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Influence of Forest and Rangeland Management on Anadromous Fish Habitat in Western North America

EFFECTS OF MINING

SUSAN B. MARTIN AND WILLIAM S. PLATTS

USDA FOREST SERVICE
ANADROMOUS FISH HABITAT
PROGRAM

U.S. Department of Agriculture Forest Service
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ABSTRACT

Methods of mining and the effects on aquatic ecosystems of mine-caused sediment, changes in pH, and toxic heavy metals are described.

KEYWORDS: Mining methods, aquatic ecosystems, sediment production, toxicity, fish habitat, anadromous fish.

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INFLUENCE OF FOREST AND RANGELAND MANAGEMENT ON ANADROMOUS FISH HABITAT IN WESTERN NORTH AMERICA

William R. Meehan, Technical Editor

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SUSAN B. MARTIN AND WILLIAM S. PLATTS

Intermountain Forest and Range Experiment Station
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PACIFIC NORTHWEST FOREST AND RANGE EXPERIMENT STATION
Forest Service, U.S. Department of Agriculture, Portland, Oregon

PREFACE

This is one of a series of publications summarizing knowledge about the influences of forest and rangeland management on anadromous fish habitat in Western North America. This paper addresses the effects of mining on anadromous fish habitat. Our intent is to provide managers and users of the forests and rangelands of Western North America with the most complete information available for estimating the consequences of various management alternatives.

In this series of papers, we summarize published and unpublished reports and data as well as observations of resource scientists and managers. These compilations should be valuable to resource managers in planning uses of forest and rangeland resources, and to scientists in planning future research. The extensive lists of references serve as a bibliography on forest and rangeland resources and their uses.

Previous publications in this series include:

1. "Habitat requirements of anadromous salmonids," by D. W. Reiser and T. C. Bjornn.

2. "Impacts of natural events," by Douglas N. Swanston.

4. "Planning forest roads to protect salmonid habitat," by Carlton S. Yee and Terry D. Roelofs.

11. "Processing mills and camps," by Donald S. Schmiege.

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INTRODUCTION

About one-third of the United States is public land. Of these 743.2 million acres (300 million hectares), about 68 percent are open to mining (Sheridan 1977).

Congress mandated, through the Multiple Use-Sustained Yield Act of 1976, that public lands be managed for multiple use. But the Mining Law of 1872 conflicts with this objective, because it guarantees a single use-mining--wherever a valuable mineral deposit is found. Thus, mining can usurp other resource uses, unless the land has been specifically withdrawn from mineral development. Although mining is one of the chief uses of public lands, it is not necessarily part of the land-use planning process.

METHODS OF MINING

Surface mining, the earliest form of mineral recovery, is the process of removing minerals on or near the soil surface after stripping vegetation and overburden (soil and rock covering). Strip mining is a form of surface mining that results in a long, continuous excavation bordered by one or two parallel waste piles (contour mining) or by a series of parallel waste piles (area mining). This method is generally used to recover deposits of coal, lignite, clay, phosphate, and gypsum.

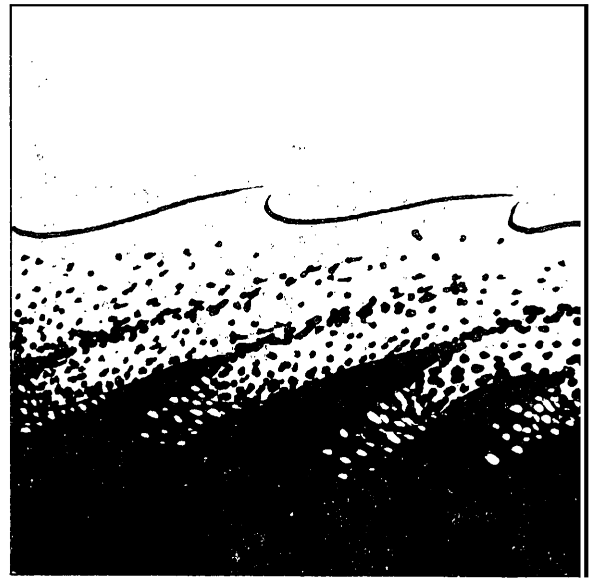
Auger mining is drilling into side slopes. Auger diameters may be as great as 7 feet (2.1 m), with holes 200 feet (71 m) or more in length. This method is frequently used to remove coal where overburden is too thick to permit economic stripping. Combinations of strip and auger mining are routinely used in Appalachia where coal strata normally occur in horizontal planes.

In quarrying or open-pit mining, the open excavations may have several working levels or benches. This method is used where the overburden is limited in proportion to the commodity recovered. Pit mining is used for removal of limestone, copper, uranium, iron, stone, sand, and gravel.

Dredge mining, which uses bucket or suction equipment to extract minerals from under water or subsurface water, is used extensively to recover gold, sand, and gravel, and some beach sand deposits containing rare earths.

In hydraulic mining, soil is eroded by high-pressure water jets. This method was used extensively in western gold mining, and it is still used in sand mining in coastal areas and in gold and coal mining in some western States.

In underground or hardrock mining, tunnels and shafts are cut into the earth. Pollution often results from the mining and milling operations themselves, as well as from tailings ponds and waste dumps associated with the mine.



MINING AND **AQUATIC** POLLUTION

Mining can pollute the aquatic environment by producing sediment, changes in pH, toxic heavy metals, and alterations in stream channel and streamflow.

SEDIMENT PRODUCTION

Sediment accrues in streams naturally and at moderate levels can be a beneficial component of anadromous fish habitat. Major disruption of the system occurs when amounts of sediment become excessive, however (Platts and Megahan 1975). Deposition of excessive fine sediment on the stream bottom eliminates habitat for aquatic insects; reduces density, biomass, number, and diversity of aquatic insects; reduces the permeability of spawning gravels; and blocks the interchange of subsurface and surface waters (Cooper 1959, Vaux 1962, McNeil 1964, Koski 1966). Toxic heavy metals can precipitate on bedload sediment particles and remain in the aquatic environment to be released later (Funk et al. 1975). Stream microorganisms can feed on sediments containing organic material and lower the dissolved oxygen content of the water (University of Pittsburgh 1972).

Sediments also contain nutrients, such as nitrogen and phosphorous. Excessive nutrient levels can lead to blooms of undesirable algal and plankton species and killing of fish from depletion of oxygen (Becker and Thatcher 1973).

The effects of suspended sediment on the aquatic system are more direct. Photosynthesis may be reduced because of light reduction; fish migration may be affected (European Inland Fisheries Advisory Commission 1965); fish may not be able to feed under turbid conditions, resulting in smaller fish (Sykora et al. 1972); and suspended solids may interfere with efficient respiration of gilled animals (Coker 1968). Young salmonids are particularly susceptible to gill irritation caused by turbid water, which in turn exposes them to infection by fungi and bacteria.

Mining can produce significant sources of bedload sediment and cause suspended solids to enter aquatic ecosystems. Glancy (1973) found annual sediment yields of 620 to 7,600 tons/mi^2 (218 to 2 670 metric tons/km^2) from mined areas, whereas undisturbed areas yielded only 60 to 930 tons/mi^2 (21 to 326 metric tons/km^2). The recovery of a stream affected by manganese strip-mining operations was monitored by Cumming and Hill (1971). Six years were required for the stream to recover fully from the mining effects after mine closure. Turbidity was the most damaging pollutant measured. Branson and Butch (1971) also concluded that the most damaging pollutant in a strip mine area was not the acid mine drainage, but the high siltation and turbidity originating from erosion of the spoil banks. Research in Kentucky indicated that sediment yields from forested areas were increased 1,000 times as a result of strip mining (Musser 1963). During a 4-year period, the average annual sediment yield from spoil banks was 27,000 tons/mi^2 (8 900 metric tons/km^2), but from control areas it was estimated at only 25 tons/mi^2 (8 metric tons/km^2). In Appalachia, Spaulding and Ogden (1968) estimated that about 1 acre-foot (1 230 m^3) of sediment was produced by every 80 acres (30 ha) of disturbed land.

Collier et al. (1970) and Curtis (1973) also documented the increased bedload sediment and suspended solids from surface-mine areas. Mine waste piles, haulroads, tailings ponds, coal and ore stockpiles, and beneficiation plants were all sources of sediment. Collier et al. showed that a partially strip-mined watershed (6.4 percent of area) had an erosion rate of 5.9 tons/acre (13.1 metric tons/ha) as compared to 0.7 tons/acre (1.5 metric tons/ha) for an unmined area. Peak flows during large rainstorms contained suspended solid concentrations of over 46 000 mg/liter from an unprotected strip-mine area in an Appalachian watershed study (Curtis 1971). If adequate vegetative cover is not established, continual exposure of fresh spoil prolongs the production of acid, soluble salt, and sediment to the aquatic system (Walker 1952, Vogel 1971, Farmer et al. 1976).

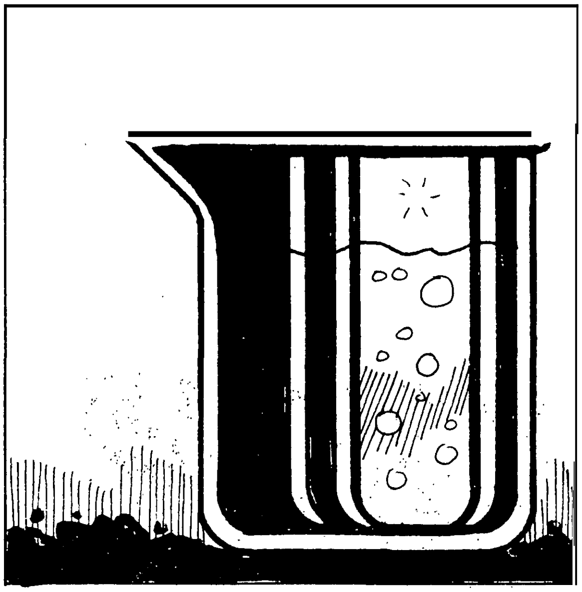
Lack of vegetative cover can also affect the extent of heavy metal contamination. Exposing rock strata to weathering and erosion results in higher zinc, copper, nickel, and iron concentrations in receiving streams (Neckers and Walker 1952, Massey and Barnhisel 1971, Massey 1972). Curtis (1971) concluded that adverse effects on hydrology can subside within 6 months under some conditions, if vegetative cover is reestablished.

In a study of the sand and gravel industry, Newport and Moyer (1974) reviewed the effects of sediment on benthic and planktonic communities, and on population, reproduction, and species composition of fish. They concluded that although a precise maximum concentration of inorganic suspended solids had not been detrimental to fish, the following were good approximations of the potential effects of sediment:

0-25 mg/liter	no harmful effects on fish
25-100 mg/liter	good to moderate fish habitat
100-400 mg/liter	unlikely to support a good fishery
400 mg/liter and above	poor fisheries habitat

Maximum allowable concentrations of suspended solids in streams classified for trout cannot exceed 10 mg/liter in several States (Hill 1974).

Inorganic silt was the pollutant limiting populations of fish and bottom organisms in reclaimed and partially reclaimed streams after strip mining for manganese (Hill 1972). Complete reclamation of spoil areas reduced siltation and turbidity, allowing recovery of stream fauna.



CHANGES IN pH

Acid is a major pollutant from mine drainage. Acid mine drainage occurs when pyritic material in sulfurous ores, usually associated with mining coal and heavy metals, is oxidized to produce ferrous sulfate and sulfuric acid. This results in a pH range of *2.0-4.5* in receiving waters, levels toxic to many forms of aquatic life (Hill 1974). A favorable pH range for fish is generally from 6.5 to 8.7 (Environmental Protection Agency 1976). Although fish can exist for short periods at slightly above and below this range, a pH lower than 6.0 is generally unfavorable for fish populations (Spaulding and Ogden 1968).

Some of the potential damages to aquatic biota from acid pollution are:

- Elimination of sensitive species and proliferation of tolerant species,
- Direct acute effects,
- Reduction in density, biomass, and diversity of aquatic organisms,
- Increase in abnormal behavior,
- Reduction in reproductive capacity of adults and in viability of eggs and alevins.

At lower pH levels, the aquatic biota may be virtually eliminated. Katz (1969) found no viable fishery at pH 3.5 and below. The dominant macroinvertebrates were chironomids and Sialis sp., generally considered indicators of poor water quality. No caddisflies (Trichoptera), mayflies (Ephemeroptera), or stoneflies (Plecoptera)--the preferred foods for salmonids-- were present at this low pH.

Studies in Norway and Sweden indicated extinction of fish populations often resulted from chronic reproductive failure because of acidification-induced effects on sensitive developmental stages (Brungs et al. 1978). Acid stress (pH 4.0-5.0) also has caused fish mortality, by interfering with the physiological mechanisms regulating active ion exchange across gill membranes (Brungs et al. 1978).

Although sublethal levels of acidity are not cumulative in fish, Anthony (1971) considered pH 5.0 or below hazardous, because they reduce the ability of fish to detoxify other poisons. Fish are apparently able to eliminate heavy metals from their bodies to a certain extent, but when the heavy metal concentration becomes too great or the pH too low, this ability is impaired.

Healthy, unpolluted streams generally have moderate numbers and many species of organisms, but polluted areas have larger numbers of a single organism. With toxic wastes--such as pollution from mining--both number and diversity of organisms are reduced. In West Virginia, more species of insects and algae occurred in an unpolluted stream (pH *4.5* or higher) than in those areas polluted by acid (pH 2.8 to 3.8) (Warner 1971). Menendez (1978) found a reduction in benthic fauna in a West Virginia stream severely affected by acid mine water.

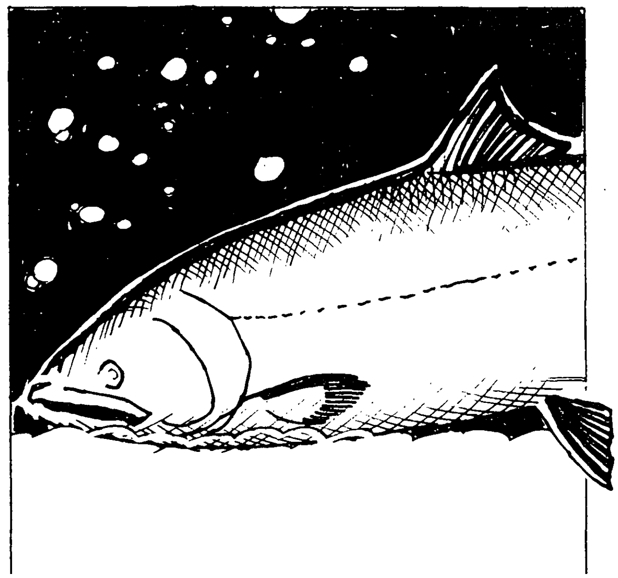
Low pH values may also affect behavior and reproduction of aquatic organisms. In a study on fathead minnows (Pimephales promelas Rafinesque), Mount (1973) found that fish behavior was abnormal and fish were deformed at pH 4.5-5.2. Abnormal eggs and reduced production and survival were observed at pH 5.9 and lower. A pH of 6.6 was marginal for vital life functions, but safe for continuous exposure.

Trojnar (1977) investigated the hatching response in eggs of brook trout (Salvelinus fontinalis (Mitchill)), incubated in gravel at pH 4.6 to 8.1, and observed the response of fry to different pH at emergence. Survival was zero when eggs were incubated at pH 8.1, and fry emerged at pH 4.0. With incubation and emergence at pH 4.6 to 5.0, fry survival increased 60 to 76 percent. This suggests an acclimation effect: fry incubated at lower, sublethal pH levels are less susceptible to acid mortality upon emergence than those reared under normal conditions (pH 6.7 to 8.2).

In addition, at low pH (pH<5.0), heavy metals that are complexed or coprecipitated with suspended solids and sediments may be released, adding another toxic pollutant to the aquatic system (Sorenson et al. 1977).

Acid drainage from large surface mines is not normally a problem in the West, as the alkaline soils in the region reduce the potential for acid drainage. Problems may arise from alkaline drainage, however. Iron is precipitated from oxygenated water as ferric hydroxide, which can smother fish eggs and cause gill irrigation (McKee and Wolf 1971). This precipitation is more complete in alkaline water than in acidic water (Ruttner 1963).

Oil-shale mining presents special alkaline pollution problems. Leachate from the shale is alkaline and high in total dissolved solids, with high concentrations of sodium, calcium, magnesium, and sulfates. Toxic amounts of aluminum may also be present in some shale. Elevated salinities from oil shale have also occurred in the lower Colorado River (Moore and Mill 1977).



TOXICITY OF HEAVY METALS

Dissolved heavy metals, commonly found in waters polluted by mine drainage, are toxic to the aquatic biota (Cairns and Scheier 1957; Tarzwell and Henderson 1960; Lloyd 1960, 1961a, 1961b; Lloyd and Herbert 1962; Mount and Stephan 1967). The actual metal concentration toxic to fish, however, is difficult to determine. Toxicity depends on fish species, age and stage of development (Lloyd 1960), water temperature (Chapman 1973), pH, dissolved oxygen concentration, total hardness, and synergism with other pollutants. In general, fish mortality results from exposure to high metal concentrations, but continuous low exposure produces chronic effects, such as behavioral changes, reproductive failure, or fry mortality (Chapman 1973). Both ultimately affect species survival.

Toxic metals commonly released by mining are arsenic, cadmium, cobalt, copper, iron, lead, manganese, mercury, nickel, and zinc. Individually, these metals may be toxic to aquatic biota, or they may exhibit a combined toxicity greater than that of the individual elements. Synergistic toxicity is common in waters polluted by heavy metals from mining. For example, in laboratory tests with a mixture of copper, cadmium, and zinc, a lethal threshold for fathead minnows was attained when each metal was present at a concentration of 40 percent or less of its individual lethal threshold (Eaton 1973).

Movements of fish in streams may be affected by the presence of heavy metals. Sprague (1964), in laboratory tests on Atlantic salmon (Salmo salar Linnaeus), found fish avoided copper and zinc at concentrations well below those found to cause mortality. Sprague reported metal concentrations as the "incipient lethal level" (ILL)--that level beyond which the organism could no longer survive for an indefinite period. Migrating Atlantic salmon avoided copper at 0.002 mg/liter (0.052 ILL) and zinc at 0.053 mg/liter (0.092 ILL). The maximum allowable concentration of copper recommended in waters inhabited by fish and other aquatic life is 0.02 mg/liter (McKee and Wolf 1971). For zinc, the concentration is 0.3 mg/liter, but for salmonids may be as low as 0.1 mg/liter (Rabe and Sappington 1970). These values are about ten times greater than those found to cause overt avoidance behavior in migrating fish, which illustrates how sensitive fish are to heavy metals.

Heavy-metal pollution may also have acute physiological effects on fish. Zitko and Carson (1977) found the ILL of zinc to juvenile Atlantic salmon varied from 0.15 to 1.0 mg/liter from fall through spring, with no detectable seasonal changes in water hardness, alkalinity, dissolved nitrogen, and humic acid concentration. The most sensitive period, as measured by the zinc ILL, coincided with the initial stages of the parr-smolt transformation and was, therefore, believed to be related to this physiological change. Studies by Lorz and McPherson (1977) also support this hypothesis.

Skidmore and Tovell (1972) demonstrated that acute exposure of rainbow trout (Salmo gairdneri Richardson) to zinc sulfate (40 mg/liter) caused a severe inflammatory reaction in the gill, with a separation of the epithelium outward from the pillar cells. This was followed by circulatory breakdown, tissue destruction, respiratory collapse, and death. Burton et al. (1972) confirmed that the major physiological change preceding death in acute toxicity studies with zinc was tissue hypoxia. The hypoxia was directly related to gill-tissue damage, which disrupts normal gas exchange at the gill surface.

The effect heavy-metal exposure has on aquatic organisms may be related to their physiological state. Boyce and Yamada (1977) investigated the difference in zinc susceptibility between juvenile sockeye salmon (Oncorhynchus nerka (Walbaum)) infected with the intestinal cestode Eubothrium salvelini (Schrank), and noninfected smolts. Under laboratory conditions, infected smolts were significantly more susceptible to 1 mg/liter of zinc than were noninfected smolts.

Chronic, low levels of heavy metals in water may cause long-term behavior changes in fish. Drummond et al. (1973) found behavioral changes in brook trout at concentrations as low as 0.005 mg/liter of copper. Studies on brook trout by McKim and Benoit (1971) showed that copper concentrations of 0.002 to 0.03 mg/liter had no effect on adult fish, but had a marked effect on survival and growth of alevins and juveniles. Zinc concentrations of 0.18 mg/liter greatly reduced egg production of the fathead minnow, a fish more resistant to heavy-metal toxicity than the salmonids (Brungs 1969). In the same study, Brungs found that fathead minnow growth was inhibited and mortalities occurred at 2.8 mg/liter of zinc. Holcombe et al. (1976) found that long-term exposure of brook trout to lead resulted in physiological changes. In three generations of fish exposed to lead concentrations ranging from 0.001 to 0.5 mg/liter, second- and third-generation trout developed spinal deformities (scoliosis). In addition, growth of third-generation trout was reduced.

Nonlethal levels of trace elements are subject to bioaccumulation in aquatic organisms (Krenkel 1973). In general, metal content in most tissues has been shown to decrease at higher trophic levels. Zinc concentrations may decrease in muscle from omnivorous species to predacious fish because of differences in metal content of the food they consume. An omnivorous fish feeds on detrital material, which likely contains high levels of metals. Among predacious fish, piscivores consume material that contains less metal than fish that feed primarily on invertebrates. Differences in metal content of tissue among fish species may also be related to fish size. Tissue levels of heavy metals, other than mercury, tend to decrease with an increase in size or age of fish (Bauer 1974).

Other aquatic organisms also show a response to the presence of heavy metal pollution. Some members of the aquatic midgefly family (Chironomidae) are tolerant of high copper concentrations--up to 2.2 mg/liter (Surber 1959). Some caddisfly larvae may be equally tolerant of heavy-metal pollution (Sprague et al. 1965). Most mayfly nymphs cannot survive in streams with heavy-metal concentrations far below those lethal to trout (McKee and Wolf 1971). For example, Warnick and Bell (1969) found 0.3 mg/liter of iron toxic to mayflies, stoneflies, and caddisflies. Fish populations, however, can tolerate iron concentrations up to 1.0 mg/liter (Ellis 1940). Enk and Mathis (1977) found higher cadmium and lead concentrations in sediments and aquatic insects than in fish in an Illinois stream. This suggests that the sediments acted as a sink for the metals, and bottom-feeding aquatic insects concentrated the metals.

The complex interactions of organisms in an aquatic system can also be influenced by the presence of heavy metals. Sodergren (1976) concluded that a reduction of invertebrate fauna in a Swedish stream polluted with cobalt resulted in a drastic reduction of juvenile Atlantic salmon.

STREAM CHANNEL AND STREAMFLOW ALTERATION

Stripping vegetation and dredging and channelizing streams can occur during mining operations. Such activities will result in increased erosion and sediment because of unstable streambanks and can lead to increased streamflows. Removal of vegetation often results in higher peak flows during runoff and lower base flows during, low-flow periods. Reduction in surface-water area or streamflow may reduce the quantity and quality of habitat available to fish and other aquatic organisms (Moore and Mill 1977). Direct use of water in milling and mining operations may also decrease stream flows.

ABANDONED MINES

Pollution from abandoned, "orphan" mines can be as much of a problem as from mines that are active. Matter et al. (1978) found that sulfate, total hardness, and silt indices were elevated in streams draining abandoned mines in Virginia. Total abundance and taxonomic richness of fish and benthic invertebrate populations were reduced in streams draining orphan-mine areas, even 10 to 20 years after abandonment.

Researchers found that an old open-pit mine in Montana caused water quality problems in the upper Stillwater River even though operations ceased there in the early 1950's (Brown and Johnston 1976). Acid drainage continued to kill the native vegetation adjacent to the mine and to destroy the aquatic ecosystem in the river. The area also has high soil erosion and stream sedimentation resulting from the lack of vegetative cover. Highly acid conditions and concentrations of heavy metals in soils within the rooting zone aggravate reclamation problems (Johnston et al. 1975).

In studies of the abandoned Blackbird Mine in central Idaho, Farmer et al. (1976) found little vegetative regrowth on a low-grade mine-waste pile. Twenty years after mine closure, soil in the dump was still severely acidic (pH 3.0 to 4.0) and infertile. Extensive rehabilitation efforts were necessary to reestablish vegetation on the waste pile.

Platts et al. (1979), in another study of the Blackbird Mine, found that heavy-metal concentrations in the Panther Creek watershed remained high after mine closure. Mean concentrations of cobalt and copper in polluted streams were toxic to aquatic biota, and levels of iron, lead, and manganese were higher than in streams draining unmined areas. Water samples from sites downstream from the mined area had consistently lower pH values than those upstream. The stream channel below the mined areas had high concentrations of fine sediment even after mine closure--the result of erosion from unreclaimed waste piles and tailings ponds. Sediment and heavy-metal pollutants completely eliminated runs of chinook salmon (*Oncorhynchus tshawytscha* (Walbaum)) and steelhead trout (*Salmo gairdneri* Richardson) that occupied the drainage before mining (Platts 1972).

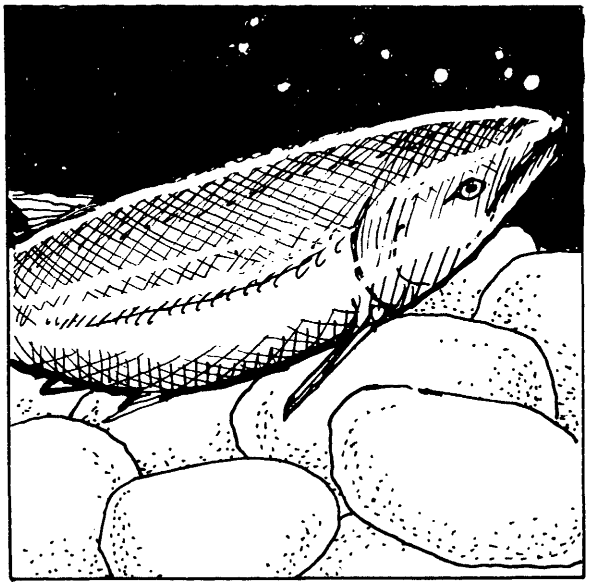
DISCUSSION

The effect of mining pollution on aquatic environments has been significant. When no reclamation efforts are made, the effects of mining on streams may be prolonged long after active operations cease.

State and Federal governments are becoming more active in regulating mining activities to conserve and protect natural resources. In most western States, a plan outlining potential environmental damage and proposed rehabilitation is required before permission to mine a site is granted. Posting of a reclamation performance bond to cover damages is also necessary.

Adequate Federal and State water-quality standards to protect aquatic environments now exist in most areas. Manpower limitations, however, make monitoring and enforcement of these regulations difficult. Idaho, for example, has only one land-reclamation specialist. This person must rely on help from other agencies to supplement information on whether mine operations are in compliance.

Although future protection of the aquatic habitat in mine areas looks promising, more funds are necessary to guarantee adequate safeguards. More research in developing reclamation measures is also needed. The aquatic environment should be treated as the important resource that it is.



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