manganese	T-Mn'	410	513 ± 25	367	470 ± 12
Mercury	T-Hg	-	-	4.68	4.57 ± 0.16
Nickel	T-Ni	27.7	29.5 ± 2.7	44.2	44.1 ± 2.0
Silver	T-Ag	<2.0	-	<2.0	-
Zinc	T-Zn	197	191 ± 17	840	824 ± 22

	BEST-1 92 10 27	BEST-1 Certfied Range
Total Metals		
Mercury	0.092	0.092 ± 0.009

Remarks regarding the analyses appear at the beginning of this report.
Results are expressed as milligrams per dry kilogram except where noted.
< = Less than the detection limit indicated
' Low recoveries are expected due to silicate matrix association.

RESULTS OF ANALYSIS - QA Data File No. 6815C

		Reagent Blank	Reagent[1] Spike	Matrix[2] Spike
		92 10 28	92 10 28	92 10 28
Physical Tests				
Moisture	%	-	-	-
Total Metals				
Arsenic	T-As	<0.05	108	95.0
Cadmium	T-Cd	<2.0	99.5	-
Cobalt	T-Co	<2.0	101	-
Copper	T-Cu	<1.0	99.1	-
Iron	T-Fe	<0.005	98.5	-
Lead	T-Pb	<10	101	-
Manganese	T-Mn	<1.0	101	-
Mercury	T-Hg	<0.005	95.2	-
Nickel	T-Ni	<2.0	99.5	-
Silver	T-Ag	<2.0	-	-
Zinc	T-Zn	<1.0	100	-

Remarks regarding the analyses appear at the beginning of this report.
Results are expressed as milligrams per dry kilogram except where noted.
< = Less than the detection limit indicated.
[1] Results expressed as percent recovery.
[2] Results expressed as percent recovery.

APPENDIX 2

DETECTION LIMITS

RESULTS OF ANALYSIS - QA Data File No. 6815C

			Detection Limits
Physical Tests			
Moisture		%	0.01
Total Metals			
Arsenic	T-As		0.05
Cadmium	T-Cd		2.0
Cobalt	T-Co		2.0
Copper	T-Cu		1.0
Iron	T-Fe	%	0.005
Lead	T-Pb		10
Manganese	T-Mn		1.0
Mercury	T-Hg		0.005
Nickel	T-Ni		2.0
Silver	T-Ag		2.0
Zinc	T-Zn		1.0
Organic Parameters			
Total Organic Carbon	C	%	0.01
Particle Size			
Gravel	(>2.00mm)	%	0.1
Sand	(2.00mm - 0.063mm)	%	0.1
Silt	(0.063mm - 4um)	%	0.1
Clay	(<4um)	%	0.1

Remarks regarding the analyses appear at the beginning of this report.
Results are expressed as milligrams per dry kilogram except where noted.
< = Less than the detection limit indicated.
Results expressed as percent recovery.
Results expressed as percent recovery.

LABORATORY REPORT

SEDIMENT TOXICITY TESTING PROGRAM

Prepared for:	NOAA Environmental Protection Agency (HW 113) 1200 6th Avenue Seattle, WA 98101
Prepared by:	EVS CONSULTANTS 2517 Eastlake Avenue, East Suite 200 Seattle, WA 98102
EVS Project No:	9/575-05.1

November 1992

Environmental Services for
Industry and Government

Seattle, U.S.A.
Vancouver, Canada

ENVIRONMENT
CONSULTANTS

Our File: 9/575-05.1

November 23, 1992

Mr. Chris Mebane
Environmental Protection Agency (HW-113)
1200 6th Avenue
Seattle, WA
98101

Dear Mr. Mebane:

<u>**Re: Freshwater Sediment Toxicity Tests**</u>

We have completed toxicity testing on nine (9) freshwater sediment samples, received on October 21, 1992, using the freshwater amphipod *Hyalella azteca*. Ten-day sediment toxicity tests were performed according to ASTM (1991). The endpoints measured were survival and growth (measured as dry weight). All samples, except sample PB1b, were significantly different from the control with respect to survival. For all samples there were no significant differences with respect to growth.

If you have any questions or require additional information, please do not hesitate to contact the undersigned at (604) 986-4331.

Yours truly,

EVS CONSULTANTS

Paul Kennedy
Paul Kennedy, B.Sc.
Biologist

G Reddy
Gonum Reddy, M.Sc.
QA/QC Officer

PAK/ubl

2517 Eastlake Ave. East
Suite 200
Seattle, WA 98102
Tel [illegible]
Fax [illegible]

195 Pemberton Avenue
North Vancouver, B.C.
Canada V7P 2R4
Tel [illegible]
Fax [illegible]

INTRODUCTION

Nine sediments identified as PB1a, PB1b, PB1c, PB2a, PB2b, PB2c, PB6a, PB6b, and PB6c were submitted to the EVS Consultants laboratory for toxicity testing using *Hyalella azteca*. Samples PB2(a-c), PB6(a-c) and PB1(a-c) were collected on October 15, 16 and 17, 1992, respectively. The samples were collected in 1-L glass containers, shipped by overnight courier and received at the laboratory October 21, 1992. All samples were stored at 4°C in the dark until testing was initiated, and leftover samples were held until the tests were completed. Tests were conducted following procedures described below.

TEST METHODS

Sediment toxicity tests using the juvenile freshwater amphipod, *Hyalella azteca*, were conducted using a test procedure based on ASTM (1991). Moderately hard water (EPA, 1989) was used for culture and testing. Reagent-grade chemicals were added to dechlorinated water to achieve a final hardness of 80-100 mg/L as $CaCO_3$. The tests were conducted in a constant environment room at 25 ± 1°C under a 16:8 h light:dark photoperiod.

Test organisms were obtained from an in-house laboratory culture. One week before testing was initiated juvenile *Hyalella* (1-mm ≤ long) were removed from the culture tank and isolated in a 13-L aquarium. The aquarium was filled with 10-L of reconstituted dechlorinated water. Two squares of rinsed cotton gauze and 2-3 pre-soaked dried maple leaves were added. The amphipods were fed 50 mL of *Selenastrum* and 50 mL of d-YCTF (yeast/cereal flakes/digested trout food) every second day.

Acute lethality of the whole fresh (unfrozen) sediments involved a 10-d exposure of juvenile *Hyalella azteca* (< 10-d old) to the test sediment. The sediments were prepared the day before testing was initiated. Large particles (rocks, etc.) and large worms were removed from the test sediment, which was then homogenized by thorough hand mixing. A 2-cm layer of test sediment was placed in 1-L glass jars and covered with 800 mL of culture water. The weight of sediment required to make a 2-cm layer was recorded for the first replicate. That weight of sediment was then added to the remaining five replicates for that sample. The negative control consisted of clean silica sand. The jars were covered with clean plastic lids and gently aerated overnight to allow the sediments to settle and allow any contaminants in the sediments to come to near equilibrium with the water. Five replicates were run per sample with an additional sixth jar, void of amphipods, serving as a reference for daily measurement of water chemistry (pH, dissolved oxygen, conductivity and temperature).

The next day (Day 0), each jar was randomly seeded with 10 amphipods. The juvenile *Hyalella* were gently poured into a glass Pyrex dish and counted into weighboats using an inverted Pasteur pipette. The animals were introduced into the jar by pipette, making sure they were released below the water surface to avoid being trapped by surface tension. Any floating amphipods were carefully resubmerged. On Day 0 each test jar received 6 mL of a 50:50 mixture of *Selenastrum*:d-YCTF and 0.2 mL of prepared Tetramin food (20 g of Tetramin fish flakes blended in 200 mL of culture water). Every second day thereafter test jars were fed 4 mL of the 50:50 mixture and 0.2 mL of prepared Tetramin.

The test was terminated after 10 d when the sediments were sieved and live and dead amphipods were removed and counted. Amphipods were considered dead when there was no response to physical stimulation. Missing amphipods were assumed to have died and decomposed prior to the termination of the test. Surviving *Hyalella* from each replicate were placed in pre-weighed aluminum weighing pans and dried at 60°C for 24 h. The pans were then weighed to obtain the total dry weight for each replicate. Mean individual dry weights were obtained by dividing the final dry weight of each replicate by the number of survivors. Statistical analyses were performed using the STATISTIX computer program (NH Analytical Software, 1986). Two-sample *t*-tests were performed, comparing each test sediment to the control sediment, to determine whether there were significant differences ($P < 0.05$) in survival or growth. Statistical analyses of growth data was not performed on samples with survival values significantly lower than the control sediment.

A positive (toxic) control was conducted using the reference toxicant, zinc sulfate ($ZnSO_4$) expressed as Zn. A 48-h LC50 test (no sediment) was conducted to measure the sensitivity of the amphipods used. Eight concentrations of zinc (10, 32, 56, 100, 180, 320, 560, and 1000 μg/L), plus a negative control, were prepared. The tests were conducted in 150-mL beakers containing 100 mL of test solution and 10 amphipods. Water quality measurements were recorded at 0 and 48 h. Mortalities were recorded at 48 h. The 48 h LC50 value was calculated using a computer program which used either the binomial, moving average or probit method, depending on the suitability of the data.

RESULTS

Test results are summarized in Table 1. Mean survival in the test sediments ranged from 0.0 out of 10 (0%) in Sample PB2c and PB6a to 6.8 out of 10 (63%) in Sample PB1c. Mean survival in the control was 9.2 out of 10.0 (92%). All samples, except PB1b were significantly different ($P<0.05$) from the control sediment with respect to survival. Statistical analyses were not performed on Samples PB2c and PB6a as there was zero survival in those samples. Mean individual dry weight ranged from 0.07 mg to 0.21 mg and was 0.15 mg in the control. None of the test sediments had mean individual dry weight values significantly different ($P<0.05$) from the control sediment. The 48-h LC50 for the zinc sulfate reference toxicant was 167 μg/L Zn. This value was within the acceptable range of 133.1 ± 42.8 [mean ± 2SD] obtained by this laboratory in previous testing.

Water quality parameters measured during the 10-d exposure period were in the following ranges: temperature, 23.5 - 26°C; dissolved oxygen, 6.0 - 8.4 mg/L; conductivity, 220 - 450 μmhos/cm; pH, 7.4 - 8.4. All water quality parameters were within acceptable ranges for the culture of *Hyalella*.

REFERENCES

ASTM. 1991. Standard guide for conducting sediment toxicity tests with freshwater invertebrates. Method E1383-90. In: Annual Book of ASTM Standards, Volume 11.04. pp. 1085-1104. American Society for Testing and Materials, Philadelphia, Pennsylvania.

EPA. 1989. Short-term methods for estimating the chronic toxicity of effluents and receiving waters to freshwater organisms, 2nd edition, US Environmental Protection Agency, Cincinnati, Ohio. EPA/600/4-89/001.

NH Analytical Software. 1986. STATISTIX: An interactive statistics program for microcomputers. NH Analytical Software, St. Paul, Minnesota. IBM Version 1.1.

Table 1. Summary of *Hyalella azteca* sediment toxicity test results.

Sample I.D.	Mean Survival[1]	Individual Mean Dry Weight (mg)[2]
PB1a	5.8 ± 2.2*	0.17 ± 0.08
PB1b	6.4 ± 2.9	0.21 ± 0.07
PB1c	6.8 ± 1.8*	0.18 ± 0.03
PB2a	3.4 ± 0.9*	0.17 ± 0.03
PB2b	4.0 ± 2.5*	0.18 ± 0.07
PB2c	0.0 ± 0.0*	0.0 ± 0.0
PB6a	0.0 ± 0.0*	0.0 ± 0.0
PB6b	1.4 ± 1.3*	0.07 ± 0.07
PB6c	5.8 ± 0.8*	0.08 ± 0.04
Control Sediment	9.2 ± 1.3	0.15 ± 0.04

1 n = 10; for survival, a value of 10.0 represents 100% survival. Dry weights represent mean individual dry weights of surviving organisms. Asterisks denote values significantly different ($P<0.05$) from the control sediment.

2 Statistical analysis of dry weight data was not performed for samples with survival values significantly lower than the control sediment.

AMPHIPOD TOXICITY TEST DATA

EVS CONSULTANTS

Dry Weight Data

Client:	NOAA	Test Species:	Hyalella azteca
Project #:	9575-05.1	Date Initiated:	October 23/1992
Work Order:	920433	Date Terminated:	November 2/1992
Test Type:	10 day growth		
		Number of Test Organisms:	10

Sample ID	Rep	Survivors	Pan Weight (g)	Final Weight (pan + biomass) (g)	Total Biomass (mg)	Individual Biomass (mg)	Mean Survival (%)	Mean Individual Biomass (mg)
Day 0	A	10	0.9888	0.9907	1.9	0.19	100.0	0.12
	B	10	0.9862	0.9870	0.8	0.08		
	C	10	0.9806	0.9821	1.5	0.15		
	D	10	0.9737	0.9747	1.0	0.10		
	E	10	0.9786	0.9793	0.7	0.07		

EVS CONSULTANTS

Dry Weight Data

Client:	NOAA	Test Species	*Hyalella azteca*
Project #:	2575-05.1	Date Initiated.	October 23/1992
Work Order	920433	Date Terminated.	November 2/1992
Test Type	10 day growth		
		Number of Test Organisms:	10

Sample ID	Rep	Survivors	Pan Weight (g)	Final Weight (pan + biomass) (g)	Total Biomass (mg)	Individual Biomass (mg)	Mean Survival (%)	Mean Individual Biomass (mg)
Control	A	10	0.9816	0.9836	2.0	0.20	92.0	0.15
	B	10	0.9896	0.9910	1.4	0.14		
	C	7	0.9856	0.9868	1.2	0.17		
	D	9	0.9814	0.9828	1.4	0.16		
	E	10	0.9818	0.9827	0.9	0.09		
FB1a	A	9	0.9885	0.9894	0.9	0.10	58.0	0.17
	B	4	0.9711	0.9714	0.3	0.08		
	C	7	0.9714	0.9726	1.2	0.17		
	D	5	0.9757	0.9768	1.1	0.22		
	E	4	0.9707	0.9718	1.1	0.27		
FB1b	A	2	0.9887	0.9893	0.6	0.30	64.0	0.21
	B	9	0.9910	0.9927	1.7	0.19		
	C	6	0.9930	0.9946	1.6	0.27		
	D	6	0.9817	0.9827	1.0	0.17		
	E	9	0.9868	0.9880	1.2	0.13		
PB1c	A	6	0.9869	0.9883	1.4	0.23	68.0	0.18
	B	8	0.9883	0.9897	1.4	0.18		
	C	8	0.9890	0.9901	1.1	0.14		
	D	4	0.9887	0.9894	0.7	0.17		
	E	8	0.9864	0.9878	1.4	0.17		
PB2a	A	3	0.9906	0.9910	0.4	0.13	34.0	0.17
	B	5	0.9896	0.9904	0.8	0.16		
	C	3	0.9822	0.9828	0.6	0.20		
	D	3	0.9903	0.9909	0.6	0.20		
	E	3	0.9913	0.9918	0.5	0.17		

EVS CONSULTANTS

Dry Weight Data

Client:	NOAA	Test Species:	Hyalella azteca
Project #:	9/575-05.1	Date Initiated:	October 23/1992
Work Order:	920433	Date Terminated.	November 2/1992
Test Type:	10 day growth		

Number of Test Organisms: 10

Sample ID	Rep	Survivors	Pan Weight (g)	Final Weight (pan + biomass) (g)	Total Biomass (mg)	Individual Biomass (mg)	Mean Survival (%)	Mean Individual Biomass (mg)
FB2b	A	3	0.9847	0.9851	0.4	0.13	40.0	0.13
	B	8	0.9890	0.9895	0.5	0.06		
	C	2	0.9936	0.9941	0.5	0.25		
	D	5	0.9938	0.9943	0.5	0.10		
	E	2	0.9961	0.9963	0.2	0.10		
PB2c	A	0					0.0	0.00
	B	0						
	C	0						
	D	0						
	E	0						
PB6a	A	0					0.0	0.00
	B	0						
	C	0						
	D	0						
	E	0						
PB6b	A	3	0.9700	0.9703	0.3	0.10	23.3	0.12
	B	2	0.9683	0.9685	0.2	0.10		
	C	2	0.9733	0.9736	0.3	0.15		
	D	0						
	E	0						
PB6c	A	6	0.9938	0.9946	0.8	0.13	58.0	0.08
	B	6	0.9663	0.9667	0.4	0.07		
	C	5	0.9661	0.9666	0.5	0.10		
	D	5	0.9668	0.9671	0.3	0.06		
	E	7	0.9734	0.9737	0.3	0.04		

Client:	NOAA
Project #:	9/575-05.1
Statistic:	Hyalella Dry Weight
WO #:	920433

VIEW DATA

CASE	CONTROL	PB1A	PB1B	PB1C	PB2A	PB2B
1	0.2000	0.1000	0.3000	0.2300	0.1300	0.1300
2	0.1400	0.0800	0.1900	0.1800	0.1600	0.0600
3	0.1700	0.1700	0.2700	0.1400	0.2000	0.2500
4	0.1600	0.2200	0.1700	0.1700	0.2000	0.1000
5	0.0900	0.2700	0.1300	0.1700	0.1700	0.1000

CASE	PB2C	PB6A	PB6B	PB6C
1	0.0000	0.0000	0.1000	0.1300
2	0.0000	0.0000	0.1000	0.0700
3	0.0000	0.0000	0.1500	0.1000
4	0.0000	0.0000	0.0000	0.0600
5	0.0000	0.0000	0.0000	0.0400

DESCRIPTIVE STATISTICS

VARIABLE	MEAN	S.D.	N	MEDIAN	MINIMUM	MAXIMUM
CONTROL	1.520E-01	4.087E-02	5	1.600E-01	9.000E-02	2.000E-01
PB1A	1.680E-01	7.981E-02	5	1.700E-01	8.000E-02	2.700E-01
PB1B	2.120E-01	7.085E-02	5	1.900E-01	1.300E-01	3.000E-01
PB1C	1.780E-01	3.271E-02	5	1.700E-01	1.400E-01	2.300E-01
PB2A	1.720E-01	2.950E-02	5	1.700E-01	1.300E-01	2.000E-01
PB2B	1.280E-01	7.259E-02	5	1.000E-01	6.000E-02	2.500E-01
PB6B	7.000E-02	6.708E-02	5	1.000E-01	0.000	1.500E-01
PB6C	8.000E-02	3.536E-02	5	7.000E-02	4.000E-02	1.300E-01
PB2C	0.000	0.000	5	0.000	0.000	0.000
PB6A	0.000	0.000	5	0.000	0.000	0.000

Amphipod survival Kruskal-Wallis non-parametric ANOVA and Tukey type multiple comparison test

Station P1		Station P2		Station P6			
Survival(out of 10)	Rank	Survival	Rank	Survival	Rank		
9	44.0	8	40.5	7	37.5		
9	44.0	5	29.0	6	34.0		
9	44.0	5	29.0	6	34.0		
8	40.5	3	20.5	5	29.0		
8	40.5	3	20.5	5	29.0		
8	40.5	3	20.5	3	20.5		
7	37.5	3	20.5	2	15.0		
6	34.0	3	20.5	2	15.0		
6	34.0	2	15.0	0	6.0		
6	34.0	2	15.0	0	6.0		
5	29.0	0	6.0	0	6.0		
4	25.0	0	6.0	0	6.0		
4	25.0	0	6.0	0	6.0		
4	25.0	0	6.0	0	6.0		
2	15.0	0	6.0	0	6.0		
n=	15	n=	15	n=	15	N=45	
m of ranks by group R1	512	R2	261	R3	256		
H statistic =	14.97		Numerator degrees of freedom = m-1.				
SE=	50.87		Denominator degrees of freedom = m(n-1)				
			n=15, m=3				
H critical value	5.99	(P<.05)	Ho: Amphipod survival is the same in all three groups				
(from table)	9.21	(P<.01)	Ha: Amphipod survival is not the same in all three groups				
			Reject Ho		(P<.01)		
Samples by rank sum	1	2	3				
	512	261	256				
SE (error mean square)	50.87						
q=(Xb-Xa)/SE							
error degrees of freedom (DF) k(n-1) = 42 (k=m=number of groups)							
Comparison	Difference (Rb-Ra)	SE	q	q (0.01, 42, 3) critical values from Zar table B.5			
1 vs. 3	256.00	50.87	5.03	4.367 - Conclusion: Reject Ho $\mu1=\mu3$			
1 vs. 2	251.00	50.87	4.93	4.367 - Conclusion: Reject Ho $\mu1=\mu2$			
2 vs. 3	5.00	50.87	0.10	4.367 - Conclusion: Accept Ho $\mu2=\mu3$			
Table values from: Zar, J.H. 1984. Biostatistical analysis (2nd ed.) Prentice Hall, Englewood Cliffs, NJ							

Amphipod survival ANOVA and Tukey's multiple means comparison test

	Station P1	Station P2	Station P6	
Station sample #				
a	9	3	0	(Number of animals surviving in each beaker out of 10 on Day 10 of the test)
	4	5	0	
	7	3	0	
	5	3	0	
	4	3	0	
b	2	3	3	
	9	8	2	
	6	2	2	
	6	5	0	
	9	2	0	
c	6	0	6	
	8	0	6	
	8	0	5	
	4	0	5	
	8	0	7	
Mean survival	6.33	2.47	2.40	
Std. dev. (+/-)	2.19	2.33	2.69	
Std dev. means	2.25			
var within groups	5.83			
var between	76.07			
F statistic	13.06			
n=15				Numerator degrees of freedom = m-1.
m=3				Denominator degrees of freedom = m(n-1)
F critical values	3.22	(P<.05)		
(from table)	5.15	(P<.01)		
Means ranked in order of magnitude	1	2	3	Table values from: Zar, J.H. 1984. Biostatistical analysis (2nd ed.) Prentice Hall, Englewood Cliffs, NJ
	2.40	2.47	6.33	
SE (error mean square)	0.62			q=(Xb-Xa)/SE
				error degrees of freedom (DF) k(n-1) = 42 (k=m=number of groups)
Comparison	Difference (Xb-Xa)	SE	q	q (0.01, 40, 3) critical values from Zar table B.5
3 vs. 1	3.93	0.62	6.31	4.367 - Conclusion: Reject Ho $\mu3=\mu1$
3 vs. 2	3.87	0.62	6.20	4.367 - Conclusion: Reject Ho $\mu3=\mu2$
2 vs. 1	0.07	0.62	0.11	4.367 - Conclusion: Accept Ho $\mu3=\mu2$