

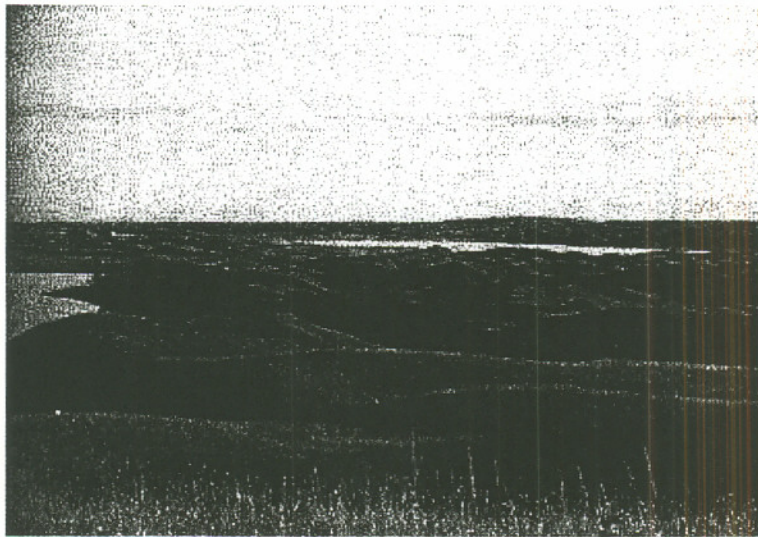


U.S. FISH & WILDLIFE SERVICE
REGION 6
CONTAMINANTS PROGRAM



**TRACE ELEMENT CONCENTRATIONS
IN PLANTS AND SEDIMENTS AT
CRESCENT LAKE NATIONAL WILDLIFE REFUGE**

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INTRODUCTION

Crescent Lake National Wildlife Refuge (NWR) is located in the Sandhills of western Nebraska, an area which contains the largest remaining tract of mid-grass and tall-grass prairie in North America. Land use surrounding the refuge is dominated by rangeland and land use practices likely result in minimal inputs of point or non-point source pollution. Industrial development in the area is minimal and the refuge complex is believed to be in fairly pristine condition. The use of persistent organochlorine pesticides on the refuge has been limited. The objectives of this study were to determine background concentrations of metals in biotic and abiotic components of the refuge and document these results to utilize as baseline information for reference and future contaminants investigations.

STUDY AREA

Crescent Lake National Wildlife Refuge

Crescent Lake NWR is located in Garden County, Nebraska, approximately 25 miles north of Oshkosh, Nebraska (Fig. 1).

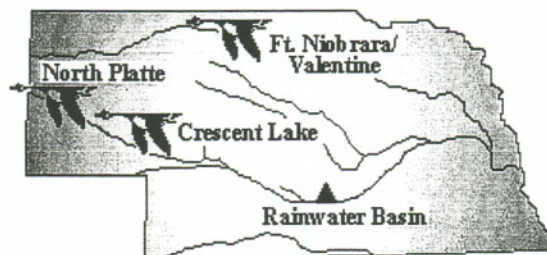


Figure 1. Location of Crescent Lake National Wildlife Refuge, Nebraska

The refuge was established in 1930 by the newly formed Migratory Bird Conservation Commission, which purchased 36,920 acres for use as a refuge and breeding area for waterfowl and other migratory birds. Crescent Lake NWR encompasses 45,996 acres which includes 43,372 acres of upland, 1,912 acres of open water and 712 acres of marsh. The remaining lands consist of meadows and grassy dunes.

The primary objective of the refuge is to provide nesting and habitat for migratory birds, especially ducks, geese, and shorebirds. Federally listed threatened species observed on the refuge include the threatened piping plover (*Charadrius melodus*) (seen only once or more since the refuge was established) and bald eagle (*Haliaeetus leucocephalus*). Endangered species occurring on the refuge include the least tern (*Sterna antillarum*) (seen only once or more since the refuge was established), and blowout penstemon (*Penstemon haydenii*). A total of 273 bird species have been observed on the refuge since 1936. Additional wildlife using the refuge include 16 species of amphibians, 16 species of reptiles, and 37 species of mammals.

Some pesticides have been used on the refuge. Toxaphene was applied to Crane, Island and Gimlet Lakes in the late 1950's and early 1960's to control common carp (*Cyprinus carpio*).

Roundup (glyphosate), Telar (chlorsulfuron), 2,4-D, and Banvel (dicamba) have been used by refuge personnel to control thistle.

Sandhills lakes generally have no surface water inlet or outlet (McCarragher 1977), are generally wind mixed, and do not thermally stratify (La Baugh 1986). Wind mixing of these shallow sandhills lakes results in re-suspension of the lake sediment causing substantial turbidity in lake water (La Baugh 1986). Alkalinity of lakes in Garden County range from 115 to 90,000 mg/L and range in pH from 8 to 10 (McCarragher 1977). Average depth of sandhills lakes is less than a meter, and the deepest lake, Blue Lake has a maximum depth of 4.2 m (McCarragher 1977).

Agricultural pesticides and fertilizers are not known to be used in the area, and land use adjacent to the refuge is primarily rangeland used for cattle grazing. Contamination from adjacent lands, therefore, is unlikely. Further, the watershed of most of the lakes are small because of the topography of the region so surface water inputs from adjacent lands is minimal.

MATERIALS AND METHODS

Samples collected for this study were collected from six different lakes on the refuge. Five lakes were sampled in 1993 (Blue, Crane, Gimlet, Island and Smith Lakes) and five lakes were sampled in 1994 (Crane, Gimlet, Hackberry, Island, and Smith Lakes). Sediment and aquatic plants were collected from each aquatic system in July of 1993, and August of 1994 (sampling locations are denoted in Fig 2). Sediment samples were obtained using a stainless steel spoon and placed in chemically cleaned glass jars. Plants were collected as close as possible to the location of sediment samples. Plants were obtained by hand and the entire plant was collected and placed in a plastic bag. Table 1 identifies the genus and locations of the collections.

Samples were sent to Hazleton Environmental Services Inc. and analyzed for trace metal content in sediment and aquatic plants. Inorganics were determined by inductively coupled plasma atomic emission spectrophotometer (ICP) scans. Arsenic and selenium were determined by atomic absorption spectrophotometry (AAS), and mercury levels were determined by cold vapor atomic absorption.

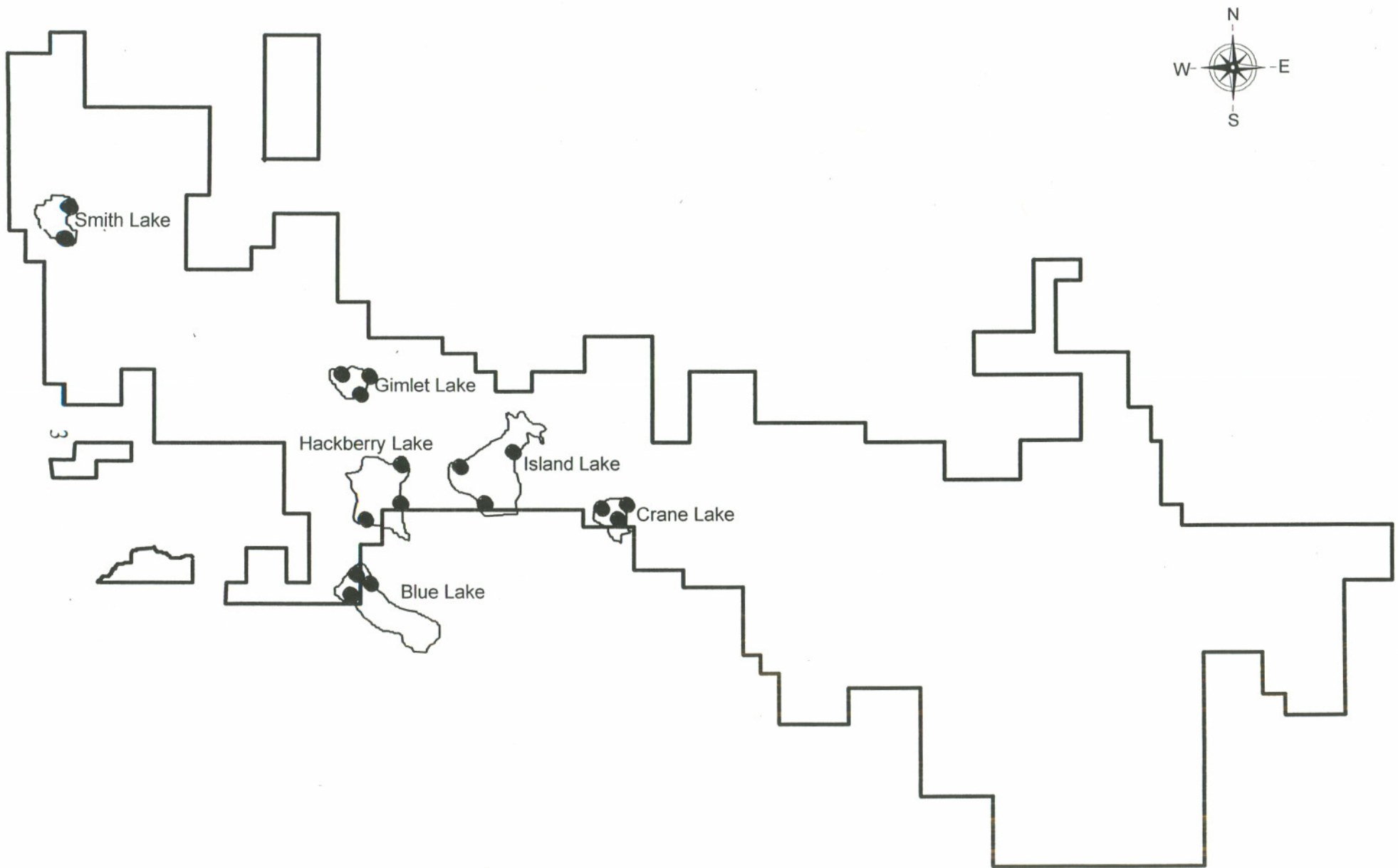
Concentrations are reported in $\mu\text{g/g}$ or parts per million (ppm) dry weight, unless noted otherwise (or if it refers to a concentration in water).

RESULTS AND DISCUSSION

Aluminum

Aluminum is abundant in the earth's crust and production and consumption of this metal is high. The largest contributor of anthropogenic aluminum to surface waters is the discharge of alum sludge from municipal water treatment plants. However, the resulting increase of aluminum in water from these additions appears to be minimal (Moore 1991). Toxicity of aluminum is dependent on pH and the high pH found in most sandhills lakes reduces the likelihood aluminum toxicity. The pH dependent toxicity of aluminum in plants, although well documented, remains unclear. One possible theory is differential speciation of aluminum at higher pHs (Parker et al. 1989). Another theory is that higher pH may reduce the plant's sensitivity to aluminum (Parker et al. 1989). Speciation of aluminum in aquatic environments is

Figure 2. Crescent Lake National Wildlife Refuge sampling locations.



further dictated by total organic carbon and other limnological parameters. Total aluminum concentrations detected should be used with caution.

Aluminum is the third most common metal in the earth's crust averaging 72,000 ppm in the conterminous U.S. (Shacklette and Boerngen 1984), and sediment concentrations from the refuge lakes were much lower ranging from 596.08 ppm to 5,869.02 ppm.

Aluminum detected in aquatic plants varied greatly, and concentrations detected in from sampled macrophytes ranged from 50.06 to 3,113.64 ppm. Comparatively, concentrations of aluminum in aquatic mosses collected in a Welsh metal mine drainage contained 54,000 ppm aluminum, and less contaminated reaches concentrations dropped to 2,000 - 7,000 ppm (Moore 1991). Levels detected in refuge lakes are similar to and lower than the less contaminated reaches of the river. High concentrations of aluminum in plants can result in decreased root growth, increased mucilage production (Crowder 1991), as well as reduction in plant biomass (Parker et al. 1989). Aluminum concentrations detected in refuge components sampled appear to be below levels of concern.

Arsenic

The major anthropogenic sources of arsenic include industrial smelters, coal-fired power plants, and production and use of arsenical pesticides. These anthropogenic inputs are significant and exceed the natural additions (i.e., rock weathering) of arsenic in the environment by a factor of three (Eisler 1994). Arsenic can cause bronchitis, pneumonia, and gangrene. It is also a mutagen, teratogen, and carcinogen (Eisler 1988).

Sediment arsenic concentrations detected from refuge lakes were mostly lower than the average for U.S. soils of 5.2 ppm (Shacklette and Boerngen 1984), and ranged from 0.89 ppm to 9.36 ppm. Comparatively, sediments collected from a lake contaminated by sodium arsenite herbicide 24 years earlier still contained from 540-740 ppm arsenic (Tanner and Clayton 1990).

Background levels of arsenic in terrestrial and freshwater flora and fauna are usually less than 1 ppm wet weight (w.w.) (Eisler 1994). Of the aquatic macrophytes sampled, only arrowhead (*Sagittaria* sp.) (from Hackberry Lake) and pondweed (*Potamogeton* sp.) (from Island Lake) exceeded this with concentrations of 1.17 ppm w.w., and 1.13 ppm w.w., respectively. However, the National Academy of Sciences (1977) reported levels in freshwater aquatic plants from areas free of arsenic treatments that ranged between 1.4 to 13.0 ppm, in comparison to 0.89 to 9.36 ppm detected from refuge plants. Further, these concentrations have not been documented to cause detrimental effects when ingested by birds (Eisler 1994). Arsenic concentrations detected on the refuge are below levels of concern.

Barium

Barium is a relatively abundant element found most frequently in the environment in the form of barite (barium sulfate) or witherite (barium carbonate) (Moore 1991). Barium is used as a drilling fluid in oil and gas wells which accounts for 90% of barium usage (Moore 1991).

Barium is also used in the production of various barium chemicals (Moore 1991).

Anthropogenic inputs of barium result from mining, refining, and processing barium ore, as well as the burning of fossil fuels (International Programme on Chemical Safety (IPCS) 1990), and drilling fluid spills (Moore 1991).

All sediment samples collected from refuge lakes were lower than the mean barium concentration of 440 ppm in soils of the conterminous U.S. (Shacklette and Boerngen 1984), with concentrations ranging from 6.3 to 242.22 ppm. Barium concentrations in sediment are usually below 100 ppm, and higher levels are usually associated with geologic deposits (IPCS 1990).

Although barium concentrations in macrophytes collected from refuge lakes ranged from 49.84 to 460.54 ppm, it is not known to accumulate in plants in sufficient quantities to cause toxicity to wildlife (IPCS 1990). Barium concentrations detected in refuge components sampled appear to be below levels of concern.

Beryllium

The major anthropogenic source of beryllium in the environment is the combustion of fossil fuels, and entry into aquatic environments occurs via atmospheric deposition, weathering of rocks and soils, as well as municipal/industrial point source inputs (USEPA 1980). The toxicity of beryllium increases in soft waters and the solubility of beryllium salts changes dramatically with changes in pH (Wilber 1980).

The mean beryllium concentration in soils of the conterminous U.S. is 0.63 ppm (Shacklette and Boerngen 1984), and sediments in Illinois range from 1.4 - 7.4 ppm (IPCS 1990). Sediment samples collected from the six refuge lakes ranged in concentrations from below the limit of detection (0.06 ppm) to 0.24 ppm. Beryllium concentrations were below the limit of detection in all aquatic plants collected from refuge lakes.

Boron

Boron is an essential trace element for higher plants but is not required by fungi or animals (Eisler 1990). Sources of boron additions to the environment include laundry products, agricultural chemicals and fertilizers, coal combustion, and mining and processing (Eisler 1990).

The average boron concentration in soils in the conterminous U.S. is 26 ppm (Shacklette and Boerngen 1984) and all sediment samples collected from lakes on the refuge were lower than this average, ranging from below the limit of detection (1 ppm) to 21.12 ppm

Aquatic macrophytes typically contain less than 20 ppm boron and range from 1.2 to 100 ppm (Eisler 1990). Aquatic plants collected on the refuge ranged from 7.21 to 42.13 ppm (1.37 to 5.86 ppm w.w.). Comparatively, the highest boron concentration in filamentous algae collected for an irrigation drainwater study in California was 280 ppm (Saiki et al. 1993), and aquatic macrophytes sampled from a boron contaminated wetland contained up to 142 ppm (Powell et al. 1997). Diets containing 30 ppm w.w. and greater of boron fed to adult mallards (*Anas platyrhynchos*) caused elevated embryo mortality (Smith and Anders 1989). Boron is believed to complex with riboflavin and other coenzymes critical for normal cellular respiration resulting in embryo mortality without causing deformities (Smith and Anders 1989). It appears that boron levels detected on the refuge are below levels of concern.

Cadmium

Cadmium is used in electroplating or in alloys as protection against corrosion, and is also used in batteries, ceramics, some biocides (Moore 1991), and the manufacturing of plastic

stabilizers (Eisler 1985a). Anthropogenic sources of cadmium include refining and smelting, manufacturing processes, and domestic wastewater (Moore 1991). Cadmium is not a biologically essential element and is toxic to all forms of life; it is a known teratogen, carcinogen, and possible mutagen (Eisler 1985a). Cadmium was below the limit of detection in sediment and aquatic vegetation collected from the refuge.

Chromium

Chromium is an essential trace element (Moore 1991) but elevated quantities can be mutagenic, teratogenic, and carcinogenic (Eisler 1986). Elevated levels of chromium are often found in surface waters near electroplating and metal finishing industries (Eisler 1986). Other sources of chromium include chromium alloy and metal producing industries, coal combustion, municipal incinerators, cement production, and cooling towers (Eisler 1986). The toxicity of chromium in aquatic environments is dependent on hardness, pH, and temperature (Eisler 1986). Further, only two oxidation states of chromium are known to be toxic (trivalent and hexavalent oxidation) (Eisler 1986), and hexavalent chromium is the more toxic form (USEPA 1985a). Concentrations of chromium from this investigation were provided in total chromium only.

Sediment samples contained chromium concentrations that ranged from below detection (1 ppm) to 7.11 ppm. The average concentration of chromium in soils of the conterminous U.S. is 37 ppm (Shacklette and Boerngen 1984). Background concentrations of chromium in sediment from the Great Lakes ranged from 9 - 86 ppm (Moore 1991). From this study, all levels detected in sediment were well below these background concentrations.

Concentrations of chromium in aquatic plants ranged from 0.5 to 3.34 ppm, much lower than concentrations causing a reduction in plant biomass. Chromium caused reduction in biomass in Ceratophyllum demersum which accumulated 876 ppm when grown in water containing 2 mg/L of chromium (Garg and Chandra 1990). Further, the highest concentration detected from aquatic plants collected from refuge lakes was still below the concentrations causing detrimental effects when ingested by black ducks (Anas rubripes) (Eisler 1986). Concentrations of chromium in samples collected from the refuge are below levels of concern.

Copper

Copper is a required nutrient for plants and animals but is toxic at levels only slightly higher than those required nutritionally (USEPA 1985b). Anthropogenic inputs of copper include industrial effluents (i.e., smelting, refining, and metal plating industries) (USEPA 1985b), copper containing algicides, paints, wood preservatives, and metal corrosion (Novotny and Olem 1994). Annual global copper releases approach 1.8 million metric tons primarily due to anthropogenic releases (Eisler 1997).

Sediment concentrations from sampled lakes were below the mean copper concentrations in the soils of the conterminous U.S. (37 ppm) (Shacklette and Boerngen 1984), and ranged from 0.47 to 5.11 ppm. In the United Kingdom, sediments from uncontaminated estuaries contained 10 ppm copper in sediments, while sediments from contaminated estuaries contained over 2000 ppm (Eisler 1997).

In this study, concentrations of copper in aquatic plants ranged from 1.87 to 94.14 ppm, and most concentrations were below 40 ppm. Concentrations of copper in aquatic macrophytes

inhabiting copper contaminated streams are much higher than those detected in refuge plants. Plants growing in contaminated reaches contained levels exceeding 600 ppm (Stokes 1979). Copper concentrations in refuge sediments and biota sampled appear to be below levels of concern.

Iron

Iron is a required nutrient for nearly all organisms (National Research Council 1979). It is used for the production of steel and is the fourth most abundant element in the earth's crust (Moore 1991). Natural erosion is responsible for the majority of iron delivered to the aquatic environment, although anthropogenic activities such as mining and municipal effluents also deliver iron to aquatic systems (Moore 1991).

All sediment concentrations were below the average of 26,000 ppm for the conterminous U.S. (Shacklette and Boerngen 1984), and ranged from 596 to 5,869 ppm. Concentrations in aquatic macrophytes collected from refuge lakes ranged from 173 to 6,873 ppm (21.2 to 708 ppm w.w.) and iron concentrations are usually high in aquatic plants (Moore 1991). Documentation of the toxic effects of iron in plants are lacking (Moore 1991). Iron concentrations detected in refuge sediments and biota sampled appear to be below levels of concern.

Lead

Lead is the fifth most utilized metal and is found in batteries, solder, and ammunition, and previously in gasoline, paint, and pesticides (USEPA 1992). Lead detection in the environment is becoming ubiquitous after centuries of anthropogenic inputs resulting from mining and smelting (Pain 1996). Lead is not an essential nutrient and is highly toxic (Pain 1995).

Lead concentrations in sediments collected from lakes on the refuge ranged from below the limit of detection (1.5 ppb) to 8.94 ppm. These concentrations are lower than the average for the conterminous U.S. of 16 ppm (Shacklette and Boerngen 1984).

Lead accumulation in aquatic plants occurs via uptake from contaminated sediment, and to a lesser degree from the water column (Demayo et al. 1982). Concentrations of lead in aquatic macrophytes ranged from below the limit of detection to 7.62 ppm. Plants inhabiting uncontaminated environments generally contain less than 1 ppm w.w. (Pain 1995). All w.w. samples of vegetation fell below this level except for one Polygonum sample from Island Lake containing 2.21 ppm. Dietary lead concentrations causing a significant reduction in blood enzyme delta-aminolevulinic acid dehydratase (ALAD) occurred in mallard drakes when lead concentrations reached 25 ppm w.w. (Finley et al. 1976). This is much higher than concentrations detected in aquatic macrophytes collected from refuge lakes. Lead concentrations detected on the refuge appear to be below levels of concern.

Magnesium

All chlorophyllous plants require magnesium and it is generally not a limiting factor in most aquatic systems (Wetzel 1983). Magnesium concentrations in sediment samples were well below average concentrations for soils in the conterminous U.S. of 9,000 ppm (Shacklette and Boerngen 1984), and ranged from 169 to 2,797 ppm. Magnesium concentrations in aquatic

macrophytes ranged from 763 to 7,352 ppm. Studies on magnesium are lacking, and at this time, concentrations detected during this study do not appear to be a concern.

Manganese

Manganese is detected widely in surface water and sediment, and concentrations can vary widely (Stubblefield et al. 1997). Manganese is used mainly in metal alloys (Moore 1991), and surface water concentrations adjacent to mining and smelting operations are often elevated as a result of point and nonpoint discharges (Stubblefield et al. 1997).

Sediment manganese concentrations sampled from the lakes were below the 330 ppm average for soils in the conterminous U.S. (Shacklette and Boerngen 1984), and ranged from 7.11 to 135.56 ppm of manganese. No biological effects guidelines have been created for manganese in sediment. Concentrations of manganese in aquatic vegetation ranged from 85 to 502 ppm. As an essential micronutrient in plants, manganese assists in nitrate assimilation in photosynthesis (Wetzel 1983). Manganese tends to saturate metal binding sites in aquatic plants protecting the plants against the effects of more toxic heavy metals (Moore 1991). Manganese is normally considered the least toxic of the trace elements for poultry and mammals (Pais and Jones 1997) and therefore, it is likely not a concern to refuge fish and wildlife.

Mercury

Mercury has no known biological function and has the potential to bioconcentrate and biomagnify (Eisler 1987). Although natural sources (e.g., volcanic activity) emit mercury to the environment (Thompson 1996), anthropogenic sources deliver nearly 9,000 metric tons of mercury per year to freshwater systems (Moore 1991). The major anthropogenic sources of mercury include coal burning power plants and the manufacturing of chemicals and metals (Moore 1991). Other anthropogenic activities resulting in mercury additions to the environment include battery and florescent light disposal, as well as the mining and processing of gold, lead, and copper (Eisler 1987).

Mercury poisoning most often results from methylmercury (the most stable form and most toxic to wildlife) (Thompson 1996). Symptoms of mercury poisoning include loss of coordination, numbness in the extremities, as well as hampered awareness and reduced mental activity (Thompson 1996). Methylmercury is formed through methylation of inorganic mercury primarily via microbial processes (Wiener and Spry 1996). Accumulation of mercury in aquatic organisms occurs via uptake from sediments, water, and food (Wren et al. 1995).

Sediment samples collected from refuge lakes contained mercury levels ranging from below the limit of detection to 0.137 ppm and were mostly lower in comparison to the average concentration of mercury in soils (0.058 ppm) of the conterminous U.S. (Shacklette and Boerngen 1984). True background concentrations of mercury in sediment likely no longer exist unless deep undisturbed sediment is sampled. Long range atmospheric transport of mercury has likely affected soils and sediments world wide (Wren et al. 1995). The two samples containing higher than average concentrations of mercury were collected from Gimlet Lake and contained concentrations of 0.137 and 0.076 ppm. The highest concentration is slightly below the biological effects range-low (ER-L) of 0.15 ppm described by Long and Morgan (1990). The ER-L was the concentration where effects may begin or be predicted among sensitive species

(Long and Morgan 1990).

Concentrations of mercury in aquatic vegetation collected from the refuge ranged from below the limit of detection to 0.128 ppm. Concentrations of mercury in macrophytes are typically not detected at levels above 0.01 ppm (Crowder 1991). Both Blue Lake and Crane Lake contained concentrations of mercury that exceeded levels described by Crowder (1991). Comparatively, concentrations of mercury in macrophytes growing near an industrial contaminants source contained up to 1.6 ppm (Crowder 1991). Toxic effects of mercury on macrophytes includes root death and discoloration and necrosis of floating leaves (Crowder 1991). The uptake of mercury by aquatic plants is enhanced under alkaline and oxidizing conditions and reduced as salinity increased (Crowder 1991). Although concentrations in aquatic plants collected from the refuge appear slightly elevated, they are much lower than levels resulting in egg laying reductions in diets of loons containing 0.3 to 0.4 ppm w.w. mercury (Barr 1986). Specifically, concentrations of mercury detected in refuge plants did not exceed 0.019 ppm w.w..

Molybdenum

Molybdenum is mainly used in the production of steel alloys as well as in the production of pigments, spark plugs, x-ray tubes, and catalysts (Eisler 1989). Anthropogenic sources of molybdenum include coal combustion, molybdenum mining and milling, and oil refining (Eisler 1989). Molybdenum is an essential nutrient for most life forms although it can be toxic. Toxicity of molybdenum is dependent on copper and inorganic sulfate intake (Eisler 1989).

Concentrations of molybdenum in sediment samples were below the limit of detection. Molybdenum in vegetation samples from the lakes ranged from below the limit of detection to 13.55 ppm. Molybdenum is biologically more available to plants in alkaline soils, and the high alkalinity of the refuge lakes may result in increased molybdenum uptake (Eisler 1989). It is unlikely that the concentrations detected in aquatic plants are detrimental based on data collected on freshwater algae containing 20,000 ppm of molybdenum without apparent detrimental effects (Eisler 1989). Further, concentrations detected in plants collected from refuge lakes were also below dietary levels causing detrimental effects in birds (Eisler 1989).

Nickel

The major anthropogenic sources of nickel in the environment are generated from the combustion of fossil fuels, electroplating, and smelting industries (USEPA 1986), as well as production of alloys, batteries, and electronics (Birge and Black 1980). The toxicity of nickel in aquatic systems is dependent on alkalinity, hardness, salinity, pH, and temperature (USEPA 1986).

Nickel concentrations in sediment samples ranged from 0.45 to 4.53 ppm, lower than the 13 ppm average for soils of the conterminous U.S. (Shacklette and Boerngen 1984). Comparatively, nickel concentrations detected in sediment from a confined disposal facility for dredged material contained levels ranging from 12 to 150 ppm (Beyer et al. 1990). Based on the previous comparisons, it appears nickel concentrations contained in sediments collected from refuge lakes are below levels of concern.

Nickel concentrations in aquatic macrophytes ranged from below the limit of detection to

5.33 ppm (below the limit of detection to 0.07 ppm w.w.). Aquatic macrophytes inhabiting uncontaminated areas typically contain nickel concentrations less than 6 ppm w.w. (Jenkins 1980). Concentrations of nickel in macrophytes from Crescent Lake NWR were lower than this benchmark. Therefore, nickel levels detected do not appear problematic for aquatic plants from the lakes sampled.

Selenium

While selenium is an essential micronutrient, levels exceeding the capacity of metabolic regulation leads to toxicity. Two anthropogenic activities which are major contributors to elevated levels of selenium include the production and use of fossil fuels and irrigation of seleniferous soils in semiarid regions of the country (Lemly 1996). High selenium concentrations (exceeding 100 ppm) in the diets of adult mallards resulted in fatality, while lower levels resulted in reduced hatching success (Eisler 1985b). In fish, toxic levels of selenium result in loss of equilibrium, loss of coordination, liver degeneration, and an increase in white blood cell count (Eisler 1985b).

Selenium concentrations in sediment collected from the refuge ranged from below the limit of detection to 0.65 ppm. Only one sediment sample collected from Crane Lake contained selenium above the limit of detection which was higher than the average concentration in soils (0.26 ppm) for the conterminous U.S. (Shacklette and Boerngen 1984). Selenium concentrations in sediment exceeding 4 ppm could be a potential concern for fish and waterfowl (Lemly and Smith 1987). However, levels of selenium in sediment collected from Crane Lake were much lower than this threshold. Trophic transfer of selenium through benthic organisms exposed to selenium in sediment is believed to be the major route of selenium accumulation in fish. However, selenium was below the limit of detection in most of the sediment samples collected from refuge lakes. Van Derveer and Canton (1997) proposed that the level of selenium in sediment is related to total organic carbon and dissolved selenium. Consequently, the sand substrate found in most sandhills lakes may not provide binding sites for selenium than a more organic-based substrate would, yielding selenium levels below the level of detection.

Aquatic vegetation contained selenium concentrations ranging from below the limit of detection (0.5 ppm) to 4.35 ppm, revealing higher than background concentrations. Selenium concentrations reported for aquatic macrophytes collected from control areas contained of 0.1 ppm to 0.4 ppm (Eisler 1985b). Only Crane Lake contained macrophytes with detectable levels of selenium, and plants were collected from this lake in both 1993 and 1994. The highest concentration detected was 4.35 ppm in a pondweed sample, and the other elevated concentrations were detected in arrowhead plants containing 3.19 and 3.24 ppm selenium. Heinz et al. (1989) found dietary concentrations of selenium causing reproductive impairment in mallards were between 4 and 8 ppm. Further, Lemly (1993) recommended considering food chain items containing 3 ppm selenium or more potentially lethal to fish and aquatic birds when used as a food source. Aquatic vegetation from Crane Lake may be a potential contaminant source to waterfowl using the macrophytes as a food source.

Strontium

Strontium is an alkaline earth element and like calcium, accumulates in bone tissue (Pais

and Jones 1997). Concentrations of strontium in sediment ranged from 2.21 to 72.35 ppm in this study. Average concentration of strontium in soils for the conterminous U.S. is 120 ppm (Shacklette and Boerngen 1984). Aquatic macrophytes collected from refuge lakes contained strontium ranging from 16.11 to 426.53 ppm. Much of the research available on strontium focuses on the radioactive form (^{90}Sr). Non-radioactive strontium is not known to be toxic (Pais and Jones 1997).

Vanadium

Vanadium is used in metallurgy, dyes, inks, and paints as well as being used as a catalyst in the production of polymeric plastics (Moore 1991). The major anthropogenic source of vanadium in the environment results from the combustion of oil and coal (Moore 1991).

Sediment concentrations ranged from 1.53 to 28.59 ppm, lower than the average concentration for soils (58 ppm) of the conterminous U.S. (Shacklette and Boerngen 1984). Further, concentrations found in this study fall within the range of normal sediment vanadium levels of 20 to 150 ppm (Moore 1991).

Concentrations of vanadium detected in aquatic macrophytes collected from refuge lakes ranged from below the limit of detection to 16.16 ppm. Vanadium appears to be relatively nontoxic to plants and concentrations in freshwater plants normally range from 0.10 to 5.7 ppm (Moore 1991). Concentrations detected in refuge sediments and biota sampled appear to be below levels of concern.

Zinc

Zinc is one of the most widely used metals worldwide and its principal uses include galvanizing steel, additive for paint, and an ingredient in rubber (USEPA 1987). The major anthropogenic sources occur from smelting operations, combustion of fossil fuels, as well as from corroded galvanized electrical transmission towers (Eisler 1993). Once zinc enters the aquatic environment it is usually partitioned into the sediments where release is dependent on high dissolved oxygen and low salinity and pH (Eisler 1993). In water, speciation is dependent on oxygen levels, pH; and salinity. However, fish exposure to toxic zinc concentrations can increase in alkaline waters because the change in pH around the gill due to the release of CO_2 may cause the release of toxic soluble zinc from the zinc precipitates present in some alkaline environments (Sorenson 1991). Zinc is a required nutrient and is essential for normal growth, reproduction, and the ability to heal. However, zinc is a priority pollutant that can be teratogenic, mutagenic, and carcinogenic (Eisler 1993).

Zinc concentrations in sediment samples ranged from 1.52 to 18.58 ppm, well below the average concentration for soils (180 ppm) in the conterminous U.S. (Shacklette and Boerngen 1984). Further, the water chemistry of the lakes in the sandhills likely reduces the availability of zinc in the sediment (i.e., high pH, low dissolved oxygen, and high salinity) (McCarragher 1977).

Levels of zinc detected in aquatic macrophytes ranged from 5.79 to 55.63 ppm. Marginal sublethal effects of dietary zinc concentrations for birds occur starting at 178 ppm (Eisler 1993). This is much higher than concentrations detected in aquatic macrophytes in this study. Zinc concentrations detected in refuge sediments and biota sampled appear to be below levels of concern.

SUMMARY

Elevated inorganic contaminant concentrations detected in a limited number of sediment and macrophyte samples collected from the water bodies of Crescent Lake NWR. In sediment, only mercury was detected at concentrations slightly above background levels. The toxicity mercury in sediment to biota however could be questionable, since pH, water hardness, temperature, and oxygen concentration as well as the form (i.e., inorganic mercury vs. methyl mercury) can influence bioavailability. Measurements of water hardness, pH, Eh, and percent organic carbon in the sediment would provide useful information on the bioavailability of sediment associated metals (McIntosh 1991).

Aquatic plants collected as part of this study did show elevated concentrations for some trace elements. Island Lake contained vegetation with slightly elevated levels of lead. Crane Lake and Blue Lake contained slightly elevated levels of mercury. Potentially toxic concentrations of selenium were detected in three of the seven plant samples collected from Crane Lake. Concentrations of some trace elements detected in this study were not easy to interpret because of the paucity of data on concentrations causing detrimental effects to the plants or their consumers. Aquatic macrophyte uptake of metals, which in many cases are required nutrients or micronutrients, depend on several environmental factors such as pH and Eh. Further, submerged aquatic vegetation accumulate metals at higher rates than emergent vegetation. This can become a significant difference as concentrations of trace metals can be 100,000 times greater in aquatic vegetation than in the water (Albers and Camardese 1993).

Although elevated levels of trace elements were detected in this study, it does not appear that the levels of inorganics detected are a concern in this refuge except for the selenium levels detected in macrophytes collected from Crane Lake. Additional sampling of several trophic levels is recommended to determine if these elevated levels of selenium are impacting other biota using Crane Lake. Further, sampling the lakes treated with toxaphene is also recommended to determine if its persistence and potential to bioaccumulate (Barron 1995) are causing any detrimental effects to biota.

Table 1. Crescent Lake NWR sampling locations, dates and species collected.

Sample	Location	Common Name	Genus	Latitude	Longitude	Date
CLBP1	Blue Lake	hardstem bulrush	Scirpus	41N39	102W30	07/20/1993
CLBP2	Blue Lake	hardstem bulrush	Scirpus	41N40	102W31	07/20/1993
CLBS1	Blue Lake	sediment		41N39	102W30	07/20/1993
CLBS2	Blue Lake	sediment		41N40	102W31	07/20/1993
CLBS3	Blue Lake	sediment		41N41	102W32	07/20/1993
CLCP1	Crane Lake	arrowhead	Sagittaria	41N41	102W21	07/20/1993
CLCP2	Crane Lake	arrowhead	Sagittaria	41N41	102W22	07/20/1993
CLCP294	Crane Lake	smartweed	Polygonum	41N41	102W22	08/17/1994
CLCP3	Crane Lake	pondweed	Potamogeton	41N41	102W22	07/20/1993
CLCP394	Crane Lake	arrowhead	Sagittaria	41N41	102W23	08/17/1994
CLCP4	Crane Lake	smartweed	Polygonum	41N41	102W23	07/20/1993
CLCP494	Crane Lake	smartweed	Polygonum	41N41	102W23	08/17/1994
CLCP94	Crane Lake	arrowhead	Sagittaria	41N41	102W21	08/17/1994
CLCS1	Crane Lake	sediment		41N41	102W21	07/20/1993
CLCS2	Crane Lake	sediment		41N41	102W22	07/20/1993
CLCS294	Crane Lake	sediment		41N41	102W21	08/17/1994
CLCS3	Crane Lake	sediment		41N41	102W23	07/20/1993
CLCS394	Crane Lake	sediment		41N41	102W23	08/17/1994
CLCS494	Crane Lake	sediment		41N41	102W23	08/17/1994
CLCS94	Crane Lake	sediment		41N41	102W21	08/17/1994
CLGLP1	Gimlet Lake	arrowhead	Sagittaria	41N47	102W30	07/20/1993
CLGLP2	Gimlet Lake	burreed	Sparganium	41N47	102W30	07/20/1993
CLGLP3	Gimlet Lake	arrowhead	Sagittaria	41N46	102W29	07/20/1993
CLGLP4	Gimlet Lake	arrowhead	Sagittaria	41N46	102W29	07/20/1993
CLGLS	Gimlet Lake	sediment		41N47	102W30	07/20/1993
CLGLS2	Gimlet Lake	sediment		41N46	102W29	07/20/1993
CLGP294	Gimlet Lake	arrowhead	Sagittaria	41N47	102W30	08/17/1994
CLGP94	Gimlet Lake	smartweed	Polygonum	41N47	102W30	08/17/1994
CLGS294	Gimlet Lake	sediment		41N46	102W29	08/17/1994
CLGS94	Gimlet Lake	sediment		41N46	102W29	08/17/1994
CLHP294	Hackberry Lake	arrowhead	Sagittaria	41N45	102W30	08/17/1994
CLHP394	Hackberry Lake	arrowhead	Sagittaria	41N45	102W30	08/17/1994
CLHP94	Hackberry Lake	smartweed	Polygonum	41N45	102W30	08/17/1994
CLHS294	Hackberry Lake	sediment		41N45	102W30	08/17/1994
CLHS394	Hackberry Lake	sediment		41N44	102W30	08/17/1994
CLHS494	Hackberry Lake	sediment		41N44	102W30	08/17/1994
CLHS94	Hackberry Lake	sediment		41N45	102W30	08/17/1994
CLIP1	Island Lake	hardstem bulrush	Scirpus	41N43	102W27	07/20/1993
CLIP2	Island Lake	hardstem bulrush	Scirpus	41N42	102W26	07/20/1993
CLIP294	Island Lake	smartweed	Polygonum	41N41	102W25	08/17/1994
CLIP3	Island Lake	smartweed	Polygonum	41N41	102W25	07/20/1993
CLIP394	Island Lake	smartweed	Polygonum	41N41	102W25	08/17/1994
CLIP94	Island Lake	pondweed	Potamogeton	41N43	102W27	08/17/1994
CLIS1	Island Lake	sediment		41N43	102W27	07/20/1993
CLIS2	Island Lake	sediment		41N42	102W26	07/20/1993
CLIS294	Island Lake	sediment		41N42	102W26	08/17/1994
CLIS3	Island Lake	sediment		41N41	102W25	07/20/1993
CLIS394	Island Lake	sediment		41N41	102W25	08/17/1994
CLIS94	Island Lake	sediment		41N43	102W27	08/17/1994
CLSI94	Smith Lake	sediment		41N55	102W48	08/17/1994
CLSI94	Smith Lake	sediment		41N55	102W48	07/20/1993
CLSL94	Smith Lake	sediment		41N54	102W47	08/17/1994
CLS094	Smith Lake	sediment		41N54	102W47	08/17/1994
CLSOS	Smith Lake	sediment		41N54	102W47	07/20/1993
CLSS	Smith Lake	sediment		41N54	102W47	07/20/1993

Table 2. Concentrations of inorganics detected in sediment collected from Crescent Lake NWR, in ppm (µg/g) dry weight.

Location	Sample	Al	As	B	Ba	Be	Cd	Cr	Cu	Fe	Hg	Mg
Blue Lake	CLBS1	831.54	0.26	2.13	10.00	< .06	< .19	1.30	0.61	966.31	< .05	231.81
Blue Lake	CLBS2	914.40	< .13	2.28	6.30	< .06	< .19	1.27	0.63	852.14	< .05	207.52
Blue Lake	CLBS3	596.08	< .13	< 1.31	4.93	< .07	< .20	1.00	0.47	660.13	< .05	169.93
Crane Lake	CLCS1	1857.52	0.92	5.12	82.59	< .13	< .38	2.10	1.39	1928.76	< .10	823.22
Crane Lake	CLCS2	2076.75	0.65	4.33	50.56	< .11	< .33	2.53	1.25	1997.74	< .09	918.74
Crane Lake	CLCS294	1608.52	0.59	3.16	33.47	< .10	< .30	1.91	1.00	1507.10	< .08	375.25
Crane Lake	CLCS3	1956.52	0.63	3.46	40.04	< .09	< .26	2.12	1.16	1902.17	< .07	512.68
Crane Lake	CLCS394	2247.23	1.25	6.35	52.77	< .18	< .54	2.94	1.65	2380.07	< .14	682.66
Crane Lake	CLCS494	1689.32	0.75	2.84	50.73	< .12	< .36	2.27	1.50	2131.07	< .09	495.15
Crane Lake	CLCS94	2892.31	1.29	5.51	83.69	< .15	< .44	3.20	2.09	2861.54	< .12	750.77
Gimlet Lake	CLGLS	7822.22	2.36	19.29	242.22	0.26	< .64	7.11	5.11	6888.89	< .17	1848.89
Gimlet Lake	CLGLS2	2537.46	1.76	11.27	105.21	< .16	< .47	2.83	2.50	2771.99	0.14	765.47
Gimlet Lake	CLGS294	5869.02	1.44	13.63	136.02	0.24	< .36	5.52	4.46	4911.84	< .10	1576.83
Gimlet Lake	CLGS94	2824.43	0.90	9.37	62.60	0.12	< .27	2.90	2.00	2557.25	0.08	772.90
Hackberry Lake	CLHS294	1287.32	0.79	2.83	23.80	< .07	< .20	1.48	0.72	1256.34	< .05	338.03
Hackberry Lake	CLHS394	2621.36	2.33	14.30	77.18	< .11	< .34	2.96	1.99	2548.54	< .09	1441.75
Hackberry Lake	CLHS494	1884.45	1.41	9.71	45.17	< .10	< .31	2.29	1.40	1842.44	< .08	913.87
Hackberry Lake	CLHS94	4047.06	3.59	21.12	171.18	< .29	< .88	5.01	4.69	3800.00	< .23	1947.06
Island Lake	CLIS1	990.74	0.53	1.31	11.69	< .06	< .19	1.23	0.48	1104.50	< .05	240.74
Island Lake	CLIS2	1330.07	0.34	3.13	14.83	< .07	< .20	1.69	0.66	1223.78	< .05	320.28
Island Lake	CLIS294	1697.05	0.45	3.32	18.79	< .07	< .20	1.94	0.74	1458.63	< .05	387.10
Island Lake	CLIS3	1607.81	0.63	4.91	21.56	< .09	< .26	2.01	1.03	1425.65	< .07	431.23
Island Lake	CLIS394	1090.01	0.60	1.45	15.75	< .07	< .21	1.38	0.77	1202.53	0.06	319.27
Island Lake	CLIS94	1042.74	0.46	1.91	14.39	< .07	< .21	1.35	0.64	928.77	< .05	269.23
Smith Lake	CLSI94	2022.16	0.48	6.77	47.37	< .07	< .21	2.22	1.19	1745.15	< .05	2105.26
Smith Lake	CLSIS	2369.15	0.47	3.40	60.47	0.08	< .2	2.48	0.98	1900.83	< .05	1180.44
Smith Lake	CLSL94	1167.32	0.17	1.85	12.62	< .06	< .19	1.69	0.69	1256.81	< .05	424.12
Smith Lake	CLSO94	3071.30	0.59	6.76	104.94	0.12	< .26	2.69	1.32	2595.98	< .07	2797.08
Smith Lake	CLSOS	3045.33	0.30	3.64	40.79	0.12	< .21	2.69	1.27	2422.10	< .05	1728.05
Smith Lake	CLSS	1974.96	0.49	5.20	49.10	0.07	< .20	2.09	0.76	1557.72	< .05	2586.93

Table 2. Continued

Location	Sample	Mn	Mo	Ni	Pb	Se	Sr	V	Zn
Blue Lake	CLBS1	11.85	< 1.30	0.67	< 1.62	< .27	3.46	2.29	2.41
Blue Lake	CLBS2	9.42	< 1.24	0.64	< 1.56	< .26	3.28	2.32	2.24
Blue Lake	CLBS3	7.11	< 1.31	0.45	< 1.63	< .26	2.21	1.53	1.52
Crane Lake	CLCS1	105.28	< 2.53	1.06	< 3.17	< .53	50.66	4.41	5.88
Crane Lake	CLCS2	67.04	< 2.20	1.43	< 2.75	< .45	29.57	4.88	5.15
Crane Lake	CLCS294	33.67	< 1.97	1.54	< 2.45	< .41	11.44	3.75	4.12
Crane Lake	CLCS3	38.59	< 1.74	1.04	< 2.17	< .36	14.64	4.64	4.57
Crane Lake	CLCS394	53.14	< 3.56	1.60	< 4.46	< .74	20.59	6.90	6.35
Crane Lake	CLCS494	71.12	< 2.38	1.21	< 2.99	< .49	17.79	4.85	5.24
Crane Lake	CLCS94	86.15	< 2.96	2.97	< 3.69	0.65	32.00	7.08	8.06
Gimlet Lake	CLGLS	135.56	< 4.23	4.53	8.18	< .89	85.33	20.13	18.58
Gimlet Lake	CLGLS2	35.83	< 3.10	3.12	< 3.88	< .65	33.88	12.35	11.89
Gimlet Lake	CLGS294	73.55	< 2.42	3.22	6.07	< .50	48.36	19.92	33.00
Gimlet Lake	CLGS94	38.17	< 1.82	1.56	2.40	< .38	23.28	10.69	11.58
Hackberry Lake	CLHS294	16.62	< 1.34	0.80	< 1.68	< .28	8.56	3.73	2.83
Hackberry Lake	CLHS394	47.09	< 2.30	1.47	4.51	< .49	58.74	8.03	8.40
Hackberry Lake	CLHS494	27.52	< 2.05	0.99	2.77	< .42	31.72	5.69	5.63
Hackberry Lake	CLHS94	95.88	< 5.82	3.66	8.94	< 1.18	72.35	28.59	15.59
Island Lake	CLIS1	7.82	< 1.28	0.79	< 1.60	< .26	4.11	2.13	2.05
Island Lake	CLIS2	14.13	< 1.33	0.78	< 1.66	< .28	7.43	3.12	2.60
Island Lake	CLIS294	14.03	< 1.32	1.33	< 1.64	< .28	7.18	3.63	3.20
Island Lake	CLIS3	21.93	< 1.75	0.92	< 2.19	< .37	10.76	4.70	3.94
Island Lake	CLIS394	13.97	< 1.41	0.71	1.87	< .28	5.72	2.21	3.32
Island Lake	CLIS94	21.51	< 1.38	0.57	< 1.72	< .28	6.32	2.46	2.62
Smith Lake	CLSI94	56.37	< 1.39	1.01	< 1.73	< .28	38.50	4.61	5.35
Smith Lake	CLSI94	32.64	< 1.34	1.14	< 1.67	< .28	31.82	4.67	4.31
Smith Lake	CLSL94	13.49	< 1.28	0.76	< 1.61	< .26	8.73	2.89	2.53
Smith Lake	CLSO94	91.22	< 1.74	2.01	2.63	< .37	58.14	5.37	14.77
Smith Lake	CLSOS	32.01	< 1.41	1.77	2.00	< .28	27.48	5.35	20.40
Smith Lake	CLSS	32.41	< 1.33	0.84	1.75	< .28	78.30	3.95	3.41

Table 3. Concentrations of inorganics detected in aquatic vegetation collected from Crescent Lake NWR in ppm (µg/g).

Sample Location	Sample Number	Al		As		B		Ba		Be	
		ppm D.W.	ppm W.W.	ppm D.W.	ppm W.W.	ppm D.W.	ppm W.W.	ppm D.W.	ppm W.W.	ppm D.W.	ppm W.W.
Blue Lake	CLBP1	63.04	9.33	1.08	0.16	9.86	1.46	70.27	10.40	< .14	< .02
Blue Lake	CLBP2	112.11	21.30	0.89	0.17	7.21	1.37	49.84	9.47	< .11	< .02
Crane Lake	CLCP1	136.94	9.86	4.44	0.32	19.44	1.40	123.06	8.86	< .28	< .02
Crane Lake	CLCP2	222.22	16.00	1.25	0.09	19.03	1.37	116.25	8.37	< .28	< .02
Crane Lake	CLCP294	363.16	34.50	2.42	0.23	18.63	1.77	104.53	9.93	< .21	< .02
Crane Lake	CLCP3	68.03	4.83	4.37	0.31	24.79	1.76	380.28	27.00	< .28	< .02
Crane Lake	CLCP394	908.79	82.70	8.13	0.74	26.70	2.43	127.47	11.60	< .22	< .02
Crane Lake	CLCP4	89.83	21.20	1.36	0.32	24.83	5.86	138.14	32.60	< .08	< .02
Crane Lake	CLCP494	135.62	29.70	2.15	0.47	25.34	5.55	118.72	26.00	< .09	< .02
Crane Lake	CLCP94	825.40	104.00	1.59	0.20	13.33	1.68	83.33	10.50	< .16	< .02
Gimlet Lake	CLGLP1	1160.65	70.80	2.79	0.17	42.13	2.57	175.41	10.70	< .33	< .02
Gimlet Lake	CLGLP2	3113.64	411.00	1.97	0.26	21.44	2.83	176.52	23.30	< .15	< .02
Gimlet Lake	CLGLP3	1815.53	187.00	5.73	0.59	41.55	4.28	283.50	29.20	< .19	< .02
Gimlet Lake	CLGLP4	935.64	94.50	3.66	0.37	16.34	1.65	135.64	13.70	< .20	< .02
Gimlet Lake	CLGP294	1526.79	171.00	7.23	0.81	24.29	2.72	117.86	13.20	< .18	< .02
Gimlet Lake	CLGP94	1201.03	233.00	1.80	0.35	11.44	2.22	103.09	20.00	< .10	< .02
Hackberry Lake	CLHP294	2504.00	313.00	9.36	1.17	19.68	2.46	146.40	18.30	< .16	< .02
Hackberry Lake	CLHP394	1865.08	235.00	6.03	0.76	26.83	3.38	125.40	15.80	< .16	< .02
Hackberry Lake	CLHP94	321.94	63.10	2.60	0.51	23.88	4.68	172.45	33.80	< .10	< .02
Island Lake	CLIP1	50.06	8.61	0.93	0.16	12.44	2.14	118.60	20.40	< .12	< .02
Island Lake	CLIP2	726.98	45.80	5.08	0.32	25.87	1.63	91.59	5.77	< .32	< .02
Island Lake	CLIP294	193.09	41.90	1.98	0.43	21.71	4.71	158.99	34.50	< .09	< .02
Island Lake	CLIP3	104.19	22.40	2.23	0.48	23.21	4.99	87.91	18.90	< .09	< .02
Island Lake	CLIP394	315.52	91.50	2.38	0.69	20.00	5.80	115.86	33.60	< .07	< .02
Island Lake	CLIP94	741.50	109.00	7.69	1.13	26.80	3.94	460.54	67.70	< .14	< .02

Table 3. Continued

Sample Location	Sample Number	Cd		Cr		Cu		Fe		Hg	
		ppm D.W.	ppm W.W.	ppm D.W.	ppm W.W.	ppm D.W.	ppm W.W.	ppm D.W.	ppm W.W.	ppm D.W.	ppm W.W.
Blue Lake	CLBP1	< .41	< .06	0.84	0.12	7.30	1.08	180.41	26.70	0.13	0.02
Blue Lake	CLBP2	< .31	< .06	0.69	0.13	8.79	1.67	222.63	42.30	< .05	< .01
Crane Lake	CLCP1	< .83	< .06	1.49	0.11	12.13	0.87	294.44	21.20	< .13	< .01
Crane Lake	CLCP2	< .82	< .06	1.69	0.12	12.08	0.87	850.00	61.20	< .13	< .01
Crane Lake	CLCP294	< .63	< .06	1.38	0.13	18.74	1.78	1027.37	97.60	< .10	< .01
Crane Lake	CLCP3	< .83	< .06	1.54	0.11	13.07	0.93	273.24	19.40	< .14	< .01
Crane Lake	CLCP394	< .66	< .06	2.02	0.18	20.44	1.86	4307.69	392.00	< .11	< .01
Crane Lake	CLCP4	< .25	< .06	0.50	0.12	4.19	0.99	618.64	146.00	0.05	0.01
Crane Lake	CLCP494	< .27	< .06	0.64	0.14	1.87	0.41	598.17	131.00	< .04	< .01
Crane Lake	CLCP94	< .48	< .06	1.59	0.20	20.32	2.56	2174.60	274.00	< .07	< .01
Gimlet Lake	CLGLP1	< .98	< .06	2.41	0.15	5.23	0.32	4967.21	303.00	< .16	< .01
Gimlet Lake	CLGLP2	< .45	< .06	3.34	0.44	35.53	4.69	4962.12	655.00	< .07	< .01
Gimlet Lake	CLGLP3	< .58	< .06	2.75	0.28	48.64	5.01	6873.78	708.00	< .09	< .01
Gimlet Lake	CLGLP4	< .58	< .06	1.78	0.18	47.43	4.79	3514.85	355.00	< .09	< .01
Gimlet Lake	CLGP294	< .54	< .06	2.19	0.25	33.75	3.78	5473.22	613.00	< .08	< .01
Gimlet Lake	CLGP94	< .31	< .06	1.91	0.37	7.01	1.36	2484.54	482.00	< .05	< .01
Hackberry Lake	CLHP294	< .48	< .06	3.21	0.40	16.00	2.00	4296.00	537.00	< .08	< .01
Hackberry Lake	CLHP394	< .47	< .06	2.64	0.33	14.76	1.86	3412.70	430.00	< .07	< .01
Hackberry Lake	CLHP94	< .31	< .06	0.78	0.15	10.41	2.04	714.29	140.00	< .05	< .01
Island Lake	CLIP1	< .35	< .06	0.72	0.12	13.49	2.32	173.26	29.80	< .05	< .01
Island Lake	CLIP2	< .95	< .06	2.27	0.14	30.48	1.92	1682.54	106.00	< .15	< .01
Island Lake	CLIP294	< .28	< .06	0.74	0.16	5.94	1.29	303.69	65.90	< .04	< .01
Island Lake	CLIP3	< .27	< .06	0.69	0.15	6.98	1.50	385.12	82.80	< .04	< .01
Island Lake	CLIP394	< .20	< .06	1.03	0.30	94.14	27.30	1668.97	484.00	< .03	< .01
Island Lake	CLIP94	< .41	< .06	2.36	0.35	8.23	1.21	1489.80	219.00	< .06	< .01

Table 3. Continued

Sample Location	Sample Number	Mg		Mn		Mo		Ni		Pb	
		ppm D.W.	ppm W.W.	ppm D.W.	ppm W.W.	ppm D.W.	ppm W.W.	ppm D.W.	ppm W.W.	ppm D.W.	ppm W.W.
Blue Lake	CLBP1	891.89	132.00	162.84	24.10	3.77	0.56	2.01	0.30	< 3.36	< .50
Blue Lake	CLBP2	763.16	145.00	127.37	24.20	< 2.08	< .40	1.13	0.21	< 2.61	< .50
Crane Lake	CLCP1	3083.33	222.00	134.31	9.67	< 5.54	< .40	1.86	0.13	< 6.93	< .50
Crane Lake	CLCP2	3055.56	220.00	188.89	13.60	< 5.5	< .40	1.72	0.12	< 6.88	< .50
Crane Lake	CLCP294	2168.42	206.00	233.68	22.20	< 4.18	< .40	< 1.25	< .12	< 5.22	< .50
Crane Lake	CLCP3	7352.11	522.00	156.34	11.10	13.55	0.96	< 1.68	< .12	< 6.97	< .50
Crane Lake	CLCP394	2329.67	212.00	232.97	21.20	5.36	0.49	1.70	0.16	< 5.45	< .50
Crane Lake	CLCP4	1661.02	392.00	101.69	24.00	< 1.67	< .39	< .50	< .12	< 2.09	< .49
Crane Lake	CLCP494	2296.80	503.00	276.26	60.50	2.39	0.52	< .55	< .12	< 2.28	< .50
Crane Lake	CLCP94	1833.33	231.00	223.02	28.10	< 3.17	< .40	1.03	0.13	< 3.96	< .50
Gimlet Lake	CLGLP1	2639.34	161.00	216.39	13.20	< 6.54	< .40	4.21	0.26	< 8.18	< .50
Gimlet Lake	CLGLP2	2310.61	305.00	190.15	25.10	< 2.98	< .39	5.33	0.70	5.17	0.68
Gimlet Lake	CLGLP3	2368.93	244.00	228.16	23.50	< 3.87	< .40	3.05	0.31	7.17	0.74
Gimlet Lake	CLGLP4	2000.00	202.00	434.65	43.90	< 3.91	< .39	1.61	0.16	< 4.89	< .49
Gimlet Lake	CLGP294	2392.86	268.00	109.82	12.30	< 3.55	< .40	1.97	0.22	< 4.45	< .50
Gimlet Lake	CLGP94	835.05	162.00	85.05	16.50	< 2.05	< .40	1.36	0.26	< 2.56	< .50
Hackberry Lake	CLHP294	2384.00	298.00	213.60	26.70	< 3.19	< .40	3.24	0.40	5.23	0.65
Hackberry Lake	CLHP394	2595.24	327.00	114.29	14.40	< 3.13	< .39	2.03	0.26	5.08	0.64
Hackberry Lake	CLHP94	2698.98	529.00	284.69	55.80	2.17	0.43	< .61	< .12	< 2.54	< .50
Island Lake	CLIP1	796.51	137.00	170.35	29.30	< 2.29	< .39	< .70	< .12	< 2.87	< .49
Island Lake	CLIP2	2761.90	174.00	292.06	18.40	< 6.32	< .40	< 1.89	< .12	< 7.89	< .50
Island Lake	CLIP294	2617.51	568.00	502.30	109.00	< 1.83	< .40	< .55	< .12	< 2.29	< .50
Island Lake	CLIP3	2079.07	447.00	256.28	55.10	< 1.84	< .39	< .55	< .12	< 2.30	< .49
Island Lake	CLIP394	1868.97	542.00	228.62	66.30	< 1.36	< .39	1.20	0.35	7.62	2.21
Island Lake	CLIP94	5129.25	754.00	361.22	53.10	< 2.71	< .40	0.97	0.14	< 3.05	< .45

Table 3. Continued

Sample Location	Sample Number	Se		Sr		V		Zn	
		ppm D.W.	ppm W.W.	ppm D.W.	ppm W.W.	ppm D.W.	ppm W.W.	ppm D.W.	ppm W.W.
Blue Lake	CLBP1	< 1.34	< .20	16.96	2.51	< .34	< .05	6.37	0.94
Blue Lake	CLBP2	< 1.04	< .20	16.11	3.06	0.34	0.06	5.79	1.10
Crane Lake	CLCP1	< 2.78	< .20	74.58	5.37	< .69	< .05	12.43	0.89
Crane Lake	CLCP2	3.19	0.23	78.89	5.68	0.69	0.05	28.33	2.04
Crane Lake	CLCP294	< 2.08	< .20	60.53	5.75	1.15	0.11	12.84	1.22
Crane Lake	CLCP3	4.35	0.31	215.49	15.30	0.89	0.06	20.70	1.47
Crane Lake	CLCP394	3.24	0.30	66.70	6.07	8.68	0.79	22.86	2.08
Crane Lake	CLCP4	< .83	< .20	77.54	18.30	2.33	0.55	27.46	6.48
Crane Lake	CLCP494	0.95	0.21	83.56	18.30	3.03	0.66	16.30	3.57
Crane Lake	CLCP94	< 1.59	< .20	46.27	5.83	1.91	0.24	11.51	1.45
Gimlet Lake	CLGLP1	< 3.28	< .20	72.95	4.45	5.56	0.34	16.33	1.00
Gimlet Lake	CLGLP2	< 1.49	< .20	73.26	9.67	10.45	1.38	12.80	1.69
Gimlet Lake	CLGLP3	< 1.94	< .20	96.80	9.97	15.44	1.59	55.63	5.73
Gimlet Lake	CLGLP4	< 1.96	< .20	65.15	6.58	3.14	0.32	15.05	1.52
Gimlet Lake	CLGP294	< 1.78	< .20	70.18	7.86	8.62	0.97	30.71	3.44
Gimlet Lake	CLGP94	< 1.03	< .20	30.62	5.94	7.47	1.45	9.18	1.78
Hackberry Lake	CLHP294	< 1.60	< .20	66.88	8.36	16.16	2.02	18.24	2.28
Hackberry Lake	CLHP394	< 1.57	< .20	81.75	10.30	9.52	1.20	16.11	2.03
Hackberry Lake	CLHP94	< 1.02	< .20	114.29	22.40	6.79	1.33	16.68	3.27
Island Lake	CLIP1	< 1.16	< .20	26.74	4.60	< .29	< .05	10.00	1.72
Island Lake	CLIP2	< 3.16	< .20	63.65	4.01	6.08	0.38	30.32	1.91
Island Lake	CLIP294	< .92	< .20	111.06	24.10	0.98	0.21	9.91	2.15
Island Lake	CLIP3	< .92	< .20	91.16	19.60	4.84	1.04	26.74	5.75
Island Lake	CLIP394	< .68	< .20	110.00	31.90	3.55	1.03	16.14	4.68
Island Lake	CLIP94	< 1.36	< .20	426.53	62.70	1.82	0.27	10.20	1.50

REFERENCES

- Albers, P.H. and M.B. Camardese. 1993. Effects of acidification on metal accumulation by aquatic plants and invertebrates. 1. Constructed wetlands. *Environmental Toxicology and Chemistry* 12:959-967.
- Barr, J.F. 1986. Population dynamics of the common loon (*Gavia immer*) associated with mercury-contaminated waters in northwestern Ontario. Occasional Paper 56. Canadian Wildlife Service, Ottawa, Ontario, Canada. 23 pp.
- Barron, M.G. 1995. Bioaccumulation and Bioconcentration in Aquatic Organisms. In D. J. Hoffman, B.A. Rattner, G.A. Burton Jr and J. Cairns Jr eds, *Handbook of Ecotoxicology*. Lewis, Ann Arbor, MI, USA, pp 652-666.
- Beyer, W.N., G. Miller and J.W. Simmers. 1990. Trace Elements in soil and biota in confined disposal facilities for dredged material. *Environmental Pollution* 65:19-32.
- Birge, W.J. and J.A. Black. 1980. Aquatic toxicology of nickel. In J.O. Nriagu ed. *Nickel in the Environment*, John Wiley and Sons, NY, New York, USA, pp 349-366.
- Crowder, A. 1991. Acidification, metals and macrophytes. *Environmental Pollution* 71:171-203.
- Demayo, A., M.C. Taylor, K.W. Taylor, and P.V. Hodson. 1982. Toxic effects of lead and lead compounds of human health, aquatic life, wildlife plants, and livestock. *CRC Critical Reviews in Environmental Controls* 12(4):257-305.
- Eisler, R. 1985a. Cadmium hazards to fish, wildlife, and invertebrates: a synoptic review. U.S. Fish and Wildlife Service Biological Report 85(1.2). 46 pp.
- Eisler, R. 1985b. Selenium hazards to fish, wildlife, and invertebrates: a synoptic review. U.S. Fish and Wildlife Service Biological Report 85(1.5). 42 pp.
- Eisler, R. 1986. Chromium hazards to fish, wildlife, and invertebrates: a synoptic review. U.S. Fish and Wildlife Service Biological Report 85(1.6). 60 pp.
- Eisler, R. 1987. Mercury hazards to fish, wildlife, and invertebrates: a synoptic review. U.S. Fish and Wildlife Service Biological Report 85(1.10). 90 pp.
- Eisler, R. 1988. Arsenic hazards to fish, wildlife, and invertebrates: a synoptic review. U.S. Fish and Wildlife Service Biological Report 85(1.12). 92 pp.

- Eisler, R. 1989. Molybdenum hazards to fish, wildlife, and invertebrates: a synoptic review. U.S. Fish and Wildlife Service Biological Report. 85(1.19). 61 pp.
- Eisler, R. 1990. Boron hazards to fish, wildlife, and invertebrates: a synoptic review. U.S. Fish Wildlife Service Biological Report. 85(1.20). 32 pp.
- Eisler, R. 1993. Zinc hazards to fish, wildlife, and invertebrates: a synoptic review. U.S. Fish Wildlife Service Biological Report. 10 (26). 106 pp.
- Eisler, R. 1994. A review of arsenic hazards to plants and animals with emphasis on fishery and wildlife resources. In J. O. Nriagu eds. Arsenic in the Environment, Part II: Human Health and Ecosystem Effects. John Wiley and Sons, New York, NY, USA, pp 185-259.
- Eisler, R. 1997. Copper hazards to fish, wildlife, and invertebrates: a synoptic review. U.S. Geological Survey, Biological Resources Division, Biological Science Report USGS/BRD/BSR--1997-0002. 98pp.
- Finley, M.T., M.P. Dieter and L.N. Locke. 1976. Sublethal effects of chronic lead ingestion in mallard ducks. *Journal of Toxicology and Environmental Health* 1:929-937.
- Garg, P., and P. Chandra. 1990. Toxicity and accumulation of chromium in Ceratophyllum demersum L. *Bulletin of Environmental Contamination and Toxicology* 44:473-478.
- Heinz, G.H., D.J. Hoffman, and L.G. Gold. 1989. Impaired reproduction of mallards fed an organic form of selenium. *Journal of Wildlife Management* 53:418-428.
- International Programme on Chemical Safety. 1990. Environmental Health Criteria # 107 Barium. World Health Organization, Geneva, Switzerland. 99 pp.
- International Programme on Chemical Safety. 1990. Environmental Health Criteria # 106 Beryllium. World Health Organization, Geneva, Switzerland. 210 pp
- Jenkins, D.W. 1980. Nickel accumulation in aquatic biota. In J.O. Nriagu ed. Nickel in the Environment. John Wiley and Sons, New York, NY, USA, pp 283-337.
- Kubota, J. 1977. Molybdenum status of the United States soils and plant. In W.R. Chappell and K.K. Petersen eds. Molybdenum in the environment volume 2: the geochemistry, cycling and industrial uses of molybdenum. Marcel Dekker Inc. New York, NY USA, pp 555-581.
- La Baugh, J.W. 1986. Limnological characteristics of selected lakes in the Nebraska Sandhills, U.S.A., and their relation to chemical characteristics of adjacent groundwater. *Journal of Hydrology* 86:279-298.

- Lemly, A.D., and G.J. Smith. 1987. Aquatic cycling of selenium: implications for fish and wildlife. U.S. Department of the Interior, Fish and Wildlife Service/ Fish and Wildlife Leaflet 12. 10 pp.
- Lemly, A.D. 1993. Guidelines for evaluation selenium data from aquatic monitoring and assessment studies. *Environmental Monitoring and Assessment* 28:83-100.
- Lemly, A. D. 1996. Selenium in Aquatic Organisms. In Beyer, G.H. Heinz and A.W. Redmon-Norwood, eds., *Environmental Contaminants in Wildlife, Interpreting Tissue Concentrations*. Lewis, Boca Raton, FL, USA, pp 427-446.
- Long, E.R., and L.G. Morgan. 1990. The potential for biological effects of sediment-sorbed contaminants tested in the national status and trends program. National Oceanic and Atmospheric Administration NOAA Technical Memorandum NOS.OMA 52. 175 pp.
- McCarragher, D.B. 1977. Nebraska's Sandhills Lakes. Nebraska Game and Parks Commission, Lincoln, NE, USA, 67 pp.
- McIntosh, A. 1991. Trace metals in freshwater sediments: a review of the literature and an assessment of research needs. In M.C. Newman and A.W. McIntosh eds., *Metal Ecotoxicology: Concepts and Applications*, Lewis, Chelsea, MI, USA, pp. 243-260.
- Moore, J.W. 1991. *Inorganic Contaminants in Surface Waters: Research and Monitoring Priorities*. Springer-Verlag, New York, NY, USA. 334 pp.
- National Academy of Sciences. 1977. Arsenic. National Academy of Science. Washington, D.C.
- National Research Council; Committee on Medical and Biological Effects of Environmental Pollutants. 1979. Iron. University Park Press, Baltimore, MD, USA. 248 pp.
- Novotny, V., and H. Olem. 1994. *Water Quality Prevention, Identification, and Management of Diffuse Pollution*. Van Nostrand Reinhold, NY, USA. 1054 pp.
- Pain, D.J. 1996. Lead in waterfowl. In Beyer, G.H. Heinz and A.W. Redmon-Norwood, eds., *Environmental Contaminants in Wildlife, Interpreting Tissue Concentrations*. Lewis, Boca Raton, FL, USA, pp 251-264.
- Pain, D.J. 1995. Lead in the environment. In D.J. Hoffman, B.A. Rattner, G.A. Burton Jr and J. Cairns Jr eds, *Handbook of Ecotoxicology*. Lewis, Ann Arbor, MI, USA, pp 356-391.
- Pais, I. And J.B. Jones. 1997. *The Handbook of Trace Elements*. St. Lucie Press, Boca Raton, FL. 223 pp.

- Parker, D.R., L.W. Zelazny, and T.B. Kinraide. 1989. Chemical speciation and plant toxicity of aqueous aluminum. In T.E. Lewis ed. *Environmental Chemistry and Toxicology of Aluminum*, Lewis, Chelsea MI, USA, pp 117-145.
- Powell, R.L. et al. 1997. Ecological risk assessment of a wetland exposed to boron. *Environmental Toxicology and Chemistry* 16:2409-2414.
- Saiki, M.K., M.R. Jennings, and W.G. Brumbaugh. 1993. Boron, Molybdenum, and selenium in aquatic food chains from the Lower San Jauquin River and its tributaries, California. *Archives of Environmental Contamination and Toxicology* 24:307-319.
- Shacklette, H. T. and J. G. Boerngen. 1984. Element concentrations in soils and other surficial materials of the conterminous United States. U. S. Geological Survey, Washington D.C. Professional Paper 1270, 105p.
- Smith, G.J., and V.P. Anders. 1989. Toxic effects of boron on mallard reproduction. *Environmental Toxicology and Chemistry* 8:943-950.
- Sorenson, E.M.B. 1991. *Metal Poisoning in Fish*. CRC Press, Ann Arbor, MI, USA. 374 pp.
- Stokes, P.M. 1979. Copper accumulations in freshwater biota. In J.O. Nriagu ed. *Copper in the Environment, Part 1: Ecological Cycling*. John Wiley and Sons Inc., New York, USA pp. 357-381.
- Stubblefield, W.A., S.F. Brinkman, P.H. Davies, T.D. Garrison, J.R. Hockett, and M.W. McIntyre. 1997. Effects of water hardness on the toxicity of manganese to developing brown trout (*Salmo trutta*). *Environmental Toxicology and Chemistry* 16:2082-2089.
- Tanner, C.C. and J.S. Clayton. 1990. Persistence of arsenic 24 years after sodium arsenite herbicide application to Lake Rotorua, Hamilton, New Zealand. *New Zealand Journal of Marine Freshwater Research* 24:173-179.
- Thompson, D.R. 1996. Mercury in birds and terrestrial mammals. In Beyer, G.H. Heinz and A.W. Redmon-Norwood, eds., *Environmental Contaminants in Wildlife, Interpreting Tissue Concentrations*. Lewis, Boca Raton, FL, USA, pp 341-376.
- U.S. Environmental Protections Agency. 1980. Ambient water quality criteria for beryllium: Washington D.C.. Office of Water Regulations and Standards, EPA 440/5-80-024.
- U.S. Environmental Protections Agency. 1985a. Ambient water quality criteria for chromium: Washington D.C.. Office of Water Regulations and Standards, EPA 440/5-84-029.

- U.S. Environmental Protection Agency. 1985b. Ambient water quality criteria for copper: Washington D.C.. Office of Water Regulations and Standards, EPA 440/5-84-031.
- U.S. Environmental Protection Agency. 1986. Ambient water quality criteria for nickel: Washington D.C.. Office of Water Regulations and Standards, EPA 440/5-86-004.
- U.S. Environmental Protection Agency. 1987. Ambient water quality criteria for zinc: Washington D.C.. Office of Water Regulations and Standards, EPA 440/5-87-003.
- U.S. Environmental Protection Agency. 1992. Ecological effects of soil lead contamination. Toxic Integration Branch. Office of Emergency and Remedial Response.
- Van Derveer, W.D. and S.P. Canton. 1997. Selenium sediment toxicity thresholds and derivation of water quality criteria for freshwater biota of western streams. *Environmental Toxicology and Chemistry* 16:1260-1268.
- Wetzel, R.G. 1983. *Limnology* second edition. Harcourt Brace College Publishers, Ft. Worth, TX, USA. 767 pp.
- Wiener, J.G., and D.J. Spry. 1996. Toxicological Significance of Mercury in Freshwater Fish. In Beyer, G.H. Heinz and A.W. Redmon-Norwood, eds., *Environmental Contaminants in Wildlife, Interpreting Tissue Concentrations*. Lewis, Boca Raton, FL, USA, pp 297-339.
- Wilber, C.G. 1980. *Beryllium- A Potential Environmental Contaminant*. Charles C. Thomas Publishing, Springfield Illinois, USA.
- Wile, M.I. and G.G. Hitchin. 1983. Patterns of accumulation of selected metals in members of the soft-water macrophyte flora of Central Ontario lakes. *Aquatic Botany* 15:53-64.
- Wren, C.D., S. Harris and N. Hartrup. 1995. Exotoxicology of Mercury and Cadmium. In D. J. Hoffman, B.A. Rattner, G.A. Burton Jr and J. Cairns Jr eds, *Handbook of Ecotoxicology*. Lewis, Ann Arbor, MI, USA, pp 392-423.