CHAPTER 11

Cyanide Hazards to Plants and Animals from Gold Mining and Related Water Issues

Highly toxic sodium cyanide (NaCN) is used increasingly by the international mining community to extract gold and other precious metals through milling of high-grade ores and heap leaching of low-grade ores. The process to concentrate gold using cyanide was developed in Scotland in 1887 and used almost immediately in the Witwatersrand gold fields of the Republic of South Africa. Heap leaching with cyanide was proposed by the U.S. Bureau of Mines in 1969 as a means of extracting gold from low-grade ores. The gold industry adopted the technique in the 1970s, soon making heap leaching the dominant technology in gold extraction (Da Rosa and Lyon 1997). The heap leach and milling processes, which involve dewatering of gold-bearing ores, spraying of dilute cyanide solutions on extremely large heaps of ores containing low concentrations of gold, or milling of ores with the use of cyanide and subsequent recovery of the gold–cyanide complex, have created a number of serious environmental problems affecting wildlife and water management. This chapter reviews the history of cyanide use in gold mining with emphasis on heap leach gold mining, cyanide hazards to plants and animals, water management issues associated with gold mining, and proposed mitigation and research needs.

11.1 HISTORY OF CYANIDE USE IN GOLD MINING

About 100 million kg cyanide (CN) are consumed annually in North America, of which 80% is used in gold mining (Eisler et al. 1999; Fields 2001). In Canada, more than 90% of the mined gold is extracted from ores with the cyanidation process, which consists of leaching gold from the ore as a gold–cyanide complex and recovering the gold by precipitation. The process involves the dissolution of gold from the ore in a dilute cyanide solution and in the presence of lime and oxygen according to the following reactions (Hiskey 1984; Gasparrini 1993; Korte and Coulston 1998):

$$(1)\quad 2Au + 4NaCN + O_2 + 2H_2O \rightarrow 2NaAu(CN)_2 + 2NaOH + H_2O_2$$

$$(2) \quad 2Au + 4NaCN + H_2O_2 \rightarrow 2NaAu(CN)_2 + 2NaOH$$

Depending on solution pH, free cyanide concentrations, and other factors, gold is recovered from the eluate of the cyanidation process using either activated carbon, zinc, or ion-exchange resins (Adams et al. 1999). Using zinc dust, for example, gold along with silver is precipitated according to the reaction (Hiskey 1984; Gasparrini 1993):

$$(3) \quad 2NaAu(CN)_2 + Zn \rightarrow Na_2Zn(CN)_4 + 2Au$$

The process known as carbon in pulp controls the gold precipitation from the cyanide solution using activated charcoal. It is used on low-grade gold and silver ores in several processing operations in the western United States, mainly to control slime-forming organisms. After precipitation, the product is treated with dilute sulfuric acid to dissolve residual zinc and almost all copper present. The residue is washed, dried, and melted with fluxes. The remaining gold and silver alloy is cast into molds for assay. Refining is accomplished via electrolysis, during which silver and platinum group elements are separated and recovered. Another method of separating gold from silver is by parting, wherein hot concentrated sulfuric or nitric acid is used to differentially dissolve the silver, and the gold is recovered from the residue (Hiskey 1984; Gasparrini 1993).

Milling and heap leaching require cycling of millions of liters of alkaline water containing high concentrations of NaCN, free cyanide, and metal cyanide complexes that are available to the biosphere (Eisler 2000). Some milling operations result in tailings ponds 150 ha in area and larger. Heap leach operations that spray or drip cyanide solution onto the flattened top of the ore heap require solution processing ponds of about 1 ha surface area. Puddles of various sizes may occur on the top of heaps where the highest concentrations of NaCN are found. Solution recovery channels are usually constructed at the base of leach heaps; sometimes, these are buried or covered with netting to restrict access of vertebrates.

All these cyanide-containing water bodies are hazardous to natural resources and human health if not properly managed (Eisler 1991, 2000; Henny et al. 1994). For example, cyanide-laced sludges from gold mining operations stored in diked lagoons have regularly escaped from these lagoons. Major spills occurred in Guyana in 1995 and in Latvia and Kyrgyzstan in the 1990s (Koenig 2000). Failure of gold mine tailings ponds killed one child in Zimbabwe in 1978 and 17 people in South Africa in 1994 after a heavy rainfall, and contaminated streams and rivers in New Zealand in 1995 (Garcia-Guinea and Harffy 1998) and elsewhere (Leduc et al. 1982; Alberswerth et al. 1989; Koenig 2000; Kovac 2000).

In September 1980, the price of gold had increased to $750 per troy ounce (1 Troy ounce = 31.1035 g) from $35 a decade earlier (Gasparrini 1993). This economic incentive resulted in improved cyanide processing technologies to permit cost-effective extraction of small amounts of gold from low-grade ores (Henny et al. 1994). The state of Nevada is a major global gold-producing area, with at least 40 active operations. Increased gold mining activity is also reported in other western states, Alaska, the Carolinas, and northern plains states. Where relatively high-grade ores (>0.09 troy ounce Au/t ore) are found, milling techniques are used, but heap leaching of low-grade ores (0.006 to 0.025 troy ounce Au/t) is the most commonly employed extraction technique (Henny et al. 1994). Heap leach facilities usually produce gold for less than $200 US/troy ounce (Greer 1993).

The amount of gold produced in the United States by heap leaching rose 20-fold throughout the 1980s, accounting for 6% of the supply at the beginning of the decade and more than 33% at the end (Greer 1993). In 1980, there were approximately 24 heap leach facilities in the U.S.; by 1991, there were 265, of which 151 were active. The rise in domestic gold production in this period from 31 tons in 1980 to 295 tons in 1990 is attributable mainly to cyanide heap leaching (Greer 1993). Although more tons of gold ore are heap leached than vat leached in the U.S. today, a greater quantity of gold is actually produced by vat leaching because that method is used on higher-grade ores and has a higher gold recovery rate (Da Rosa and Lyon 1997). In 1989, cyanide heap leaching produced 3.7 million troy ounces from 129.8 million tons of ore, and cyanide vat leaching produced 4.3 million troy ounces of gold from 40.6 million tons of ore (Da Rosa and Lyon 1997).

Heap leaching occurs when ore, stacked on an impermeable liner at the ground surface, is sprayed or dripped with a dilute (usually about 0.05%) NaCN solution on the flattened top for a period of several months. Large leach heaps may include 1 to 25 million tons of ore, tower 100 meters high or more, and occupy several hundred hectares. As the solution percolates through the heap, gold is complexed and dissolved. For best results, heap-leached ores need to be porous, contain fine-grained clean gold particles, have low clay content, and have surfaces accessible to leach solutions. After the gold-containing solution is collected in a drainage pond, the gold is chemically precipitated, and the remaining solution is adjusted for pH and cyanide concentration and recycled to precipitate more gold. Eventually the remaining solution is treated to recycle the cyanide or to destroy it to prevent escape into the environment.

Cyanide and other contaminants may be released through tears and punctures in pad liners; leaks in liners carrying the cyanide solution; open ponds, piles, and solution ponds that can overflow; nitrogen compounds released during cyanide degradation; and release of lead, cadmium, copper, arsenic, and mercury, present in ore, that can be mobilized during crushing or leaching (Hiskey 1984; Alberswerth et al. 1989; Greer 1993; Wilkes and Spence 1995; Mosher and Figueroa 1996; Korte and Coulston 1998; White and Schnabel 1998; Korte et al. 2000; Tarras-Wahlberg et al. 2000) (Table 11.1). The amount of hydrogen cyanide that escapes into the atmosphere from gold mining operations is estimated at 20,000 tons annually, where it is quite stable; the half-time persistence of HCN in the atmosphere is about 267 days (Korte and Coulston 1998).

Cyanide is also used in agitation leaching on ores that require finer grinding than those subjected to heap leaching, and in pressure leaching and pressure cyanidation, in which cyanide penetrates at high temperature and pressure into compact ores where the gold occurs in fine fractures (Gasparrini 1993).

Table 11.1 Cyanide and Metals Concentrations in Water and Sediments Downstream of Portovela-Zaruma Cyanide-Gold Mining Area, Ecuador; Dry Season, 1988

Component and Toxicant	Observed vs. Recommended Safe Value
	Water
Free cyanide	6–13 µg/L vs. 24-hr maximum safe level of <3.5 µg/L
Total cyanide	220–2600 µg/L vs. chronic exposure value of <5.2 µg/L
Arsenic	2–264 µg/L vs. chronic exposure value of <190 µg/L
Cadmium	<0.005-0.7 µg/L vs. chronic exposure value of <0.4 µg/L
Copper	0.3–23.2 µg/L vs. chronic exposure value of <3.6 µg/L
Lead	0.04–2.5 µg/L vs. chronic exposure value of <2.5 µg/L
Mercury	<0.0022–1.1 µg/L vs. chronic exposure value of <0.1 µg/L
	Sediments
Arsenic	403–7700 mg/kg dry weight (DW) vs. no adverse effect level of <17 mg/kg DW
Cadmium	1–48 mg/kg DW vs. no probable effect level of <3.5 mg/kg DW
Copper	303–5360 mg/kg DW vs. no probable effect level of <197 mg/kg DW
Lead	9–4470 mg/kg DW vs. no probable effect level of <91 mg/kg DW
Mercury	0.1–5.8 mg/kg DW vs. no probable effect level of <0.45 mg/kg DW

Source: Modified from Tarras-Wahlberg et al. 2000.

Individual mines often cover thousands of hectares, and mining companies sometimes lease additional thousands of hectares for possible mining (Clark and Hothem 1991). Ultimately, mining converts the site into large flat-topped hills of crushed ores, waste rock, or extracted tailings and large open pits. This alteration may result in permanent damage to wildlife habitat, although most areas, with the general exception of open pits, are reclaimed through revegetation. Between 1986 and 1991, cyanide in heap leach solutions and mill tailings ponds at gold mines in Nevada alone killed at least 9500 birds, mammals, reptiles, and amphibians. Dead birds representing 91 species, especially species of migratory waterfowl, shorebirds, and gulls, comprised about 90% of the total number of animals found dead, mammals 7% (28 species), and amphibians and reptiles together 3% (6 species; Henny et al. 1994). In more recent years, the Nevada Division of Wildlife, through its toxic pond permit program (Nevada Administrative Code 502.460 through 502.495) and cooperative work with mining companies, significantly reduced the number of cyanide-related deaths of vertebrate wildlife.

Heap leaching operations are closely monitored by regulatory agencies. In California, for example, at least six permits are necessary before cyanide extraction may commence: (1) a water use permit, obtained from the California Water Board; (2) a waste discharge permit, obtained from the California Regional Water Quality Board; (3) an air quality permit, from the California Air Pollution Control District; (4) a conditional use permit, from the local county; (5) an operations plan permit, from the U.S. Bureau of Land Management; and (6) a radioactive material license, from the California Department of Health Sciences (Hiskey 1984).

Under certain alkaline conditions, cyanide may persist for at least a century in groundwater, mine tailings, and abandoned leach heaps (Da Rosa and Lyon 1997). Cyanide destruction by natural reaction with the ore, soil, clay, and microorganisms has been advanced as the major mechanism for returning a site to an environmentally safe condition. To legally shut down the operation, concentrations <0.2 mg/L of weak acid dissociable cyanide (metal-bound cyanide dissociable in weak acids, WAD) are required (White and Schnabel 1998). The use of cyanide to extract gold was banned in Turkey by the Turkish Supreme Court in 1999 because of accidental releases into the environment of untreated cyanide wastes stored in open ponds and resultant harm to human and ecosystem health (Korte et al. 2000). In Turkey, where more than 250,000 tons of crushed rocks with mean gold content of 3 g/t were subjected to 125,000 tons of sodium cyanide in 365,000 m^3 water every year, more than 2 million m^3 untreated cyanide/heavy metals solution had accumulated in waste ponds. Other countries that are considering prohibition of the cyanide leaching gold recovery process include the Czech Republic, Greece, and Romania (Korte et al. 2000).

Alkaline chlorination of wastewaters is one of the more widely used methods of treating cyanide wastes. In this process, cyanogen chloride (CNCl) is formed, which is hydrolyzed to the cyanate (CNO^-) at alkaline pH. If free chlorine is present, CNO^- can be further oxidized (Simovic and Snodgrass 1985; Marrs and Ballantyne 1987). The use of sulfur dioxide in a high-dissolved-oxygen environment with a copper catalyst reportedly reduces total cyanide in high-cyanide rinse waters from metal plating shops to less than 1 mg/L; this process may have application in cyanide detoxification of tailings ponds (Robbins 1996).

Other methods used in cyanide waste management include lagooning for natural degradation, evaporation, exposure to ultraviolet radiation, aldehyde treatment, ozonization, acidification–volatilization–neutralization, ion exchange, activated carbon absorption, electrolytic decomposition, catalytic oxidation, treatment with hydrogen peroxide, and biological treatment with cyanide-metabolizing bacteria (Towill et al. 1978; Way 1981; Marrs and Ballantyne 1987; Smith and Mudder 1991; Mosher and Figueroa 1996; Ripley et al. 1996; Dictor et al. 1997; Adams et al. 1999). Additional cyanide detoxification treatments include the use of $FeSO_4$; $FeSO_4$ plus CO_2, H_2O_2, and $Ca(OCl)_2$; dilution with water; and $FeSO_4$ plus H_2O_2, and $(NH_4)HSO_3$ (Adams et al. 1999; Eisler et al. 1999). In Canadian gold mining operations, the main treatment for cyanide removal is to retain wastewaters in impoundments for several days to months; removal occurs through volatilization, photodegradation, chemical oxidation, and secondarily through microbial oxidation (Simovic and Snodgrass 1985).

In general, because chemical treatments do not degrade all cyanide complexes, biological treatments are used (Figueira et al. 1996). Biological treatments include (1) oxidation of cyanide compounds and thiocyanate by *Pseudomonas paucimobilis* with 95% to 98% reduction of cyanides in daily discharges of 15 million L; (2) metabolism of cyanides by strains of *Pseudomonas*, *Acinetobacter*, *Bacillus*, and *Alcaligenes* involving oxygenase enzymes; and (3) bacterial cyanide degraders involving cyanide oxygenase, cyanide nitrilase, and cyanide hydratase (Figueira et al. 1996).

Microbial oxidation of cyanide is reportedly not significant in mine tailings ponds because of the high pH (>10), low number of microorganisms, low nutrient levels, large quiescent zones, and cyanide concentrations >10 mg/L (Simovic and Snodgrass 1985). However, cyanide-resistant strains of microorganisms are now used routinely to degrade cyanide. Biological degradation of cyanide in which CN^- is converted to CO_2, NH_3, and OH^- by bacteria, when appropriate, is considered the most cost-effective method in cyanide detoxification and has been used in cyanide detoxification of heap leaches containing more than 1.2 million tons (Mosher and Figueroa 1996). Concentrations of 10^5 cells of *Pseudomonas alcaligenes*/mL can reduce cyanide from 100 to <8 mg/L in 4 days at elevated pH (Zaugg et al. 1997). Strains of *Escherichia coli* isolated from gold extraction liquids metabolically degrade cyanide at concentrations up to 50 mg HCN/L in the presence of a glucose-cyanide complex (Figueira et al. 1996). Ammonia accumulated as the sole nitrogen by-product and was used for growth of *E. coli* involving a dioxygenase enzyme that converted cyanide directly to ammonia without cyanate formation (Figueira et al. 1996).

Removal of free cyanide, thiocyanate, and various metallocyanides from a synthetic gold milling effluent was accomplished using biologically acclimatized sludge; the adapted microbial consortium removed >95% of free cyanide, thiocyanate, copper, and zinc from the original effluent in about 8 hours (Granato et al. 1996). Biological treatment of a leachate containing cyanide was accomplished with a mixed culture of microorganisms, *Pseudomonas* and other species isolated from waste-activated sludge of the Fairbanks, Alaska, municipal wastewater treatment plant, provided with cyanide as the sole carbon and nitrogen source (White and Schnabel 1998). Microorganisms consumed cyanide and produced ammonia in an approximate 1:1 molar yield, reducing initial concentrations of 20.0 mg CN/L to <0.5 mg/L. When supplied with glucose, excess ammonia was readily consumed. This process may have application as a mobile system in the treatment of leachate from cyanidation extraction of gold from ores (White and Schnabel 1998).

Cyanide degradation has also been reported in various strains of cyanide-resistant yeasts isolated from wastewaters of gold mining operations. One strain of *Rhodotorula rubra* was able to use ammonia generated from abiotic cyanide degradation as its sole nitrogen source in the presence of a reducing sugar in aerobic media at pH 9.0 (Linardi et al. 1995). Similar results are reported for strains of *Cryptococcus* sp., *Rhodotorula glutinis*, *R. mucilaginosa*, and *Cryptococcus flavus* isolated from samples of Brazilian gold ores and industrial effluents (Gomes et al. 1999; Rezende et al. 1999).

In soils, cyanide seldom remains biologically available because it is either complexed by trace metals, microbially metabolized, or lost through volatilization (Towill et al. 1978: Marrs and Ballantyne 1987). Cyanide ions are not strongly adsorbed or retained on soils, and leaching into the surrounding groundwater will probably occur. Under aerobic conditions, cyanide salts in the soil are microbially degraded to nitrites or form complexes with trace metals. Under anaerobic conditions, cyanides denitrify to gaseous nitrogen compounds that enter the atmosphere. Mixed microbial communities that can metabolize cyanide and were not previously exposed to cyanide are adversely affected at 0.3 mg HCN/kg; however, these communities can become acclimatized to cyanide and then degrade wastes with higher cyanide concentrations.

Acclimatized microbes in activated sewage sludge can often convert nitriles to ammonia at concentrations as high as 60.0 mg total CN/kg (Towill et al. 1978).

In regard to cyanide use and toxicity on the recovery of gold and other precious metals, most authorities (as summarized in Eisler 1991, 2000; Eisler et al. 1999) currently agree on nine points:

1. Metal mining operations consume most of the current cyanide production.
2. The greatest source of cyanide exposure to humans and range animals is cyanogenic food plants and forage crops, not mining operations.
3. Cyanide is ubiquitous in the environment, with gold mining facilities only one of many sources of elevated concentrations.
4. Many chemical forms of cyanide are present in the environment, including free cyanide, metallocyanide complexes, and synthetic organocyanides, but only free cyanide (the sum of molecular hydrogen cyanide [HCN] and the cyanide anion [CN^-]) is the primary toxic agent, regardless of origin.
5. Cyanides are readily absorbed through inhalation, ingestion, or skin contact, and are readily distributed throughout the body via blood. Cyanide is a potent and rapid-acting asphyxiant; it induces tissue anoxia through inactivation of cytochrome oxidase, causing cytotoxic hypoxia in the presence of normal hemoglobin oxygenation.
6. At sublethal doses, cyanide reacts with thiosulfate in the presence of rhodanese to produce the comparatively harmless thiocyanate, most of which is excreted in the urine. Rapid detoxification enables animals to ingest high sublethal doses of cyanide over extended periods without adverse effects.
7. Cyanides are not mutagenic or carcinogenic.
8. Cyanide does not biomagnify in food webs or cycle extensively in ecosystems, probably because of its rapid breakdown.
9. Cyanide seldom persists in surface waters owing to complexation or sedimentation, microbial metabolism, and loss from volatilization.

11.2 CYANIDE HAZARDS

Cyanide hazards to aquatic plants and animals, terrestrial vegetation, birds, and mammals from heap leach and milling gold mining operations are briefly reviewed.

11.2.1 Aquatic Ecosystems

Fish kills from accidental discharges of cyanide-containing gold mining wastes are common (Eisler et al. 1999; Eisler 2000). In one case, mine effluents containing cyanide from a Canadian tailings pond released into a nearby creek killed more than 20,000 steelhead (*Oncorhynchus mykiss*; Leduc et al. 1982). In Colorado, overflows of 760,000 L NaCN-contaminated water from storage ponds into natural waterways killed all aquatic life along 28 km of the Alamosa River (Alberswerth et al. 1989). In 1990, 40 million L of cyanide wastes from a gold mine spilled into the Lynches River in South Carolina from a breached containment pond after heavy rains, killing an estimated 11,000 fish (Greer 1993; Da Rosa and Lyon 1997). In 1995, 160,000 L cyanide solution from a gold mine tailings pond near Jefferson City, Montana, were released into a nearby creek with loss of all fish and greatly reduced populations of aquatic insects (Da Rosa and Lyon 1997). In August 1995, in Guyana, South America, a dam failed with the release of more than 3.3 billion L cyanide-containing gold mine wastes into the Essequibo River, the nations' primary waterway, killing fish for about 80 km and contaminating drinking and irrigation water (Da Rosa and Lyon 1997).

On January 30, 2000, a dike holding millions of liters of cyanide-laced wastewater gave way at a gold extraction operation in northwestern Romania (owned jointly by Australian and Romanian firms), sending a waterborne plume into a stream that flows into the Somes, a Tisza tributary that crosses into Hungary (Koenig 2000). At least 200 tons of fish were killed, and endangered European otters (*Lutra lutra*) and white-tailed sea eagles (*Haliaeetus albicilla*) that ate the tainted fish were threatened. After devastating the upper Tisza, the 50-km-long pulse of cyanide and heavy metals spilled into the Danube River in northern Yugoslavia, killing more fish before the now-dilute plume filtered into the Danube delta at the Black Sea, more than 1000 km and 3 weeks after the spill. This entire ecosystem was previously heavily contaminated by heavy metals from mining activities (Kovac 2000). Villages close to the accident were provided with alternate water sources. Hungarian officials were most concerned that heavy metals in the Tisza River might enter flooded agricultural areas, with subsequent accumulation by crops and entry into the human food chain (Kovac 2000).

In Zimbabwe, where gold mining is the primary mining activity, tailings from the cyanidation process are treated to ensure that cyanide concentrations in the receiving waters are <5 μg CN^-/L (Zaranyika et al. 1994). Effluents from two gold mines in Zimbabwe, where gold is extracted by the cyanide process, contained 210 and 2600 mg CN^-/L, respectively. However, cyanide levels in the receiving stream were much lower at 2.1 μg CN^-/L and <0.2 μg/L at 500 and 1000 meters, respectively, downstream from the point where effluents entered the receiving body of water (Zaranyika et al. 1994).

Data on the recovery of poisoned ecosystems were scarce. In one case, a large amount of cyanide-containing slag entered a stream from the reservoir of a Japanese gold mine as a result of an earthquake (Yasuno et al. 1981). The slag covered the stream bed for about 10 km from the point of rupture, killing all stream biota; cyanide was detected in the water column for only 3 days after the spill. Within 1 month, flora was established on the silt covering the above-water stones, but there was little underwater growth. After 6 to 7 months, populations of fish, algae, and invertebrates had recovered, although the species composition of algae was altered (Yasuno et al. 1981).

Fish are the most cyanide-sensitive group of aquatic organisms tested. Under conditions of continuous exposure, adverse effects on swimming and reproduction usually occurred between 5.0 and 7.2 μg free CN/L and on survival between 20 and 76 μg/L (Eisler 1991, 2000). Reproductive impairment in adult bluegills (*Lepomis macrochirus*) occurred following exposure to 5.2 μg CN/L for 289 days (USEPA 1989). Concentrations of 10 μg HCN/L caused developmental abnormalities in embryos of Atlantic salmon (*Salmo salar*) after extended exposure (Leduc 1978). These abnormalities, which were absent in controls, included yolk sac dropsy and malformations of eyes, mouth, and vertebral column (Leduc 1984). Exposure of naturally reproducing female rainbow trout (*Oncorhynchus mykiss*) to 10 μg HCN/L for 12 days during the onset of the reproductive cycle produced a reduction in plasma vitellogenin levels and a reduction in ovary weight; vitellogenin is a major source of yolk (Ruby et al. 1986). Oocyte growth was reduced in female rainbow trout (Ruby et al. 1993a) and spermatocyte numbers decreased in males (Ruby et al. 1993b) following exposure to 10 μg HCN for 12 days. Free cyanide concentrations as low as 10 μg/L can rapidly and irreversibly impair the swimming ability of salmonids in well-aerated water (Doudoroff 1976). Exposure of fish to 10 μg HCN/L for 9 days was sufficient to induce extensive necrosis in the liver, although gill tissue showed no damage. Intensification of liver histopathology was evident at dosages of 20 and 30 μg HCN/L and exposure periods up to 18 days (Leduc 1984). Other adverse effects on fish of chronic cyanide exposure included susceptibility to predation, disrupted respiration, osmoregulatory disturbances, and altered growth patterns. Free cyanide concentrations between 50 and 200 μg/L were fatal to sensitive fish species over time, and concentrations >200 μg/L were rapidly lethal to most species of fish (USEPA 1989). The high tolerance of mudskippers (*Boleophthalmus boddaerti*; 96-hour LC50 of 290 μg/L) and perhaps other species of teleosts is attributed to a surplus of cytochrome oxidase and inducible cyanide-detoxifying mechanisms and not to a reduction in metabolic rate or an enhanced anaerobic metabolism (Chew and Ip 1992).

Fish retrieved from cyanide-poisoned environments, dead or alive, can probably be consumed by humans because muscle cyanide residues were considered to be lower than the currently recommended value of 50 mg/kg diet for human health protection (Leduc 1984; Eisler 2000). Cyanide concentrations in fish from streams poisoned with cyanide ranged between 10 and 100 μg total CN/kg whole-body fresh weight (FW) (Wiley 1984). Gill tissues of cyanide-exposed salmonids contained from 30 to >7000 μg/kg FW under widely varying conditions of temperature, nominal water concentrations of free cyanide, and duration of exposure (Holden and Marsden 1964). Unpoisoned fish usually contained <1 μg total CN/kg FW in gills, although values up to 50 μg/kg FW occurred occasionally. Lowest cyanide concentrations in gill occurred at elevated (summer) water temperatures; at lower temperatures, survival was greater and residues were higher (Holden and Marsden 1964).

Among aquatic invertebrates, adverse nonlethal effects occurred between 18 and 43 μg/L, and lethal effects between 30 and 100 μg/L although some deaths occurred between 3 and 7 μg/L for the amphipod *Gammarus pulex* (Eisler 2000). Aquatic plants are comparatively tolerant to cyanide; adverse effects occurred at >160 μg free CN/L (Eisler 2000). Adverse effects of cyanide on aquatic plants are unlikely at concentrations that cause acute effects to most species of freshwater and marine fishes and invertebrates (USEPA 1980).

Biocidal properties of cyanide in aquatic environments are modified by water pH, temperature, and oxygen content; life stage, condition, and species assayed; previous exposure to cyanides; presence of other chemicals; and initial dose tested (Eisler et al. 1999; Eisler 2000). There is general agreement that cyanide is more toxic to freshwater fishes under conditions of low dissolved oxygen; that pH levels within the range 6.8 to 8.3 have little effect on cyanide toxicity but enhance toxicity at more acidic pH; that juveniles and adults are the most sensitive life stages and embryos and sac fry the most resistant; and that substantial interspecies variability exists in sensitivity to free cyanide (Eisler et al. 1999; Eisler 2000). Initial dose and water temperature modify the biocidal properties of HCN to freshwater teleosts. At low lethal concentrations near 10 μg HCN/L, cyanide is more toxic at lower temperatures; at high, rapidly lethal HCN concentrations, cyanide is more toxic at elevated temperatures (Kovacs and Leduc 1982a, 1982b; Leduc et al. 1982; Leduc 1984). By contrast, aquatic invertebrates are most sensitive to HCN at elevated water temperatures, regardless of dose (Smith et al. 1979).

Season and exercise modify the lethality of HCN to juvenile rainbow trout; higher tolerance to cyanide was associated with higher activity induced by exercise and higher temperatures, suggesting a faster detoxification rate or higher oxidative and anaerobic metabolism (McGeachy and Leduc 1988). Low levels of cyanide that are harmful when applied constantly may be harmless under seasonal or other variations that allow the organism to recover and detoxify (Leduc 1981). Acclimatization by fish to sublethal levels of cyanide through continuous exposure was thought to enhance their resistance to potentially lethal concentrations, but studies with Atlantic salmon and rainbow trout were inconclusive (Kovacs and Leduc 1982a; Alabaster et al. 1983).

Cyanides seldom persist in aquatic environments (Leduc 1984). In small, cold oligotrophic lakes treated with NaCN (1 mg/L), acute toxicity to aquatic organisms was negligible within 40 days. In warm shallow ponds, no toxicity was evident to aquatic organisms after application of 1 mg NaCN/L. In rivers and streams, cyanide toxicity fell rapidly on dilution (Leduc 1984). Cyanide was not detectable in water and sediments of Yellowknife Bay, Canada, between 1974 and 1976 despite the continuous input of cyanide-containing effluents from an operating gold mine. Nondetection was attributed to rapid oxidation (Moore 1981).

Several factors contribute to the rapid disappearance of cyanide from water: bacteria and protozoans may degrade cyanide by converting it to carbon dioxide and ammonia; chlorination of water supplies can result in conversion to cyanate; an alkaline pH favors oxidation by chlorine; and an acidic pH favors volatilization of HCN into the atmosphere (USEPA 1980).

Cyanide interacts with other chemicals, and knowledge of these interactions is important in evaluating risk to living resources. Additive, or more than additive, toxicity of free cyanide to aquatic fauna may occur in combination with ammonia (Smith et al. 1979; Alabaster et al. 1983) or arsenic (Leduc 1984). Formation of the nickel-cyanide complex markedly reduced the toxicity of both cyanide and nickel at high concentrations in alkaline pH; at lower concentrations and acidic pH, nickel-cyanide solutions increased in toxicity by more than 1000 times, owing to dissociation of the metallocyanide complex to form hydrogen cyanide (Towill et al. 1978). In 96-hour bioassays with fathead minnows, *Pimephales promelas*, lethality of mixtures of sodium cyanide and nickel sulfate were influenced by water alkalinity and pH. LC50 values decreased with increasing alkalinity and increasing pH, being 0.42 mg CN/L at 5 mg $CaCO_3$/L and pH 6.5, to 730 mg CN/L at 192 mg $CaCO_3$/L and pH 8.0 (Doudoroff 1976).

11.2.2 Birds

Cyanide waste solutions following gold extraction are released into the environment to form ponds, sometimes measuring hundreds of hectares in surface area. In the U.S., these ponds are often located in arid regions of western states and attract wildlife including migratory birds (Pritsos and Ma 1997). Between 1983 and 1992, at least 1018 birds representing 47 species were killed when they drank cyanide-poisoned water from heap leach solution ponds at a gold mine in the Black Hills of South Dakota (Da Rosa and Lyon 1997); in 1995, heap leach ponds from this site overflowed after heavy rains, spilling into a nearby creek with fatal results to all resident fishes (Da Rosa and Lyon 1997). Many species of migratory birds, including waterfowl, shorebirds, passerines, and raptors, were found dead in the immediate vicinity of gold mine heap leach extraction facilities and tailings ponds, presumably as a result of drinking the cyanide-contaminated waters (Clark and Hothem 1991; Henny et al. 1994; Hill and Henry 1996; Da Rosa and Lyon 1997). About 7000 dead birds, mostly waterfowl and songbirds, were recovered from cyanide extraction gold mine leach ponds in the western U.S. between 1980 and 1989; no gross pathological changes related to cyanide were observed in these birds at necropsy (Allen 1990; Clark and Hothem 1991). No gross pathology was evident in cyanide-dosed birds (Wiemeyer et al. 1986), which is consistent with laboratory studies with cyanide and other animal groups tested and examined (Eisler 2000). In one case, waterfowl deaths were recorded in cyanide-containing ponds of an operating gold mine located in western Arizona shortly after the mine began operations in 1987 (Sturgess et al. 1989). Deaths ranged from single birds to flocks of more than 70. At least 33 species of birds, including waterfowl, wading birds, gulls, raptors, and songbirds, and three species of mammals (bats, fox) were found dead in these ponds. Most of the waterfowl deaths were located in desert areas where the nearest water was 8 to 80 km distant.

To protect wildlife, various techniques were used, including cyanide recovery, cyanide destruction, physical barriers, hazing, and establishment of decoy ponds. Techniques that were 92% successful (i.e., 8% mortality) cost mine owners about \$8.58 per dead bird. This 92% survival was considered unsatisfactory by the U.S. Bureau of Land Management, and mine owners were forced to spend \$295 for each dead bird found to reach 99% protection. Under existing legislation, however, zero mortality (100% survival) is the only acceptable solution (Sturgess et al. 1989). It is probable that 100% protection may not be possible using the best available technology. Songbird deaths were associated with hardrock gold mining in the Black Hills, South Dakota (Parrish 1989). This operation used the cyanide heap leaching process. Exposed collection ditches resembled small streams and were particularly attractive to songbirds, mostly red crossbills (*Loxia curvirostrata*) and pine siskins (*Carduelis pinus*), with fatal results. These ditches are now covered to prevent wildlife contact. Ponds containing cyanide solution were found to attract migrant waterfowl, and flagging devices were installed to dissuade waterfowl from landing, with partial success (Parrish 1989).

Free cyanide levels associated with high avian death rates have included 0.12 mg/L in air, 2.1 to 4.6 mg/kg body weight (BW) via acute oral exposure, and

Table 11.2 Single Oral Dose Toxicity of Sodium Cyanide (mg NaCN/kg body weight) Fatal to 50% of Selected Birds and Mammals (Listed from Most Sensitive to Most Tolerant)

Species	Oral LD50 (95% Confidence Limits)	Reference[a]
Mallard, *Anas platyrhynchos*	2.7 (2.2–3.2)	1
Human, *Homo sapiens*	3.0 estimated	2
American kestrel, *Falco sparverius*	4.0 (3.0–5.3)	3
Coyote, *Canis latrans*	4.1 (2.1–8.3)	4
Black vulture, *Coragyps atratus*	4.8 (4.4–5.3)	3
Laboratory rat, *Rattus norvegicus*	5.1–6.4	5,6
Little brown bat, *Myotis, lucifugus*	8.4 (5.9–11.9)	7
Eastern screech-owl, *Otus asio*	8.6 (7.2–10.2)	3
House mouse, *Mus musculus*	8.7 (8.2–9.3)	7
Japanese quail, *Coturnix japonica*	9.4 (7.7–11.4)	3
European starling, *Sturnus vulgaris*	17 (14–22)	3
Domestic chicken, *Gallus domesticus*	21 (12–36)	3
White-footed mouse, Peromyscus leucopus	28 (18–43)	7

[a] 1, Henny et al. 1994; 2, Way 1981; 3, Wiemeyer et al. 1986; 4, Sterner 1979; 5, Ballantyne 1987; 6, Egekeze and Oehme 1980; 7, Clark et al. 1991.

1.3 mg/kg BW administered intravenously. In cyanide-tolerant species, such as the domestic chicken (*Gallus domesticus*), dietary levels of 135 mg total CN/kg ration resulted in growth reduction of chicks, but 103 mg total CN/kg ration had no measurable effect on these chicks (Eisler 1991; Hill and Henry 1996). First signs of cyanide toxicosis in sensitive birds appeared between 0.5 and 5 minutes post-exposure, and included panting, eye blinking, salivation, and lethargy (Wiemeyer et al. 1986). In more tolerant species, signs of toxicosis began 10 minutes post-exposure. At higher doses, breathing in all species tested became increasingly deep and labored, followed by gasping and shallow intermittent breathing. Death usually followed in 15 to 30 minutes, although birds alive at 60 minutes frequently recovered (Wiemeyer et al. 1986). The rapid recovery of some cyanide-exposed birds may be due to the rapid metabolism of cyanide to thiocyanate and its subsequent excretion. Species sensitivity to cyanide seems to be associated with diet, with birds that feed predominantly on flesh being more sensitive to NaCN than species that feed mainly on plant materials, with the possible exception of mallards (*Anas platyrhynchos*), as judged by acute oral LD50 values (Table 11. 2).

Some birds may not die immediately after drinking lethal cyanide solutions. Sodium cyanide rapidly forms free cyanide in the avian digestive tract (pH 1.3 to 6.5), whereas formation of free cyanide from metal cyanide complexes is comparatively slow (Huiatt et al. 1983). A high rate of cyanide absorption is critical to acute toxicity, and absorption may be retarded by the lower dissociation rates of metal–cyanide complexes (Henny et al. 1994). In Arizona, a red-breasted merganser (*Mergus serrator*) was found dead 20 km from the nearest known source of cyanide, yet its pectoral muscle tissue tested positive for cyanide (Clark and Hothem 1991). A proposed mechanism to account for this phenomenon involves weak acid dissociable (WAD) cyanide compounds. Cyanide bound to certain metals, usually copper, is dissociable in weak acids such as stomach acids. Clark and Hothem (1991) suggested that drinking of lethal cyanide solutions by animals may not result in immediate death if the cyanide level is sufficiently low; these animals may die later when additional cyanide is liberated by stomach acid. In Canada, regulations typically require measurement of total cyanide and WAD cyanide in mine effluents (Ripley et al. 1996). More research seems needed on WAD cyanide compounds and delayed mortality.

Cyanide is a respiratory poison because of its affinity for the cytochrome oxidase complex of the mitochondrial respiratory chain (Keilin 1929; Nicholls et al. 1972). High dosages of cyanide are lethal through inhibition of cytochrome oxidase via cessation of mitochondrial respiration and depletion of ATP (Jones et al. 1984). Mallards given single oral doses of KCN (1.0 mg KCN/kg BW) at cyanide concentrations and amounts similar to those found at gold mine tailings ponds (40 mg CN/L), although it is NaCN that is used almost exclusively in mining, had elevated concentrations of creatine kinase in serum, suggesting tissue damage (Pritsos and Ma 1997). At 0.5 mg KCN/kg BW, mitochondrial function, an indicator of oxygen consumption, and ATP concentrations were significantly depressed in heart, liver, and brain (Ma and Pritsos 1997). Rhodanese and 3-mercaptopyruvate sulfurtransferase, two enzymes associated with cyanide detoxification, were induced in brain but not in liver or heart of KCN-dosed mallards. Although cyanide concentrations as high as 2.0 mg KCN/kg BW (at 80 mg CN/L) were not acutely toxic to mallards, the long-term effects of such exposures were not determined and may have serious consequences for migratory birds exposed sublethally to cyanide at gold mine tailings ponds.

Under the Migratory Bird Treaty Act, cyanide-containing ponds must be maintained at a level that does not result in deaths of migratory birds (Pritsos and Ma 1997). At present, there is negligible mortality of most avian species at ponds maintained at 50 mg CN^-/L. However, some deaths of migratory birds have been recorded at <50 mg CN^-/L, and sublethal effects have been demonstrated in mallards in water containing 20 mg CN^-/L. These effects include significant decreases in excised liver and brain tissue ATP levels and significant decreases in mitochondrial respiration rates in heart, liver, and brain tissues. It is clear that water containing <50 mg CN^-/L can cause generalized tissue damage in birds (Pritsos and Ma 1997), and this needs to be addressed in future regulatory actions.

11.2.3 Mammals

Gold and silver mining are probably the most widespread sources of anthropogenic cyanides in critical wildlife habitat, such as deserts in the western United States (Hill and Henry 1996). Between 1980 and 1989, 519 mammals, mostly rodents (35%) and bats (34%), were found dead at cyanide extraction gold mine mill tailings and heap leach ponds in California, Nevada, and Arizona (Clark and Hothem 1991). The list also included coyote (*Canis latrans*), badger (*Taxidea taxus*), beaver (*Castor canadensis*), mule deer (*Odocoileus hemionus*), blacktail jackrabbit (*Lepus californicus*), and kit fox (*Vulpes macrotis*), as well as skunks, chipmunks, squirrels, and domestic dogs, cats, and cattle. Also found dead at these same ponds were 38 reptiles, 55 amphibians, and 6997 birds. At the time of this study (1980 to 1989), there were approximately 160 cyanide extraction gold mines operating in California, Arizona, and Nevada, and these mines were operating within the geographic ranges of 10 endangered, threatened, or otherwise protected species of mammals. Bats comprised 6 of the 10 listed species. Because bats were not identified to species, members of these six protected species could have been among the 174 reported dead bats (Clark and Hothem 1991). A population of Townsend's big-eared bats (*Plecotus townsendii*), one of the 10 protected species, may have been extirpated by cyanide at a nearby mine in California, as quoted in Eisler et al. (1999). Badgers were another of the 10 protected species; 6 were counted among the 519 mammals found dead.

A vat leach gold mine in South Carolina with a large tailings pond reported 271 dead vertebrates found in the immediate vicinity between December 1988 and the end of 1990; 86% were birds, 13% mammals (29 of the 35 dead mammals were bats) and the rest reptiles and amphibians (Clark 1991). Bighorn sheep (*Ovis canadensis*) were found dead in August 1983 on a cyanide heap leach pile in Montana; in 1991, gulls died after landing on an unnetted cyanide pond, and deer died after consuming cyanide solution that had trickled beneath a fence (Da Rosa and Lyon 1997).

In Nevada, the state with the most heap leach sites, cyanide spills occurred weekly during the 1980s (Greer 1993). In South Dakota, a company's state-of-the-art leach pond was leaking cyanide solution at the rate of 19,000 L daily. Also, some companies allegedly punched holes in the heap leach liner when mining ended to allow drainage for more than 1 billion L of cyanide solution (Greer 1993).

In 1983, the drinking water supply of a Montana community was contaminated with 600,000 L cyanide-containing wastes from a gold mine tailings pond (Da Rosa and Lyon 1997). In 1986, an additional 7500 L leached from this same site and allegedly was responsible for the death of five cows. In 1994, cyanide was discovered in a residential drinking water supply near a gold ore processing facility in Montana. The cyanide had leaked from the mill's wastewater ponds located upgradient of the community (Da Rosa and Lyon 1997). In 1989, 350,000 L of cyanide solution spilled from a leach unit in California and polluted a reservoir used for municipal, recreational, and agricultural purposes (Greer 1993). In 1986, one mine operator in Montana dumped 76 million liters of treated cyanide solution onto 8 ha of land when a solution pond threatened to overflow after a rainstorm, with resultant contamination of a nearby creek (Da Rosa and Lyon 1997). And on February 22, 1994, 14 people were drowned when a tailings dam collapsed during a rainstorm in South Africa, releasing a wave of tailings and mine sediments into housing occupied by gold mine workers (Da Rosa and Lyon 1997).

Signs of acute cyanide poisoning in livestock usually occur within 10 minutes and include initial excitability with muscle tremors, salivation, lacrimation, defecation, urination, and labored breathing, followed by muscular incoordination, gasping, and convulsions; death may occur quickly, depending on the dose administered (Towill et al. 1978; Cade and Rubira 1982). Acute oral LD50 values for representative species of mammals ranged between 4.1 and 28.0 mg HCN/kg BW and overlapped those of birds (see Table 11.2). Despite the high lethality of large single exposures, repeated sublethal doses, especially in diets, are tolerated by many species for extended periods, perhaps indefinitely (Eisler 1991). Livestock found dead near a cyanide disposal site had been drinking surface water runoff that contained up to 365 mg HCN/L (USEPA 1980). Rats exposed for 30 days to 100 or 500 mg KCN/L drinking water had mitochondrial dysfunction, depressed ATP concentrations in liver and heart, and a depressed growth rate; little effect was observed at 50 mg KCN/L (Pritsos 1996). The adverse effect on growth is consistent with the biochemical indicators of energy depletion. However, the concentrations should be viewed with caution as CN may have volatilized from the water solutions before ingestion by the rats, due to presumed neutral pH.

Hydrogen cyanide in the liquid state can readily penetrate the skin (Homan 1987). Skin ulceration has been reported from splash contact with cyanides among workers in the electroplating and gold extraction industries, although effects in those instances were more likely due to the alkalinity of the aqueous solutions (Homan 1987). In one case, liquid HCN ran over the bare hand of a worker wearing a fresh air respirator; he collapsed into unconsciousness in 5 minutes, but ultimately recovered (USEPA 1980). No human cases of illness or death caused by cyanide in water supplies are known (USEPA 1980). Accidental acute cyanide poisonings in humans are rare (Towill et al. 1978); however, a male accidentally splashed with molten sodium cyanide died about 10 hours later (Curry 1963).

11.2.4 Terrestrial Flora

Mixed microbial populations capable of metabolizing cyanide and not previously exposed to cyanide were adversely affected at 0.3 mg HCN/kg substrate; however, these populations can become acclimatized to cyanide and can then degrade wastes containing cyanide concentrations as high as 60 mg/kg (Towill et al. 1978).

Cyanide metabolism in higher plants involves amino acids, *N*-hydroxyamino acids, aldoximes, nitriles, and cyanohydrins (Halkier et al. 1988). Cyanide is a weak competitive inhibitor of green bean (*Phaseolus vulgaris*) lipooxygenase, an enzyme that catalyzes the formation of hydroperoxides from polyunsaturated acids (Adams 1989). In higher plants, elevated cyanide concentrations inhibited respiration through iron complexation in cytochrome oxidase, and ATP production and other processes dependent on ATP (Towill et al. 1978). At lower concentrations, effects include inhibition of germination and growth, although sometimes cyanide enhances seed germination by stimulating the pentose phosphate pathway and inhibiting catalase (Solomonson 1981). The detoxification mechanism of cyanide is mediated by rhodanese, an enzyme widely distributed in plants (Solomonson 1981; Leduc 1984). The rate of production and release of cyanide by plants to the environment through death and decomposition is unknown (Towill et al. 1978).

11.3 CYANIDE MITIGATION AND RESEARCH NEEDS

Aquatic birds are naturally attracted to large open ponds, and efforts to deter or chemically repel them have been generally ineffective (Hill and Henry 1996). However, some chemical repellents showed promise at reducing consumption of dump leachate pond water when tested on European starlings (*Sturnus vulgaris*), especially *o*-aminoacetophenone and 4-ketobenztriazine (Clark and Shah 1993). Exclusion from cyanide solutions or reductions of cyanide concentrations to nontoxic levels are the only certain methods of protecting avian and mammalian wildlife from cyanide poisoning (Henny et al. 1994). Mortality of migratory birds from cyanide toxicosis may be curtailed at small ponds associated with leach heaps by screening birds from toxic solutions (Hallock 1990). Fencing and covering of small solution ponds with polypropylene netting have proved effective for excluding most birds, bats, and larger mammals, provided that the fencing and netting are properly maintained (Henny et al. 1994). Fences installed around cyanide-containing ponds at a heap leach gold mine in South Dakota successfully prevented deer and elk from entering; this, and other practices, reduced overall wildlife mortality at this site by about 95% (Parrish 1989). Reclamation of leach heaps to establish suitable wildlife areas is ongoing by mining corporations and involves mechanical creation of new wildlife habitats of slopes, ledges, and crevices, and revegetation of these habitats, usually with native plants, but sometimes with introduced species (Parrish 1989).

A few mines in Nevada are now covering surfaces of small ponds with 4-inch (10.2-cm)-diameter, high-density polyethylene balls (Eisler et al. 1999); birds are no longer attracted to these ponds as water sources. Although initial costs of the balls are higher than installation of netting, there are no maintenance expenses for the balls, whereas netting needs continual maintenance. Gold mine operators in southern California and Nevada used plastic sheeting to cover the cyanide leach pond, resulting in a cessation of wildlife mortality. The comparatively high cost of this process was soon recouped through reduced evaporation of water and cyanide (Eisler et al. 1999).

Cyanide concentrations in the water column of mill tailings ponds (160 to 207 mg/L) were reduced at one Nevada site using naturally detoxified recycled tailings water (Henny et al. 1994). Lowering the cyanide concentrations in tailings ponds with hydrogen peroxide has been successful at a few mines in Nevada (Allen 1990), but this procedure was still preliminary (Clark and Hothem 1991). To reduce the potential for puddling on ore heaps, ores should be less compacted; this can be accomplished by reducing the clay content of the ores and stacking ores using conveyer belts rather than trucks (Henny et al. 1994). Puddling can also be reduced by careful monitoring of solution application rates and maintenance of solution distribution systems. Wildlife have been excluded from leaching solution on the heaps by substituting drip lines for sprinklers and covering the drip lines with a layer of gravel (Henny et al. 1994; Hill and Henry 1996). Some mines use small net panels over areas of puddling to exclude birds (Henny et al. 1994).

Water hyacinth (*Eichorinia crassipes*) has been proposed as the basis of a cyanide removal technology, although large-scale use has not been implemented. Hyacinths can survive for at least 72 hours in a nutrient solution containing up to 300 mg CN/L and can accumulate up to 6700 mg CN/kg DW plant material. On this basis, 1 ha of hyacinths has the potential to absorb 56.8 kg cyanide in 72 hours, and this property may be useful in reducing the level of CN in untreated wastewaters where concentrations generally exceed 200 mg CN/L (Low and Lee 1981). Large-scale use of water hyacinths for this purpose has not yet been implemented, possibly due to disagreement over appropriate disposal mechanisms (Eisler et al. 1999).

A model biological treatment plant was established to remove cyanides and toxic metals from a daily discharge of 15,000,000 L wastewater of an underground gold mine in South Dakota (Whitlock 1990). Wastes from this mine, along with wastes from other mines, industry, mining camps, and municipalities had been discharged for more than 114 years into a common drainage system, which became devoid of all life. In the 1970s, all tailings were impounded and, in cooperation with the U.S. Environmental Protection Agency (USEPA), the mine owners developed a technology to reduce cyanide and heavy metals by 95 to 98%. The formerly lifeless receiving stream now supports an established trout fishery. In the process, cyanide was degraded to carbon and nitrogen; ammonia to nitrate; and heavy metals removed by adsorption or absorption. Indigenous species of *Pseudomonas* bacteria were used in rotating biological contactors to accomplish the process. Effluent total and WAD cyanide steadily declined between 1984 (0.45 mg total CN/L; 0.09 WAD cyanide) and 1990 (0.06 mg total CN/L; <0.01 mg WAD cyanide/L), and the cost per cubic meter of effluent declined from $0.20 to $0.10 (Whitlock 1990).

Free cyanide criteria currently proposed for the protection of natural resources include <3 μg/L medium for aquatic life and <100 mg/kg diet for birds and livestock (Eisler 2000). For human health protection, free cyanide values are <10 μg/L drinking water, <50 mg/kg diet, and <5 mg/m^3 air (Eisler 2000). Additional research is needed to establish legally enforceable standards and threshold limit values for potentially toxic cyanides in various forms, including HCN and inorganic cyanide. More research is merited on low-level, long-term cyanide intoxication in birds and mammals by oral and inhalation routes in the vicinities of high cyanide concentrations, especially on the incidence of nasal lesions, thyroid dysfunction, and urinary thiocyanate concentrations (Towill et al. 1978; Egekeze and Oehme 1980). Research is also needed on threshold limits in water where birds and mammals may be exposed, including the role of CN–metal complexes, and on sublethal effects of free cyanide on vertebrate wildlife (Eisler et al. 1999). In aquatic systems, research is needed on (1) long-term effects of low concentrations of cyanide on growth, survival, metabolism, and behavior of a variety of aquatic organisms (Towill et al. 1978; Leduc et al. 1982; Eisler 1991); (2) adaptive resistance to cyanide and the influence, if any, of oxygen, pH, temperature, and other environmental variables (Leduc 1981, 1984); and (3) usefulness of various biochemical indicators of cyanide poisoning, such as cytochrome oxidase inhibition (Gee 1987) and vitellogenin levels in fish plasma (Ruby et al. 1986).

Analytical methodologies need to be developed that differentiate between free cyanide (HCN and CN^-) and other forms of cyanide, and that are simple, sensitive (i.e., in the μg/L range), and accurate (Smith et al. 1979; Leduc et al. 1982). Procedures need to be standardized that ensure prompt refrigeration and analysis of all samples for cyanide determination because some stored samples generate cyanide while others show decreases (Gee 1987). Periodic monitoring of cyanide in waterways is unsatisfactory for assessing potential hazards because of cyanides's rapid action, high toxicity, and low environmental persistence. A similar case is made for cyanide in the atmosphere. Information is needed on the fate of cyanide compounds in natural waters, relative contributions of natural and anthropogenic sources, and critical exposure routes for aquatic organisms (as summarized in Eisler 2000).

Finally, the use of leaching solutions (lixiviants) other than cyanide should be considered as alternatives. Thiourea was first mentioned as a lixiviant at least 60 years ago (Fields 2001). In the presence of a suitable oxidizing agent, thiourea will solubilize gold to produce a positively charged Au^+ complex (Savvaidis 1998; Adams et al. 1999; Saleh et al. 2001) as follows:

$$Au + 2CS(NH_2)_2 \rightarrow Au[CS(NH_2)_2]^+ + e^-$$

The only reported full-scale thiourea leach is at a mine in Australia (Adams et al. 1999). Products in the effluent include thiourea, formamidine, disulfide, cyanamide, sulfur, nitrate, urea, carbon dioxide, ammonia, and sulfide (Adams et al. 1999). However, thiourea is listed by USEPA as a potential carcinogen and is also relatively long-lived in the environment (Adams et al. 1999; Fields 2001). Other commercial alternatives to cyanide for gold dissolution are various halogens, thiocyanate, and thiosulfate (Adams et al. 1999). All are considered less cost-effective than cyanide, however, and collectively present a variety of environmental problems. Other lixiviants for gold under consideration include polysulfide, various nitriles, and cyanamide. However, it does not seem likely that cyanide will be replaced in the near future as a lixiviant for gold. The environmental impact of cyanide is short term; it is the release and stabilization in solution of base metals that may have a longer-term impact (Adams et al. 1999).

11.4 WATER MANAGEMENT ISSUES

To provide the quantities of ore needed for heap leach facilities, large pits are dug. One prospective open-pit mine is expected to measure 0.9 km deep, 1.44 km wide, 2.4 km long, and involves more than 1 billion tons of rock (Greer 1993). Many mining pits intrude below the water table and must be continually pumped dry. After the mine closes, the pumping ceases and the pit fills to become a small lake. Pit lakes have the potential to become acidic and may eventually contain elevated concentrations of various elements. As the level of potentially toxic water rises, it can begin to infiltrate into groundwater (Fields 2001).

Open-pit mining allegedly disturbs 50 times more earth than underground mining to produce the same amount of gold. Open-pit mining is frequently associated with stripped vegetation and topsoil and potential loss of breeding, wintering, and feeding habitat for wildlife. If the pits are left unfilled, the ecological damage may be irreversible (Greer 1993). In Bolivia, the cyanidation method of gold extraction uses 3 million liters of water daily to treat 17,000 tons of minerals, with resultant destruction of agricultural and grazing lands and loss of at least two lakes (Garcia-Guinea and Harffy 1998).

The combination of open-pit mining and heap leaching and milling generates large quantities of waste soil and rock overburden and residual tailings water from ore concentration. Surface mining of gold may generate two to five times more waste as it does ore, and up to 90% of this ore ends up as tailings. The wastes, especially the tailings, may contain residual cyanide, acids, organic toxicants, nitrogen compounds, and oils, as well as iron, copper, zinc, lead, arsenic, nickel, mercury, and cadmium (Greer 1993).

Discharge of this waste into freshwater and marine ecosystems is an inexpensive but ecologically ill-advised disposal method. Suspended solids from mining and from waste processing cause turbidity, reducing light penetration and inhibiting photosynthesis. Fish gills are sensitive to the sharp-edged, irregular pieces of rock and become vulnerable to secondary invaders, such as fungi (Greer 1993). Disposal of tailings and overburden into marine environments is no longer practiced in the United States (Greer 1993). Australia forbids the dumping of tailings and overburden into rivers, and marine disposal in that country is being phased out. The situation is different in the less-industrialized countries, where the least expensive waste disposal route is used. In Papua New Guinea, for example, where gold production exceeds 100 tons annually (up from 34 tons in 1990), more than 600 million tons of tailings were discharged into the Kawerong/Jaba River system over a period of 20 years. All aquatic life in the Jaba River was destroyed, and this disaster was one of the reported causes of the island's civil war in 1989 (Da Rosa and Lyon 1997). Other river systems in Papua New Guinea are more severely affected, with subsequent contamination of the Gulf of Papua and the Torres Strait, with major impact on fish and shrimp stocks (Greer 1993).

Gold mine operators in Nevada and elsewhere are digging large open pits to reach extensive, deep deposits of low-grade gold ore (Plume 1995; Maurer et al. 1996; Da Rosa and Lyon 1997; Eisler and Wiemeyer 2004). To prevent flooding in the mine pits and to permit efficient earth moving of surface soils, it is necessary to withdraw groundwater and use it for irrigation, discharge it to rapid infiltration basins or, in some cases, discharge it into a nearby watercourse (USBLM 2000). Surface waters are diverted around surface mining operations. After cessation of mining operations and pumping, lakes will form in the open-pit mines dug to levels below that of the surrounding groundwater (Da Rosa and Lyon 1997). At least six of Nevada's open-pit mines have filled with water that does not meet federal criteria for the protection of human drinking water and aquatic life for heavy metals and acidity (Da Rosa and Lyon 1997).

11.4.1 Affected Resources

Miners usually pump water by using a combination of in-pit wells and perimeter wells. In-pit wells pump out water that has entered the mine site, and perimeter wells intercept groundwater before it can seep into the pit. The lowering of the water table decreases groundwater elevation kilometers away from the mining site (Da Rosa and Lyon 1997). In the Humboldt River basin, Nevada, gold miners divert pumped water to irrigate fields, to create wetlands in what is naturally desert, or to supply water to another user. Only a small fraction of the pumped water is restored to the original aquifer. Under Nevada law, miners must replace the lost water from dried-out springs or wells, deepen the well that has been dried out, or reduce pumping so that prior water levels are restored (Da Rosa and Lyon 1997). This subsection briefly reviews the potential effects of water management actions on resources in gold mining communities in northern Nevada beginning in 1992. Specific resources addressed include geological structures, groundwater and surface water resources, riparian areas and wetlands, terrestrial wildlife, aquatic habitat and fisheries, special status species, livestock grazing, socioeconomics, and Native American religious concerns (USNAS 1999; USBLM 2000).

Geological impacts were demonstrated by the formation of at least three sinkholes within 5 km of groundwater drawdown and discharge into the Humboldt River. The areas that could be susceptible to sinkhole development are generally undeveloped areas underlain by carbonate rock. Critical mine-related facilities such as waste rock storage facilities, heap leach pads, and mill and tailings facilities are not located within these areas (USBLM 2000). Ground subsidence may also occur as a result of groundwater withdrawals (USBLM 1996). Changes in groundwater levels caused by mining activities show that faults, mineralized zones, and differences in fracture permeability and bedrock lithology control groundwater flow in the mountain blocks, creating complex flow paths; effects of geological structures on groundwater flow become apparent when large stresses are placed on the system (Maurer et al. 1996).

As a result of mine dewatering operations, groundwater levels at the end of 1998 had decreased 110 to 466 meters in the vicinity of the three mines examined (Maurer et al. 1996; USBLM 2000). Several springs near the mines have dried up or shown a reduction in flow as a result of mine dewatering and drought (Plume 1995). The flow and vegetation in a nearby creek were significantly reduced (USNAS 1999; USBLM 2000). Reductions in the baseflow of perennial springs and streams could affect surface water rights within the drawdown area, directly impacting rights for irrigation, stock watering, domestic use, mining and milling, municipal, and other uses (Plume 1995; USBLM 2000). Surface discharge of excess mine dewatering water and other waters to the Humboldt River was initiated in 1992, with major cumulative discharges predicted during the years 1999 through 2006, and significant discharges in 2007 through 2011 (USBLM 2000). The largest percentage increase in discharge would occur during the late summer and fall months when flows were low. The increased Humboldt River flows would probably not create additional flooding upstream, with negligible long-term impacts on surface water rights within the Humboldt River Basin, an interior terminal basin with no outlet. Mine discharges into the Humboldt Sink at the terminus of the river on occasion have exceeded water quality regulations mandated for arsenic (since corrected) and have contributed increasing, but allowable, amounts of total dissolved solids, boron, copper, fluoride, and zinc. These increased loads may potentially result in increased concentrations of these chemicals in the sink wetlands where concentration by evaporation is of concern (USBLM 2000).

Changes in regional hydrology may affect wetlands, especially in arid regions (USNAS 1999). Many arid region wetlands develop at spring orifices, and the spring pools sometimes support threatened or endangered species. In Nevada, small changes in the hydrologic head may lower the water table several meters, causing destruction of springs and their associated wetlands (USNAS 1999). About 281 ha (618 acres) of riparian vegetation occur within the drainage areas where perennial waters could be reduced by groundwater drawdown (USBLM 2000). Elevated water tables from mine dewatering discharges in low-lying areas adjacent to the Humboldt River would be conducive to the establishment of riparian vegetation and wetlands plants; this situation would reverse when mine dewatering discharges cease (USBLM 2000). Metal-contaminated water and sediments that reach wetlands may create a contaminated substrate for plants that accumulate metals (USNAS 1999).

Contaminated soils and sediments from mine sites have the potential to affect bed, bank, and floodplain sediments, as well as riparian areas and wetlands downgradient from the mine. Terrestrial vegetation diversity may be altered by construction of roads or use of offroad vehicles related to development of mines (USNAS 1999). Small alterations of topography by mining exploration activities may create new habitats for hydrophytes in low areas and xerophytes on elevated terrain. Offroad vehicles alter the stability of wetlands by creating ruts that drain the water. The use of soft-tired vehicles can produce linear depressions that create pools and tend to dry up the remaining wetlands. Offroad vehicles used for exploration may carry propagules of nonnative plant species into relatively pristine areas. Invasion of nonnative species is enhanced through mining exploration, which may be the first mechanical intrusion into these areas (USNAS 1999).

Reduction in surface or subsurface flows could result in a reduction or loss of vegetation cover for wildlife. This, in turn, could reduce breeding, foraging, and cover habitats for wildlife; increase animal displacement and loss; reduce prey availability; reduce overall biological diversity; produce possible genetic isolation in Lahontan cutthroat trout (*Oncorhynchus clarki henshawi*); reduce carrying capacity for terrestrial wildlife; and might result in population declines (USBLM 2000). Incremental habitat loss would affect a variety of big game species, upland gamebirds, waterfowl, shorebirds, raptors, songbirds, nongame mammals, and area reptiles and amphibians. The eventual reduction in flows within artificially created wetlands in the study area would result in a transition back into an upland plant community with reduction in use by waterfowl. Increased leaching of minerals and salts from saturated soils into the soil surface and subsurface layers may, however, result in a plant community of salt-tolerant species that will eventually affect wildlife composition in the area (USBLM 2000). Water discharges into the Humboldt River would produce a net increase in water flows, increasing overall water availability for consumption, supporting riparian and wetland vegetation, restoring portions of wetland and marsh habitats, and producing additional breeding, foraging, and resting habitat for resident and migratory wildlife (USBLM 2000). These effects would be most pronounced during the late summer and fall months associated with low-flow periods. Seasonal flooding may cause habitat loss for nesting and foraging of some species, but would be offset by enhancement of existing backwater and slough areas. Increased annual flows may result in additional open water during winter, with increased foraging opportunities. After mine dewatering discharges cease, the effects on terrestrial wildlife are expected to be minimal.

Reduction in stream flows as a result of mine dewatering would directly impact habitat for aquatic resources, including periphyton and invertebrates that support native fish species (USBLM 2000). Habitat reductions could result in decreased biodiversity and biomass in these communities. If stream segments become dry as a result of reduced flows, aquatic habitat and associated biota would disappear

(USBLM 2000). Increased flows to the Humboldt River would increase the habitat for fish and their food organisms. However, the possible elimination of shallow pools and channels could decrease nursery ground habitat. Increased flows may alter the diversity of fish populations because introduced species may be able to disperse and use wider areas of the river and likely compete with native species (USBLM 2000). Increased sediment levels associated with increased flows may affect aquatic biota near outfalls.

Potential reductions in available water and riparian habitat may adversely affect sensitive species of shrews, bats, eagles, hawks, owls, grouse, ibis, and terns (USBLM 2000). Possible adverse effects include habitat loss, death of individuals, reduced prey availability, reduced diversity, genetic isolation, and population declines. However, mine dewatering activities would have no measurable effect on ospreys (*Pandion haliaetus*) and pelicans. Increased flows to the Humboldt River and the Humboldt Sink would provide additional nesting, brooding, foraging, and resting habitat for species that occur along the river, including sensitive species of wildlife and ospreys. The greater availability of open water areas could provide additional foraging habitat for wintering bald eagles (*Haliaeetus leucocephalus*) along the river. Reductions in flow to upgradient feeder streams may adversely affect habitat and survival of the Lahontan cutthroat trout, the Columbia spotted frog (*Rana luteiventris*), and the California floater (*Anodonata californiensis*), a freshwater mussel. Water level reductions may also occur in springs occupied by springsnails, some of which may be endemic (USBLM 2000).

The potential long-term loss of water sources may cause long-term loss of permitted active grazing use or affect livestock distribution and forage use within grazing allotments. In addition, perennial creeks located within these allotments could be affected by groundwater drawdown, which could affect grazing management operations (USBLM 2000). Slightly increased water levels within the Humboldt River during the mine dewatering period would probably increase the areal extent of wetlands adjacent to the river channel producing increased forage and increased water availability for livestock; this would reverse after mine dewatering ceases.

Reductions in groundwater levels may impose economic hardship on those using water for domestic, industrial, commercial, agricultural, and husbandry purposes (Plume 1995; USBLM 2000). Excess mine water discharged into the Humboldt River would constitute a positive effect to water rights holders in the basin, although adverse effects from increased flow may include flooding of irrigated fields during periods of high flow (USBLM 2000). If additional mine discharges occur during periods of high flow, the storage capacity of a reservoir may be exceeded, with flood damage expected to the reservoir gates and agricultural fields downstream from the reservoir. Predicted decreases in Humboldt River flow (about 2 to 3% of the average annual flow) from groundwater drawdown could extend to the year 2019. This decrease may limit some agricultural operations to irrigate late season hay or to water livestock. Specific irrigators with junior water rights may be the most seriously affected; however, mine operators have committed to augmenting low flows if necessary using senior water rights that they own or control (USBLM 2000).

Native Americans in the vicinity, primarily members of the Western Shoshone, believe that disruption of water resources would impact their lives and the spirit forces associated with these waters, plants, and animals to the extent that Shoshone cultural traditions could not be maintained (USBLM 2000).

11.4.2 Pit Lakes

In Nevada, very low concentrations of gold (0.7 g/metric ton) can be mined economically from bulk mineral deposits, and large open-pit mines have been and are currently being developed for this purpose (Shevenell 2000; Eisler and Wiemeyer 2004). Many of these mining operations in Nevada are extracting ore from below the water table and are withdrawing large volumes of water to maintain dry operating conditions. A single gold mine in Eureka County, Nevada, allegedly has pumped as much groundwater from its open mining pit in 1 year as the entire annual consumption of a city with a population of 500,000 (DaRosa and Lyon 1997). In 1994, gold mining operations in the Humboldt River basin withdrew enough water to supply all domestic water users in the greater Seattle area (population 1.1 million) over that same period (Da Rosa and Lyon 1997). After mining is completed and dewatering activities stopped, the pits will begin to fill with water with the ultimate lake surface approaching the elevation of the premining groundwater level (Shevenell 2000). Between 1992 and 2002, about 35 mines in Nevada had a lake in their open-pit mines after dewatering and cessation of mining. Of the existing pit lakes at eight different gold mines, most had near-neutral pH but exceeded drinking water standards for arsenic, sulfate, or total dissolved solids for at least one sampling period. Pit lakes have lower evaporation rates than natural lakes because they are at higher elevations, and the surface-to-depth ratio of pit lakes is more than 1000 times smaller than that of natural lakes. Shevenell (2000) concludes that the ultimate water quality and limnology of the pit lakes and their potential impact on wildlife have not been adequately evaluated.

Open-pit mining is common in the gold mining industry where the ore bodies are large and overburden depths are limited (Braun 2002). Open-pit depths typically extend below the groundwater and may exceed 250 meters beneath the ground surface; they usually require dewatering to access the ore body. As economic development of the pit continues, dewatering systems are continually expanded. At mine closure, dewatering ceases and groundwater inflow to the excavation begins. Pit lakes form when groundwater, surface water, and other postmining drainage accumulate inside inactive open pits below the groundwater level. Pit lake water quality varies considerably owing to variations in groundwater inflow, direct precipitation, and contact with pit wall precipitation or runoff water. Interactions between wall rock and groundwater, evaporation, and geochemical processes operating within the pit lake also affect water quality (Braun 2002). A potential worst-case scenario may occur where both inflow and evaporation are moderate and there is an outflow of evapoconcentrated water containing high concentrations of toxic constituents (Atkinson 2002).

Questions that commonly arise about pit lake hydrology concern the number and amounts of contaminants from groundwater and surface water retained in pit lakes over time and management of stormwater, other surface waters, and groundwater inputs into the lake (Moreno and Sinton 2002). Managing water resources at open-pit mines frequently involves modeling based on long-term analyses of flow rates, water levels, and geochemistry of pit lake waters. Modeling analysis of groundwater flow, for example, was helpful in predicting the range of flow rates, lake levels, and surrounding groundwater levels at different stages in a mine's development (Moreno and Sinton 2002). Risk assessment of pit lakes is recommended on a case-by-case basis because each pit lake is unique, depending on local hydrogeology, the size of the pit lake, and climatic conditions (Atkinson 2002).

When pit lakes exceed local or federal surface water or groundwater quality standards, three main options for mitigation are recommended: neutralize the pit lake in place through treatment; prevent the formation of a pit lake by pumping groundwater; and regulate the pit lake level at a certain height to prevent commingling with other aquifers (Kuipers 2002). Treatment options include physical treatment processes that consist mainly of screening and filtration techniques to remove particulate matter; chemical treatment, the most common, to raise pH through lime precipitation of sulfates, and other methods to remove contaminants, such as arsenic; and biological processes, including sulfate reduction processes to treat acid drainage and remove sulfates as metals. These treatment options are preferable to the use of clean water for dilution or relying on faulty land application disposal systems (Kuipers 2002).

Gold mining pit lakes in Nevada, when filled, will contain more water than all reservoirs combined within the borders of this arid state (Miller 2002). Pit lakes are important to Nevada, and their water quality will determine their future use, as well as their effects on the aquifer, wildlife, and ecosystems. Oxidation reactions of groundwater with walls and host rock of the pit release sulfate, acid, and metals into the lake. Although oxidation of the pit wall rock releases sulfuric acid, calcium carbonates present in the wall rock also dissolve and neutralize the acid. In many Nevada pit lakes, the water that enters the pit lake will be near neutral, while in others the pit lake pH may be as low as 2.9. Metal solubilities of cadmium, zinc, and copper are high at acidic pH; at acidic pit lakes with high metal loadings, concentrations as high as 2 mg Cd/L, 172 mg Cu/L, and 550 mg Zn/L are reported. Oxidation rates decline steeply as the pit fills and the water covers the reactive surfaces. Aggressive refilling of a pit lake by groundwater inflow and additional pumping may be useful in limiting oxidation, but the acidic water created in the surrounding host rock during dewatering will eventually flush into the lake, with long-term management required. Backfilling with waste or other rock is a costly management option, but should be considered if the infill material is calcareous, as is the case with many desert soils. If no neutralization capacity is available in the backfill material, low pH problems will present additional management problems. "Pit lakes represent an in-perpetuity commitment of groundwater resources and their management needs further study, particularly in arid climates where water is the limiting resource for agricultural and municipal development" (Miller 2002).

State positions on pit water quality issues vary; however, the death of migratory birds at a pit lake is of concern to the U.S. Fish and Wildlife Service, the lead agency in enforcing violations of the Migratory Bird Treaty Act (Braun 2002). Waters of the Berkeley pit in Butte, Montana, were lethal to lesser snow geese (*Chen caerulescens caerulescens*) that used the lake in 1995 (USNAS 1999). Nevada regulates pit lake water quality standards on a case-by-case basis, enforcing a state regulation that loosely states that pit lake water cannot degrade surrounding groundwater quality or adversely affect the health of human or terrestrial life (Bolen 2002). Other states may require that pit lake water conform with aquifer water quality standards (Arizona); be suitable for human drinking, cooking, and food processing purposes after conventional treatment (Montana); and meet livestock, agriculture, or domestic water quality standards, depending on geographic location (Wyoming). Most states now require a bond from mine operators, with amounts ranging from nominal to millions of dollars, depending on the state (Bolen 2002). Aquatic communities may also form in pit lakes; these organisms have the potential to biomagnify various compounds and trace elements from pit lake waters. If pit lakes become attractive to migratory birds or other species, these species will be exposed to the contaminants that may be present. The introduction of fish to pit lakes may, in some cases, present unacceptable risks of contaminant exposure to fish-eating birds.

"Perpetual management of mine pit lakes is an unavoidable component of future U.S. land management" (Kempton 2002). This goal is best accomplished in a research-focused, adaptive management framework based on trust funds, a central repository for reports, and formation of a technical management group to assimilate prediction and remediation information (Kempton 2002).

11.5 WATER QUALITY AND MANAGEMENT RESEARCH NEEDS

The long-term environmental impacts of pit lakes are poorly known at this time, and long-term predictions are currently made on the basis of short-term data. Accordingly, the relation between predicted and actual outcomes needs to be evaluated (USNAS 1999). Research is needed on the chemistry, hydrology, and biology of pit lakes and their surroundings to minimize the environmental influence of future pit lakes. Pit lakes now filling need to be monitored over time to evaluate lake chemistry changes. Studies are also needed on the potential development of biological communities in pit lakes and their influence on aquatic biota, and on avian and terrestrial wildlife (USNAS 1999).

Food chain effects in pit lakes should be determined before stable biological habitats are artificially established. Additional investigations are recommended to establish water quality criteria on selected metals for protection of aquatic life and for establishment of acceptable metal burdens in aquatic prey of migratory birds; this is of particular interest in pit lakes located in arid areas with increasing metals concentrations attributed to high evaporation rates.

More research is recommended on mine area dewatering and the discharge of surplus water, especially to surface waters. Water balance models for different hydrogeological settings need to be developed to address local and regional interrelations among surface flow, pit lake hydrology, and hydraulic head of shallow and deep aquifers (USNAS 1999). These models may permit long-term predictions of the consequences of alteration of surface waters and the interruption, use, and withdrawal of groundwater. Results of studies on mine water discharges into nearby watercourses, for example, mine water discharges into the Humboldt River in Nevada, will aid in providing long-term predictions of the response of riverine ecosystems to hydrologic and biological changes (USNAS 1999). Finally, a better understanding is needed of the risks associated with increased loads of various contaminants from mine watering discharges to surface waters that flow to important wetlands in terminal systems where evapoconcentration is a factor.

11.6 SUMMARY

Cyanide extraction of gold through milling of high-grade ores and heap leaching of low-grade ores requires cycling of millions of liters of alkaline water containing high concentrations of potentially toxic sodium cyanide (NaCN), free cyanide, and metal–cyanide complexes. Some milling operations result in tailings ponds of 150 ha and larger. Heap leach operations that spray or drip cyanide onto the flattened top of the ore heap require solution processing ponds of about 1 ha in surface area. Puddles of various sizes occur on the top of heaps, where the highest concentrations of NaCN are found. Exposed solution recovery channels are usually constructed at the base of leach heaps. All these cyanide-containing water bodies are hazardous to wildlife, especially migratory waterfowl and bats, if not properly managed. Accidental spills of cyanide solutions into rivers and streams have produced massive kills of fish and other aquatic biota. Freshwater fish are the most cyanide-sensitive group of aquatic organisms tested, with high mortality documented at free cyanide concentrations >20 μg/L and adverse effects on swimming and reproduction at >5 μg/L.

Exclusion from cyanide solutions or reductions of cyanide concentrations to nontoxic levels are the only certain methods of protecting vertebrate wildlife from cyanide poisoning; a variety of exclusion/cyanide reduction techniques are presented and discussed. Additional research is recommended on: (1) effects of low-level, long-term cyanide intoxication in birds and mammals by oral and inhalation routes in the vicinity of high cyanide concentrations; (2) long-term effects of low concentrations of cyanide on aquatic biota; (3) adaptive resistance to cyanide; and (4) usefulness of various biochemical indicators of cyanide poisoning.

To prevent flooding in mine open pits and to enable earth moving on a large scale, it is often necessary to withdraw groundwater and use it for irrigation, discharge it to rapid infiltration basins, or, in some cases, discharge it to surface waters. Surface waters are diverted around surface mining operations. Adverse effects of groundwater drawdown include formation of sinkholes within 5 km of groundwater drawdown; reduced stream flows with reduced quantities of water available for irrigation, stock watering, and domestic, mining and milling, and municipal uses; reduction or loss of vegetation cover for wildlife, with reduced carrying capacity for terrestrial wildlife; loss of aquatic habitat for native fishes and their prey; and disruption of Native American cultural traditions. Surface discharge of excess mine dewatering water and other waters to main waterways may contain excess quantities of arsenic, total dissolved solids, boron, copper, fluoride, and zinc. When mining operations cease and the water pumps are dismantled, these large open pits may slowly fill with water, forming lakes. The water quality of pit lakes may present a variety of pressing environmental problems.

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