Stillwater National Wildlife Refuge
Wetland Contaminant Monitoring

(Regional Study ID: 93-1C02A)

by

Peter L. Tuttle, Nevada State Office

C. Anne Janik, Stillwater National Wildlife Refuge

and

Stanley N. Wiemeyer, Nevada State Office

Region One, Fish and Wildlife Service

July 1996

EXECUTIVE SUMMARY

The quantity and quality of wetlands in Lahontan Valley have declined since the initiation of large-scale agriculture in Lahontan Valley in the early 1900's. Recent investigations have also docwnented concentrations of inorganic contaminants in water, sediment, and biological samples in excess of concentrations associated with adverse effects to fish and wildlife. Under the auspices of the Truckee-Carson-Pyramid Lake Water Settlement Agreement, the Department of the Interior has implemented a program to acquire rights for water to restore and maintain a portion of the historic wetlands in Lahontan Valley. Although inflow to wetlands will be partially restored, the effects of increased inflow to wetland contamination is uncertain. In 1990, the Nevada State Office, in conjunction with Stillwater National Wildlife Refuge (NWR), instituted a monitoring program on Stillwater NWR to assess implications of inorganic contaminants to fish, wildlife, and hwnan health and to assess changes in inorganic contaminant concentrations in wetlands maintained with freshwater and agricultural drainage water. This program entailed the measurement of water quality parameters and the determination of trace element concentrations in water, sediment, and biological matrices.

This study found that dissolved solids, as measured by specific conductance, and 12 of 19 chemical elements monitored in the study exceeded levels of concern in 10% or more of the water, sediment and/or biological samples. Total dissolved solids, and the elements alwninwn, arsenic, boron, mercury, and zinc frequently exceeded concentrations associated with adverse biological effects. Chromiwn, copper, iron, lead, molybdenwn, and seleniwn were found at lower levels of concern. Mercury, which exceeded recommended levels for hwnan consumption in fish and waterfowl, was the only element that appeared to pose a direct threat to hwnan health.

Our study identified statistical correlations between concentrations of a nwnber of trace elements in unfiltered water and concentrations in sediment and/or biological matrices. This suggests that a reduction of certain trace element concentrations in sediment and food chains could be achieved through control of trace element concentrations in water. Correlations between specific conductance and concentrations of some elements in unfiltered water were also identified. Relationships were strongest for arsenic, boron, and molybdenwn, although there were weaker correlations for copper and zinc. This suggests that a reduction of total dissolved solids in water would contribute to a reduction of concentrations of certain trace elements in water, and therefore, in sediment and food chains. Again, the greatest reductions may be realized in the concentrations of arsenic, boron, and molybdenwn. Additionally, elevated arsenic, molybdenwn, and copper have been associated with toxicity of water in Lahontan Valley. Therefore, control of specific conductance has the potential to reduce toxicity of water to aquatic organisms. The lack of correlations between specific conductance and concentrations of alwninwn and mercury in unfiltered water indicate that control of dissolved solids would be ineffective in controlling concerns with these elements.

Recommendations of this study include:

1) To the extent possible and practicable, flow-through management of marshes should be restored. Such management would promote the movement of waterborne contaminants through sequential wetlands. Because contaminant mobility, concentration, and accwnulation are related to water volwne and hydrologic retention time, the best results may be attained through the movement of larger volwnes of water. Therefore, delivery of

water through wetland units, as opposed to delivery via canals running parallel to natural flow gradients, should be considered. Additionally, the degradation of water quality **in** sequential wetlands would promote restoration of habitat variability. Therefore, management of water in this manner may further the objective of Stillwater NWR management, as defined **in** the Truckee-Carson-Pyramid Lake Water Settlement Agreement, to "restore and protect natural biological diversity."

2) Hydrologic retention time **in** wetlands will affect concentration of dissolved solids, and, therefore, concentrations of some agricultural drainage-related trace elements. Measures to reduce water retention time **in** wetlands should be considered. One method of reducing hydrologic retention time is the reduction of impoundment sizes.

3) The relationship between specific conductance and some agricultural drainage-related trace elements indicates that specific conductance may be used as a cost-effective tool for the monitoring and management of dissolved solids and some trace elements. Stillwater NWR should consider instituting a specific conductance monitoring program. Monitoring data would be necessary to define and refine management practices.

4) Mercury **in** Stillwater NWR wetlands may present the greatest risk to migratory birds and humans. Our study **did** not indicate that mercury impacts could be reduced through increasing flows through wetlands, reduction of impoundment sizes, or managing specific conductance. Desiccation of wetlands to volatilize mercury should be further evaluated. Because mercury contamination on Stillwater NWR appears to be confined to a few wetlands, permanent desiccation of mercury-contaminated wetlands may also be considered.

TABLE OF CONTENTS

LIST **OF** TABLES

LIST OF FIGURES

INTRODUCTION

Concern for the quality and quantity of wetlands in arid regions of the United States has increased during recent years. Development and conversion of wetlands to agricultural land has directly contributed to major losses of historic wetlands in several western states. Diversion of water prior to entering wetlands has resulted in desiccation of some wetlands and alteration of natural hydrologic characteristics of others. In some cases, previously flow-through wetlands have become hydrologically isolated and serve as sinks in local hydrographic subbasins. Human modifications of natural hydrologic regimes has also caused substantial changes in the biogeochemical cycling of many dissolved constituents (Lemly et al. 1993). Drainage from agricultural areas, including operational spills, surface runoff from fields, and subsurface drainage, may constitute a significant percentage of water entering wetlands. Such water may contain elevated concentrations of dissolved solids, including a variety of major and trace elements, which have been mobilized from soils or local groundwater. Evaporative water loss, including surface evaporation and evapotranspiration, may concentrate dissolved solids in water. Certain trace elements have also been found to accumulate in aquatic organisms and magnify in food chains. In several examples, dissolved solids and trace elements have become concentrated to toxic levels in water and food chain organisms in wetlands (Schroeder et al. 1988; Hoffman et al. 1990b; Moore et al. 1990; Setrnire et al. 1990). Elevated concentrations of certain trace elements have been associated with reproductive failure, malformation of embryos, and mortality of large numbers of fish and migratory birds in some areas. Wetlands occurring at the termini of agricultural systems are particularly vulnerable to these impacts.

In western Nevada, wetlands located near the terminus of the Carson River in Lahontan Valley, Churchill County, have been impacted by agricultural activities. This area includes Stillwater National Wildlife Refuge (NWR). Wetland acreage, water quality, wetland productivity, and biological diversity have been affected (Hoffman et al. 1990, Kerley et al. 1993, Hoffman 1994). In several cases, concentrations of dissolved solids and trace elements exceeded concentrations associated with adverse effects to fish and wildlife. To partially restore and maintain wetlands in Lahontan Valley, the Department of the Interior has implemented a water rights acquisition program. However, the effects of increasing freshwater inflows to wetlands to trace element concentrations in wetland components were uncertain. In 1990, the Nevada State Office, in conjunction with Stillwater NWR, instituted a monitoring program on Stillwater NWR to assess implications of inorganic contaminants to fish, wildlife, and human health and to assess changes in inorganic contaminant concentrations in wetlands maintained with freshwater and agricultural drainage water. This report presents findings of this monitoring program.

STUDY AREA

Stillwater NWR is located in Lahontan Valley in the lower Carson River drainage basin (Figure 1). The Carson River basin encompasses 10,300 square kilometers in west-central Nevada and east-central California. The climate in Lahontan Valley is that of a mid-latitude desert. Temperatures range from 4°C to 41°C (U.S. Bureau of Reclamation 1990). Annual average precipitation is approximately 13 centimeters (cm) and the rate of evapotranspiration ranges from 137 to 152 cm annually (Morgan 1982). Geology of the valley is characterized by unconsolidated, fine-grained lake and playa deposits, fan gravels, and delta deposits

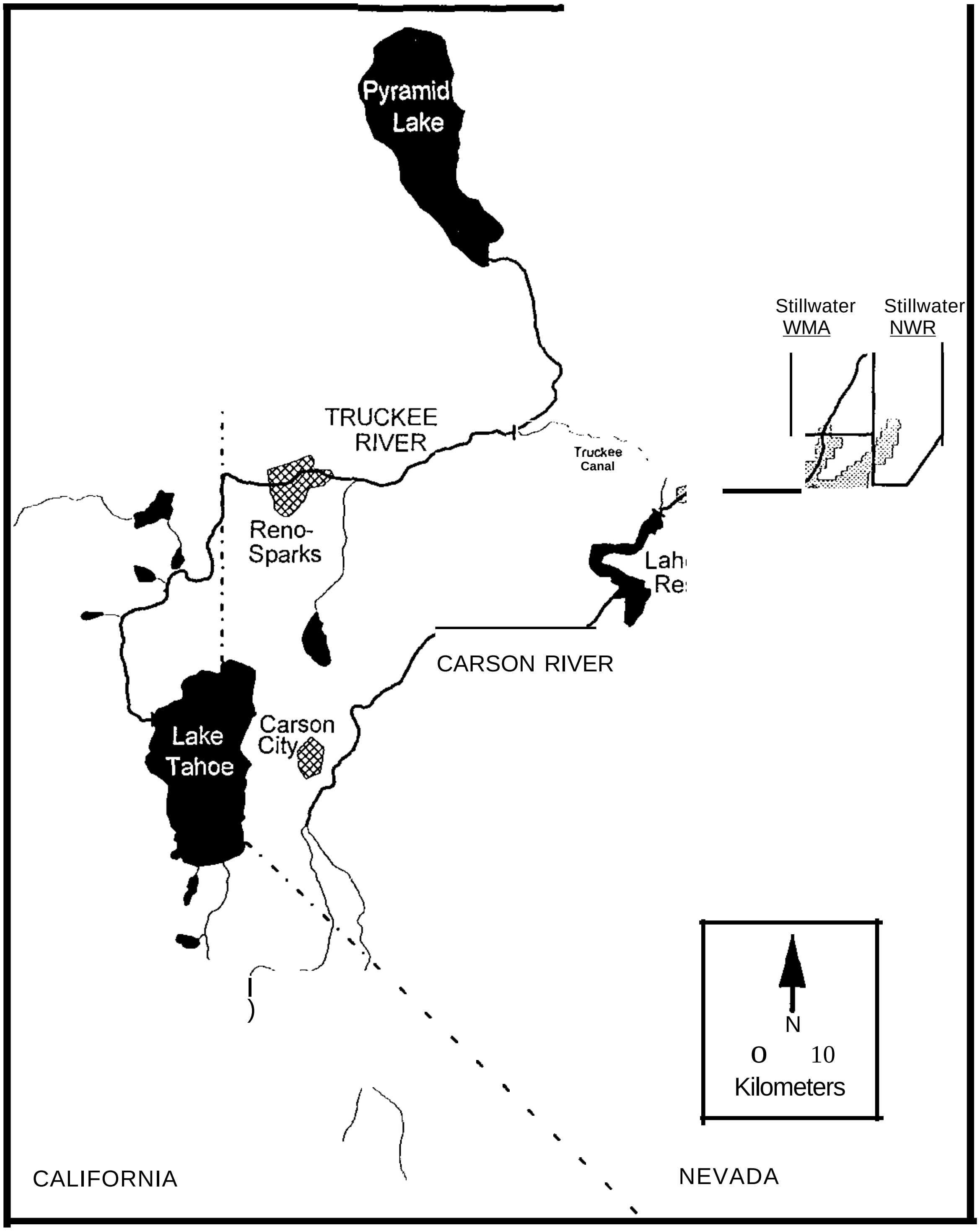

Figure 1. Map of the Carson and Truckee River drainages showing Stillwater National Wildlife Refuge, Churchill County, Nevada.

(Hoffman et al. 1990b). Surrounding mountains are composed of igneous, sedimentary, and metamorphic rock. Soils in the area are typically alkaline and consist primarily of sand and clay of medium texture (U.S. Bureau of Reclamation 1990).

Historically, Lahontan Valley wetlands were primarily maintained by flows from the Carson River. Because Carson River flow originates from snowmelt from the eastern slopes of the Sierra Nevada, annual and seasonal wetland inflow are highly variable. High discharge typically corresponds with peak snowmelt. Wetlands in Lahontan Valley historically encompassed an estimated annual average of 60,705 hectares (ha); however, because Carson River discharge varied, wetland size may have exceeded 80,940 ha at times (Kerley et al. 1993). Within Lahontan Valley, Carson River water flowed uncontrolled through a network of natural channels and sloughs to support a series of wetlands. The river channel meandered throughout the Valley and periodically switched course. The Carson River flowed through three primary river channels since the mid-1800s (Hoffinan 1994). River water most often flowed first to Carson Lake, then to Stillwater Marsh via Stillwater Slough (Kerley et al. 1993). Water flowing from Stillwater Marsh discharged to the Carson Sink.

Little technical information is available on historical water quality of Lahontan Valley wetlands. However, historical accounts of the abundance and diversity of plants, fish, and wildlife indicate that overall water quality was better than current conditions (Simpson 1876; Russell 1885; Bailey 1898; Schmitt 1988). Water entering Lahontan Valley wetlands under historic conditions was estimated to have a total dissolved solids (TDS) concentration of 161 mgIL (Kerley et al. 1993). The TDS concentration of water historically entering Stillwater Marsh was estimated to be 237 mgIL. Frequent flushing of marshes under unrestricted flow conditions minimized the accumulation of dissolved solids and maintained the chemical integrity of the wetlands.

Little quantitative information on historic wildlife use of Lahontan Valley wetlands is available. However, accounts of early explorers suggest a rich and diverse biological assemblage (Kerley et al. 1993). More recently, the value of these ecologically important wetlands have been recognized (Thompson and Merritt 1988). Wetlands in Lahontan Valley provide habitat for the largest and most diverse assemblage of migratory and wetland-dependent birds in Nevada. As many as 206 avian species, most of which are migratory, have been identified in Lahontan Valley. These ecologically important wetlands provide foraging, nesting, and staging habitat for waterfowl, shorebirds, and colonial nesting birds on the Pacific Flyway. Because of its central location, wetlands in Lahontan Valley have been identified as a key point on this migratory route. Annual use in recent years has included 75% of Nevada's ducks, 50% of the Canada geese *(Branta canadensis),* and 65% of the tundra swans *(Cygnus colombianus;* Thompson and Merritt 1988). Waterfowl usage of Stillwater NWR in recent years has been estimated at almost 15 million bird use-days annually. This is down from an estimated 27 million bird use-days in the early 1970s. Under optimal conditions up to 15,000 waterfowl are produced annually. Because of the importance to migratory shorebirds, wetlands in Lahontan Valley were designated a Hemispheric Shorebird Reserve. As many as 250,000 shorebirds and numerous other wading and aquatic birds may use Lahontan Valley wetlands annually (Thompson and Merritt 1988).

One endangered, one threatened, and six species of concern (former category 2 candidate avian species proposed for listing under the Endangered Species Act of 1973, as amended) have been identified in Lahontan Valley. The Valley provides wintering habitat for bald eagles *(Haliaeetus leucocephalus).* As many as 70 bald eagles have been reported in recent years. American peregrine falcons *(Falco peregrinus anatum)* have been observed in Lahontan Valley, but are considered uncommon migrants in the area (Herron et al. 1985). Five of the species of concern, white-faced ibis *(Plegadis* chihi), trumpeter swan *(Cygnus buccinator),* black tern *(Chlidonias niger),* western least bittern *(Ixobrychus exilis),* and western snowy plover *(Charadrius alexandrinus),* are primarily marsh inhabitants.

Concern for loss and degradation of Lahontan Valley wetlands prompted protection efforts as early as 1935. Stillwater NWR and Wildlife Management Area (WMA) were created in 1948 under a cooperative agreement between the U.S. Fish and Wildlife Service, the Nevada Department of Wildlife, and the Truckee-Carson Irrigation District (the custodian of the Newlands Project, a Department of the Interior-sponsored irrigation project) in an effort to protect and manage the remaining marsh. This agreement designated approximately 90,650 ha of Bureau of Reclamation land as the Stillwater WMA. Approximately 9,710 ha of this area was designated as the Stillwater NWR, a non-hunting sanctuary. Since the creation of the Newlands Project in 1902, wetlands in Lahontan Valley have been primarily maintained by agricultural drainwater and water spilled from Lahontan Reservoir for power generation. Precautionary releases have occasionally provided water. Major drains discharging to Stillwater NWR include Diagonal, Paiute, Paiute Diversion, TJ, and Hunter drains, as well as Stillwater Slough. A system of water delivery canals, dikes, and levees have been constructed on Stillwater NWR to effectively manage wetlands. Water has been impounded in 16 main marsh units. These units have been managed as permanent and seasonally flooded marshes. Water levels have been controlled to afford the greatest benefit of available water. Goals of marsh management are to provide maximum production of aquatic and emergent vegetation for food and nesting habitat for migratory birds. Water management goals have also included the prevention of avian disease epidemics such as cholera and botulism.

Under the accords of the Truckee-Carson-Pyramid Lake Water Settlement Agreement (Title II of Public Law 101-618), the size of Stillwater NWR was increased to approximately 31,370 ha through the transfer of lands from Stillwater WMA. The management objectives of Stillwater NWR were amended under Public Law (P.L.) 101-618. Management objectives now include:

1. Maintaining and restoring natural biological diversity within the refuge;
2. Providing for the conservation and management of fish and wildlife and their habitats within the refuge;
3. Fulfilling the international treaty obligations of the United States with respect to fish and wildlife; and,
4. Providing opportunities for scientific research, environmental education, and fish and wildlife oriented recreation.

P.L. 101-618 also authorized the acquisition of sufficient water to maintain a long-term average of 10,120 ha of wetlands in Lahontan Valley, where approximately 5,670 ha of wetlands are to be maintained on Stillwater NWR. Rights to irrigation-quality water will be acquired from willing sellers in the Carson Division of the Newlands Project. In 1986 and 1987, the median TDS concentration of irrigation-quality water released from Lahontan Reservoir was 168 mg/L (Hoffinan et aI1990b). However, P.L. 101-618 also stipulates that wetlands in Lahontan Valley will continue to receive agricultural drainwater.

CONTAMINANT CONCERNS

The quantity and quality of Lahontan Valley wetlands have declined sharply since the early 1900s, based upon investigations under the National Irrigation Water Quality Program from 1986 to 1990, with the creation of the Newlands Project being the most significant contributing factor (Hoffman et al. 1990b; Rowe et aJ. 1991; Lico 1992; Hallock and Hallock 1993; Hoffman 1994). Upstream water diversion and consumption has decreased the amount of water entering wetlands at the terminus of the Carson River. As a result, wetland acreage in Lahontan Valley has declined by an estimated 90% during the last 90 years (Kerley et al. 1993). Water diversion has also modified historic flow patterns and hydrologic characteristics. Since Newlands Project construction, Carson River flows have been impounded in Lahontan Reservoir. Water to support project operations has also been diverted from the Truckee River via the Truckee Canal. Water is released from Lahontan Reservoir to a system of delivery canals to flood-irrigate agricultural lands. As a result of water regulation and consumption, the flow-through characteristics of Lahontan Valley wetlands were eliminated and water is only discharged from Stillwater Marsh during periods of excessive precipitation and runoff. Modification of wetlands on Stillwater WMA and NWR also resulted in retention of available water for longer periods. As a result, flushing of dissolved solids from wetlands occurred with less frequency. This resulted in an accumulation of dissolved solids and salts, including potentially toxic trace elements, in wetlands.

Irrigation has also led to a rise in shallow ground-water table elevations. Shallow groundwater in Lahontan Valley is typically of poor quality (Lieo 1992). To counter this rise, a system of agricultural drains was also constructed to prevent loss of crop production. These drains discharged to down-gradient wetlands. Following regulation of the Carson River, wetlands have been maintained by drainwater and operational spills from the Newlands Project. Reliance on drainwater increased as a result of the 1973 Operational Criteria and Procedures for the Newlands Project. The primary factor was the elimination of power generation at Lahontan Dam during non-irrigation periods in an effort to reduce diversions from the Truckee River.

Newlands Project drainwater typically contains elevated concentrations of TDS, including several trace elements of concern to fish and wildlife. Concentrations of certain trace elements often exceeded Nevada water quality standards (Hoffman et al. 1990b). In 1986 and 1987, the median IDS concentration in water released from Lahontan Reservoir (irrigation quality water) was 168 mg/L. The median TDS concentration of water entering wetlands (primarily drainwater) was 1,590 mg/L. The maximum TDS concentration observed was 53,400 mg/L in a sample collected from Hunter Drain.

As a result of agricultural water consumption and increased loading of dissolved solids, water quality in Lahontan Valley wetlands has also significantly declined (Kerley et al. 1993). This has led to declines in the abundance and diversity of plants, fish, and wildlife in wetland areas. Concentrations of arsenic, boron, chromium, copper, lithium, mercury, molybdenum, selenium, zinc, dissolved solids, sodium, and unionized anunonia in water, sediment, and/or biota also have exceeded baseline levels, Federal and State criteria for the protection of aquatic life or the propagation of wildlife, and published fish and wildlife effect levels. These constituents naturally occur in soils of Lahontan Valley. With the exception of mercury, elevated levels are believed to be the result of agricultural activities. Elevated levels of mercury are the result of pre-l900 mining activities in the Carson River basin. Arsenic, boron, mercury, and selenium were identified as contaminants of primary concern. Water in certain drains was also found to be acutely toxic to test fish and invertebrates (Finger et al. 1993).

These findings identified a need for more comprehensive contaminant information for wetlands of Stillwater NWR. In 1990, Stillwater NWR biologists, in conjunction with the Division of Refuge Operations Support and the Nevada State Office, initiated a study to characterize the extent and degree of contamination on Service lands and to evaluate the effects of using irrigation quality water as opposed to drainwater for wetland management. This information would enable wildlife managers to develop management programs that would provide the greatest benefit to fish and wildlife resources. The objectives of the study were to:

1. Determine baseline concentrations of contaminants in water, bottom sediments, aquatic macrophytes, aquatic invertebrates, fish, and migratory bird eggs and livers in selected marsh units of the Stillwater NWR;

2. Monitor changes of contaminant concentrations in wetland components in marsh units managed with freshwater or irrigation drainage water;

3. Determine if contaminant concentrations in edible tissues from waterfowl representative of Stillwater NWR exceed the U.S. Environmental Protection Agency's (U.S. EPA) acceptable daily intake for selected trace elements in the human diet; and,

4. Provide information to assist refuge managers in water allocation and management decisions that support wildlife and habitat objectives of the Stillwater NWR.

METHODS

Water quality parameters and trace element concentrations in water, sediment, and biota were determined in four study wetlands to establish baseline levels and monitor changes. The study wetlands were: I) Dry Lake, 2) Lead Lake, 3) Swan Lake Check, and 4) East Alkali Lake (Figure 2). These units were selected based upon long-term water management plans at Stillwater NWR. These plans called for Dry Lake to receive irrigation-quality water delivered from Lahontan Reservoir. Dry Lake began receiving irrigation quality water exclusively in 1989. Lead Lake would be supported primarily with drainage water. Swan Lake Check and East Alkali Lake were to receive some mixture of irrigation-quality water and agricultural drainage water.

Study units were sampled twice annually. Samples were collected during peak waterfowl and shorebird nesting (May/June) to assess conditions and contaminants that may affect avian production. Samples were also collected during the peak of waterfowl and shorebird fall migration to assess conditions that may affect migrating birds. Water quality data and samples for trace element analysis were collected at study units during each sampling period. Within each study unit, five randomly selected stations were established within a permanent I ha

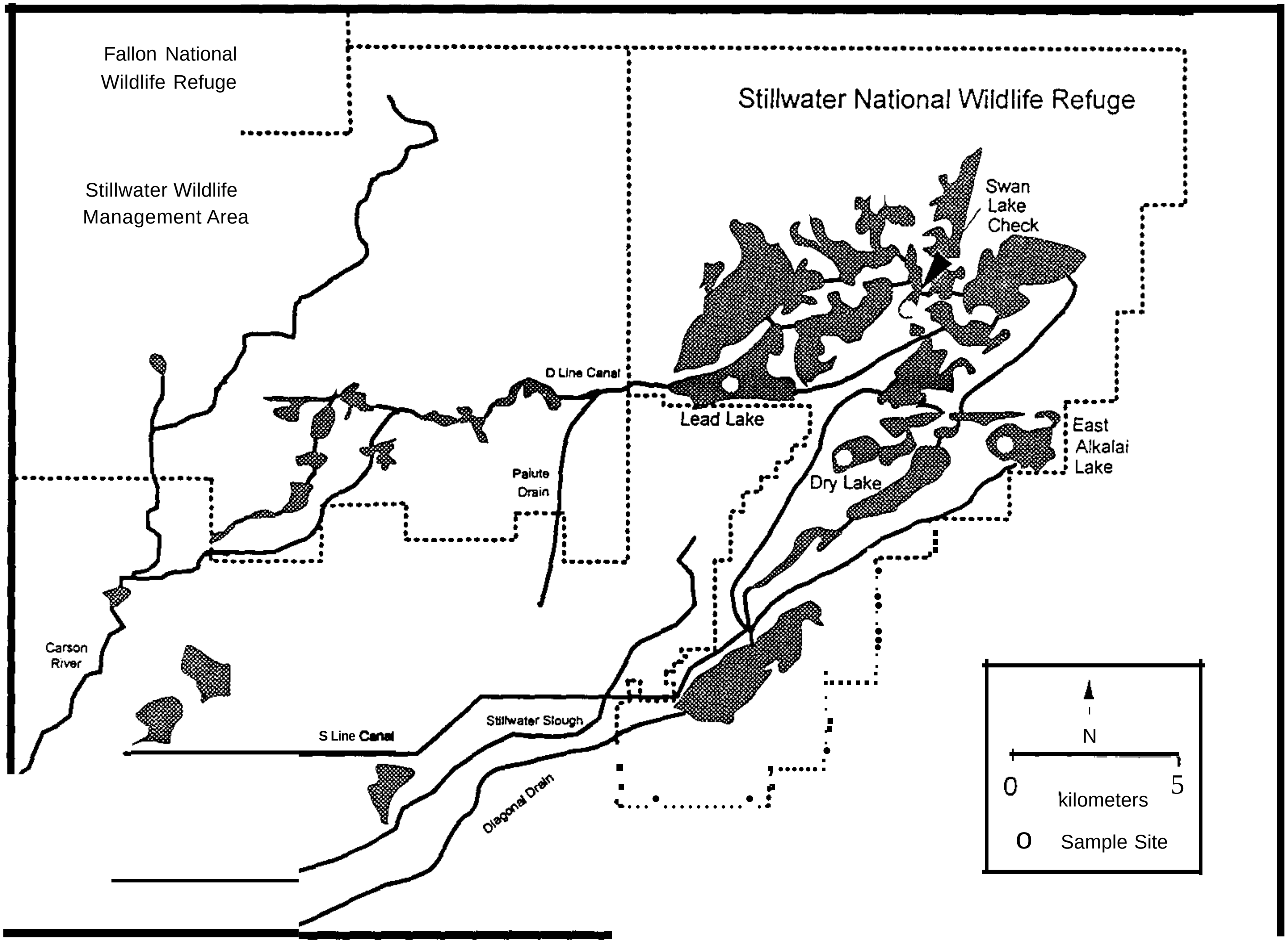

Figure 2. Map of Stillwater National Wildlife Refuge showing locations of wetlands monitored in the Stillwater National Wildlife Refuge Wetland Contaminant Monitoring Program.

sampling site, situated within a major aquatic plant bed, located near water inflow points. Sites were considered to be representative of feeding habitat for migratory and nesting aquatic birds based on existing bird-use data.

During the study period (1990 to 1993), precipitation in western Nevada and, subsequently, discharge in the Carson River below Lahontan Reservoir, was below normal. Total discharges in the Carson River below Lahontan Reservoir (U.S. Geological Survey [USGS] Gage 10312150) during 1990 to 1993 water years (October 1 through September 30) were 68%, 46%, 35%, and 72% of the average annual discharge, respectively. This regional drought, coupled with court-mandated increases in water-use efficiencies in the Newlands Project, resulted in less water delivered to wetlands throughout Lahontan Valley, including Stillwater NWR, than would have occurred in normal water years. This water shortage resulted in desiccation of much of the Lahontan Valley wetlands. In 1989, total wetland size was estimated at approximately 3,080 ha valley-wide. Total wetland size fell to less than 150 ha by 1992.

Lack of water and corresponding desiccation of wetlands precluded sampling of all wetlands in all sampling periods. With the exception of the initial period of sample collection (spring 1990), one or more study units were dry during scheduled sampling periods. Physical and chemical data and samples for contaminant analysis were not collected during these periods. Desiccation of units also affected establishment of plant, invertebrate, and fish communities. As a result, many of these sample matrices were not available in some units even though the unit was flooded at the time of sampling. Pondweed was available in only one drainwater unit during one sampling period. Therefore, evaluation of this matrix was restricted to Dry Lake. Fish were only available in Dry Lake on an intermittent basis and species composition was not consistent when available. Bird nesting attempts and production during the course of the study were also below normal, presumably from desiccation and degraded habitat conditions. Consequently, waterfowl and shorebird eggs and juvenile tissues were only collected during the spring of 1990. Therefore, fish, avian eggs, and avian livers were omitted from statistical evaluation. Because of limited observations, Lead Lake was also omitted from consideration.

Water shortages, which also affected water management on the refuge, affected results. During periods of low water inflow, water was retained in primary marsh units longer than it would have been in "normal" water years. Therefore, evaporative water losses (evaporation and evapotranspiration) contributed to a greater concentration of dissolved solids in water in these units than would have occurred in "normal" water years.

Water quality parameters, including temperature, dissolved oxygen, pH, specific conductance, and salinity, were monitored at all sampling stations within each study unit. Temperature was with a hand-held thermometer. Dissolved oxygen was measured with a Yellow Springs Instrument Model 57 Oxygen meter. An Orion Research Model SA250 pH meter was used to measure pH. Specific conductance and salinity were measured with a Yellow Springs Instrument Model 33 S-C-T Meter.

Whole water (unfiltered) and composite samples of sediment, aquatic vegetation, nektonic and benthic invertebrates, and fish were collected from each station for trace element analyses. Whole water, which included suspended detrital particulate matter, was collected to assess the total mass load of trace elements occurring in the water column and potentially available to organisms. Unfiltered water was collected in mid-water column with a Wheaton sampler. Water samples were acidified to pH ≤ 2.0 upon collection. Sediment samples (composite of 3 subsamples) were collected with a Wildco model number 2422 H12 core sampler. The top 3 cm of sediment were used. Samples of aquatic vegetation *(Potamegeton,* herein referred to as pondweed) were collected by hand. Nektonic invertebrates (corixids) and benthic invertebrates (chironomids) were collected with a kick net and later sorted in the laboratory. Tui chub *(Gila bic%r)*, carp *(Cyprinus carpio)*, and pumpkinseed *(Lepomis gibbosus)* were collected with unbaited minnow traps.

Eggs and livers of representative waterfowl and shorebirds were collected to assess trace element accumulation and relative threats to migratory birds. A maximum often egg and ten juvenile liver samples each of American coots *(Fulica americana)* and black-necked stilts *(Himantopus mexicanus)* were collected from each study unit during the spring nesting season. Pre-flighted juveniles were collected to ensure that exposure to, and accumulation of trace elements occurred at the study wetlands. To assess possible threats to human health, muscle and liver samples were taken from popular waterfowl game species that were collected during the hunting season. Full-grown green-winged teal *(Anas crecca)* and northern shovelers *(Anas clypeata)* were collected. Birds were collected using a shotgun with steel shot and dissected in the laboratory.

Water samples were placed in pre-cleaned 250 ml nalgene containers, stored on ice in the field and stored at 4 DC upon return to the laboratory. Sediment and pondweed samples were placed in pre-cleaned 60 ml glass containers with teflon-lined lids, stored on ice in the field, and frozen upon return to the laboratory. Fish and invertebrates were placed in 500 ml nalgene containers with teflon-lined lids, stored on ice in the field, and frozen upon return to the laboratory; samples were later thawed, sorted, and placed in glass containers with teflon-lined lids, and refrozen. Bird eggs were stored on ice in the field and stored at 4 DC upon return to the laboratory. Eggs were later opened using pre-cleaned stainless steel instruments in the laboratory and contents were placed in 60 ml glass containers with teflon-lined lids and frozen. Whole juvenile and adult birds were placed in plastic bags, stored on ice in the field, and frozen upon return to the laboratory. Birds were later thawed and appropriate tissues were removed using pre-cleaned stain1ess steel instruments. Bird tissues were placed in 60 ml glass containers with teflon-lined lids and frozen. Samples were shipped to the Environmental Trace Substances Research Center, Columbia, Missouri for chemical analysis. Water samples were shipped overnight in a cooler with blue ice. Frozen samples were shipped overnight in a cooler containing dry ice.

Chemical analyses were conducted following methods described in Patuxent Analytical Control Facility (1990). Arsenic and selenium concentrations were determined using hydride generation atomic absorption spectroscopy. Mercury concentrations were determined using cold vapor atomic absorption spectroscopy. Concentrations of aluminum, barium, beryllium, boron, cadmium, chromium, copper, iron, lead, magnesium, manganese, molybdenum, nickel, strontium, vanadium, and zinc were determined with inductively coupled plasma emission spectroscopy.

Quality control and laboratory performance was assessed and approved by Patuxent Analytical Control Facility personnel. Quality control techniques included the use of procedural blanks, duplicate samples, spiked samples, and reference materials. Laboratory performance was within the acceptable precision range prescribed in Patuxent Analytical Control Facility (1990).

Linear regression was used to examine relationships among specific conductance and trace element concentrations in unfiltered water and among trace element concentrations in unfiltered water and residues in sediment and biotic matrices. Significance was assigned to a level of $P \leq 0.05$. A value of one-half the detection limit was assigned to samples with contaminant concentrations less than the detection limit. If contaminant concentrations were below detection limits in more than 50 percent of the samples statistical examination was not attempted.

RESULTS AND DISCUSSION

RESIDUE CONCENTRATIONS AND IMPLICATIONS TO BIOTA

Implications to fish and wildlife were assessed through exceedances of State and Federal water quality standards and exceedances of published fish and wildlife effect levels. Table 1 provides Nevada standards for protection of aquatic life or other beneficial uses for selected trace elements in water. Table 2 provides selected concern and effect concentrations for invertebrates, fish, and wildlife in water, sediment, whole fish, avian diet, avian eggs, and avian livers used in the following discussions. Concentrations of trace elements in samples matrices, other than water, in Table 2 and the following discussions are expressed on a dry weight basis unless otherwise noted. In some cases, distinction between concern and effect concentrations is arbitrary. A concern concentration designation was assigned to a value so noted in the literature or to a value associated with relatively minor effects (e.g., LC_1 or decreased growth rate for a limited time period). Effect concentrations were assigned to values so noted in the literature or to values causing substantial effects (e.g., LC_{50}, reduced survival or production, or teratogenic effects). No discussion of contaminant concentrations is presented where information was not found for specific elements or matrices.

Specific Conductance

Specific conductance is an indirect measure of TDS in water. Nevada Administrative Code (NAC) 445.125 classifies surface waters in Stillwater Marsh as Class D waters, which have no restrictions on dissolved solids concentrations. However, beneficial uses of Class D waters include protection of aquatic life and propagation of wildlife. During the study, specific conductance in the four wetland study units ranged from 1,140 to 17,460 rnicrosiemens per cm

Wide variations were found between seasons and among wetlands and years (Table 3). The highest values were found in East Alkali Lake and Swan Lake Check during the spring of 1991 (Table 3). Both units were nearly desiccated and evaporation is assumed to have concentrated dissolved solids. Specific conductance in Dry Lake was also highly variable between seasons and years. Water availability and delivery appears to be responsible. The highest specific conductance values in Dry Lake were found in 1992, the driest year of our study. Similarly, the lowest values were found in 1993, the wettest year. Records of water delivery to Dry Lake are only available for the 1992 water year; a total of 487 acre-feet of water was delivered between May and August.

Major elements, such as sodium and potassium, accounted for more than 99 percent of the dissolved solids in water of Lahontan Valley (Hoffman et al. 1990b). Trace elements represented less than 1 percent. Water released from Lahontan Reservoir was a dilute calcium-sodium-bicarbonate of sodium sulfate type water (Lico 1992). Water in wetlands on Stillwater NWR was most often of a slightly to moderately saline sodium-chloride type.

Significant mortality of mallard *(Anas platyrhynchos)* ducklings occurred when relegated to water with a specific conductance greater than 20,000 (Mitcham and Wobeser 1988).

Table I. Regulatory standards applicable to designated waters in Nevada. Standards are from Nevada Administrative Code (NAC) 445.1339.

Constituent	Municipal or Domestic Supply (µg/L)	Aquatic Life (µg/L)	Irrigation (µg/L)	Watering of Livestock (µg/L)
Arsenic	50	180[a,b]	100	200
Barium	1000			
Beryllium	0		100	
Boron		—[b]	750	5000
Cadmium	10	—[b,c]	10	
Chromium	50	—[a,b,c]	100	1000
Copper		—[b,c]	200	500
Iron		1000	5000	
Lead	50	—[b,c]	5000	100
Manganese			200	
Mercury	2	0.012		10
Molybdenum		19		
Nickel	13.4	—[b,c]	200	
Selenium	10	5		
Zinc		—[b,c]	2000	25000

[a] The arsenic standards for aquatic life are specific for As+3. The 96-hour average aquatic life standard is given.

[b] The standard applies to dissolved fraction only.

[c] Standards for aquatic life are based on water hardness, which is expressed as mg!L $CaC0_3$. Formulae for 96-hour average standards for specific elements are as follows:

Cadmium:	0.85exp[0.7852In(hardness)-3.490]
Chromium:	0.85exp[0.8190 In(hardness)+1.561]
Copper:	0.85exp[0.8545In(hardness)-1.465]
Lead:	0.25exp[I.273 In(hardness)-4.705]
Nickel:	0.85exp[0.8460 In(hardness)-1.l645]
Zinc:	0.85exp[0.8473In(hardness)+0.7614]

Birds given only water with a specific conductance between 7,500 to 20,000 suffered sublethal effects on growth, feathering, and other physiological functions. Weight gain of ducklings reared on water with a conductivity greater than 4,000 was depressed.

During our study, specific conductance did not exceed 20,000 (Table 3). Specific conductance exceeded 7,500 during one or more sampling periods in two of the drainwater units (East Alkali Lake and Swan Lake Check). Specific conductance exceeded 4,000 in all units during one or more sampling periods.

Table 2. Selected effect and concern concentrations for metals and trace elements. Concentrations, other than water are in dry weight unless otherwise noted.

Constituent	Category	Water µg/L	Concentration (µg/g) Sediment	Whole fish	Avian diet	Avian egg	Avian liver
Aluminum	Concern	87[a]					
	Effect	100b			S,OOO'		
Arsenic	Concern		33[d]	0.22"			
	Effect	40[f]	8S[d]	2.1g'	30[h]		4.5'
Boron	Concern	200[j]			30[k]	13[k]	17[k]
	Effect	S2200'			1000k	49[k]	SIk
Cadmium	Concern		Sd	O.OS"			
	Effect	l.om	9[d]		200[n]		
Chromium	Concern	21.5[o]	80[d]	4.0[p]	loq		4.0[p]
	Effect	190[o]	14S[d]	soq	soq		
Copper	Concern	3.4[o]	70[d]	0.9"			
	Effect	110[o]	390[d]				
Iron	Concern		21200'				
Lead	Concern	J.O'	3S[d]	0.22"	2S'		2°
	Effect	3.5[v]	1I0[d]		12S[w]		8°
Manganese	Concern	388[o]	460'				
Mercury	Concern		O.lSd	0.17"			
	Effect	O.lx	1.3[d]	0.62[y]•	0.5[z]	0.83[z]•	6.6[z]•
Molybdenum	Concern	28[o]			200"		
	Effect	790[o]			SOO"	16[ab]	
Nickel	Concern		30[d]				
	Effect		SOd				
Selenium	Concern	J.S"	1[ad]	4a,	3[af]	4[ag]	lOah
	Effect	3.0"	4[ad]	10"	Saf	loag	30[ah]
Vanadium	Concern	9.0[o]			100ai		6S7[al]
	Effect	170[o]					
Zinc	Concern		120[d]	34.2f'	178[aj]		
	Effect	32[ak]	270[d]		3000[al]		401[al]

• Wet weight basis.

Table 2. Continued.

---

[a] U.S. EPA (1988); recommendation that aluminum in water should not exceed more than once every 3 years.
[b] Hall et al. (1988); toxic to centrarchids at pH between 6.9 and 7.3.
[c] Sparling (1990); growth and survival of mallard *(Anas platyrhynchos)* and black ducks *(Anas rubripes)* was reduced at normal dietary concentrations of calcium and phosphorus.
[d] Long and Morgan (1991); concern concentration represents an Effect Range-Low (lower lOth percentile) and effect concentration represents an Effect Range-Median (median) of sediment-based bioassays causing effect.
[e] Schmitt and Brumbaugh (1990); 85th percentile of whole fish (wet weight) in the National Contaminant Monitoring Program.
[f] U.S. EPA (1985a); mortality and malformation of fish and amphibian embryos and larvae.
[g] Gilderhus (1966); decreased growth and survival of juvenile bluegill *(Lepomis macrochirus)*.
[b] Camardese et al. (1990); growth, development, and physiology of mallard ducklings affected.
Stanley et al. (1994); reduced growth of mallard ducklings.
[j] Birge and Black (1977); LC_1 of embryo-larval toxicity tests to goldfish *(Carassius auratus)*.
[k] Smith and Anders (1989); reduced weight gain of mallard ducklings through 21 days at 30 in diet; reduced body weight of hatchlings at 300 in diet (13 in eggs and 17 in juvenile liver); reduced hatching success, hatch weight, duckling survival, and duckling weight at 1000 in diet in egg and in juvenile liver).
Gersich (1984) 2I-day LC_{50} for *Daphnia magna*
[m] Hughes (1973); LC_{50} for striped bass *(Morone saxatilis)*.
[n] Cain et al. (1983); mild to severe kidney lesions to mallard duckling. White et al. (1978); slight to severe kidney lesions in adult mallards at a dietary concentration of 200 and slight to moderate gonad alteration at a dietary exposure of 20 µg/g.
[o] Birge et al. (1979a) 28-day LC_1 (concern concentration) and LCso(effect concentration) for embryo-larval toxicity tests to rainbow trout *(Salmo gairdneri)*.
[p] Eisler (1986); a concentration of 4 suggests chromium contamination.
[q] Haseltine et al. (1985); survival of ducklings suppressed.
[r] Persaud et al. (1993); lower effect level guideline (lower 5th percentile of sediment-based bioassays causing effect).
[s] U.S. EPA (1985b); reproduction of *Daphnia magna* impaired 10%.
[t] Finley et al. (1976); biochemical effects observed.
[u] Eisler (1988); concentration in avian liver exceeding 2 was elevated with 8 associated with lead poisoning.
[v] Wong et al. (1981); LC_{50} for rainbow trout.
[w] Hoffman et al. (1985); reduced growth and abnormal development of American kestrels *(Falco sparverius)*.
[x] Birge et al. (l979b); rainbow trout LC_{50} in 28-day flow-through bioassay.
[Y] Snarski and Olson (1982); reduced reproduction of fathead minnows *(Pimephales promelas)*. Kania and O'Hara (1974); mean whole-body concentration as low as 0.7 wet weight was associated with diminished predator-avoidance behavior in mosquitofish *(Gambusia ajfinis)*.
[z] Heinz (1979); diet (7% moisture) and liver tissue concentrations associated with reduced reproduction and duckling behavioral effects. Egg concentrations associated with reduced hatch rate and juvenile survival. Hen mallards, across generations had a mean of 1.3 (wet weight) in liver, whereas males had a mean of 4.4 Nicholas and Osborn (1984); nephrotoxic lesions in European starlings *(Sturnus vulgaris)* fed a diet containing 1.1 mercury; liver contained 6.55 (dry weight).
[aa] Eisler (1989); concern concentration included reduced growth; effect level included reproductive impairment.
[ab] Friberg et al. (1975); concentration was embryotoxic.

Table 2. Concluded.

[ac] Skorupa et al. (1996); validated lowest observable effect in water for fish and wildlife via bioaccumulation was 1.5 to 3.0 Lemly (1996); 2.0 on a total recoverable basis in filtered samples, hazardous to health and long-term survival of fish and wildlife populations.
[ad] Skorupa et al. (1996); 1 μg/g in sediment was the minimum concentration associated with effects on avian reproduction, whereas 3 μg/g in sediment was the minimum concentration associated with effects on fish; EC_{100} of>4.0 μg/g in sediment for fish and birds in a variety of freshwater systems. See also Lemly and Smith (1987).
[ae] Lillebo et al. (1988); concern and effect concentrations for selenium in whole fish. See also Lemly (1996) and Skorupa et al. (1996).
[af] Lemly and Smith (1987) and Lemly (1996) identify a concern level for avian diet OO μg/g. Skorupa and Ohlendorf (1991) and Skorupa et al. (1996) identified a critical dietary threshold of 5.0 μg/g.
[ag] Skorupa et al. (1996); suscptibility of captive mallard hatchlings to duck hepatitis virus was significantly increased at 4 μg/g, whereas the lowest observed adverse effect level for mallards was 10 μg/g. Skorupa and Ohlendorf (1991) identify a critical embryotoxic and teratogenic threshold between 13 and 24 μg/g in avian eggs.
[ah] Skorupa et al. (1992); baseline concentrations rarely exceed 10 μg/g in avian liver; high risk of adverse biological effect at 30 μg/g in avian liver. Heinz (1996); reproductive impairment is possible when the liver of egg laying females contain >3 μg/g (wet weight, or about 12 μg/g dry weight); important sublethal effects may occur when the liver of young or adults contain greater than 10 μg/g (wet weight, or about 40 μg/g dry weight).
[ai] White and Dieter (1978); altered lipid metabolism to adult mallards.
[aj] Stahl et al. (1978); immuno suppression in domestic chickens.
[ale] U.S. EPA (1987), LC,o to aquatic invertebrates.
[al] Gasaway and Buss (1972); reduced mallard survival.

Aluminum

Water

Nevada does not have water quality standards for aluminum. However, the U.S. EPA recommends that aluminum concentrations in water should not exceed 87 more than once every 3 years when the pH is between 6.5 and 9.0 (U.S. EPA 1988). Toxicity of aluminum varies considerably with chemical species and complexation (U.S. EPA 1988). Speciation is affected by several environmental parameters, particularly pH. Adverse impacts to aquatic species from elevated concentrations of aluminum in water have been demonstrated (Hall et al. 1988; Woodward et al. 1991; Farag et al. 1993; DeLonay et al. 1993). However, much of this research focused on the toxicity of aluminum as affected by acidification of aquatic habitats. Aluminum in water and sediments tends to be more toxic when mobilized by low pH (U.S. EPA 1988). In general, early life stages were increasingly sensitive to aluminum as pH decreased. Aluminum was toxic or caused significant sublethal effects at concentrations ≥50 when pH 6.0. Monomeric aluminum was toxic to centrarchids at concentrations between 100 and 150 at pH values of 6.9 to 7.3 (Hall et al. 1988). In our study, aluminum concentrations in

Table 3. Mean values for water quality parameters in four wetlands on Stillwater National Wildlife Refuge, 1990-93.

Wetland and Sampling Period	Air Temp. (°C)	Water Depth (em)	Water Temp. (°C)	pH	Specific Conductance ($\mu S/cm$)	Dissolved Oxygen *(mg/L)*
East Alkali Lake						
Spring 1990	28	37	25	9.4	8390	11.1
Fall 1990	19	33	17	8.8	4980	11.5
Spring 1991	24	13	25	9.5	17460	15.7
Fall 1991	Dry					
Spring 1992	Dry					
Fall 1992	Dry					
Spring 1993	Dry					
Fall 1993	Dry					
Swan Lake Check						
Spring 1990	29	27	28	8.8	3260	11.0
Fall 1990	19	16	18	8.9	2430	13.0
Spring 1991	29	7	30	9.1	16270	11.7
Fall 1991	Dry					
Spring 1992	Dry					
Fall 1992	Dry					
Spring 1993	Dry					
Fall 1993	Dry					
Lead Lake						
Spring 1990	19	57	20	9.2	5400	6.7
Fall 1990	Dry					
Spring 1991	Dry					
Fall 1991	Dry					
Spring 1992	Dry					
Fall 1992	Dry					
Spring 1993	Dry					
Fall 1993	27	13	20	8.3	1140	7.9

water collected ranged over two orders of magnitude, with the mean exceeding 10,000 (Table 4). All of the water samples exceeded the U.S. EPA recommendation of 87 Because of the high pH in the study area (Table 3), implications to aquatic life are

Table 4. Comparison of aluminum concentrations (dry weight, except water) in unfiltered water, sediment, and biological matrices collected from four wetlands on Stillwater National Wildlife Refuge, 1990-93. Numbers include mean, (number of samples'), and [sample range].

Sample Matrix	Total	Wetlands: Dry Lake	East Alkali Lake	SwanLake Check	Lead Lake
Water (µg/L)	10,280 (73) [120-34800]	12,000 (33) [2390-27300]	15,7400 (15) [3980-34800]	5,670 (15) [2100-14900]	3,340 (10) [120-8260]
Sediment (µg/g)	32,969 (74) [17100-57800]	29,693 (34) [17100-57800]	36,260 (15) [25900-54500]	33,780 (15) [21900-46900]	37,950 (10) [29900-42600]
Pondweed (µg/g)	7,290 (29) [1380-18900]	6,823 (24) [1380-18900]	9,528 (5) [4270-16700]		
Chironomid (µg/g)	13,797 (54) [3355-34700]	14,742 (21) [3355-34700]	14,694 (14) [7200-32500]	11,937 (15) [4500-18000]	12,675 (4) [11800-13800]
Corixid (µg/g)	3,957 (61) [13-16100]	4,225 (29) [769-10000]	3,608 (13) [35-9560]	4,997 (14) [642-16100]	404 (5) [13-1270]
Fish[b] (µg/g)	402 (19) [54-1490]	402 (19) [54-1490]			
Stilt egg (µg/g)	_'(411) [<2.0-2.5]				-' (4/1) [<2.0-2.5]
Coot egg (µg/g)	-' (28/1) [<2.0-10.3]	-' (6/1) [<2.0-10.3]	-' (10/0) [<2.0]	-' (2/0) [<2.0]	-' (10/0) [<2.0]
Stilt liver (µg/g)	-' (36/11) [<2.0-33.4]	-' (10/4) [<2.0-33.4]	-' (10/2) [<2.0-20.3]	-' (5/1) [<2.0-4.1]	-' (11/4) [<2.0-12.8]
Coot liver (µg/g)	6.9 (41/26) [<2.0-46.5]	10.7 (10/8) [<2.0-46.5]	4.2 (11/7) [<2.0-9.3]	5.5 (10/6) [<2.0-19.3]	7.3 (10/5) [<2.0-41.3]

• If one or more samples were below analytical detection limits, the second number in rarentheses is the number of samples which exceeded detection limits.
Fish include juvenile tui chub, pumpkinseed, and carp.
, More than 50% of the samples were below detection limits.

<u>Aquatic</u> <u>Vegetation</u> <u>and</u> <u>Invertebrates</u>

Avian species are adversely affected by excessive aluminum in diet. Growth and survival of mallard and black ducks *(Anas rubripes)* were affected by dietary aluminum, calcium, and phosphorus (Sparling 1990). A dietary concentration of 10,000 µg/g aluminum caused mortality at normal dietary levels of calcium and phosphorus. Growth was reduced and behavior was affected at 5,000 µg/g. Elevated concentrations of aluminum have been correlated with avian eggshell malfonnation (Nyholm 1981). Conversely, Miles et al. (1993) found that egg and bone strength was affected more by calcium and phosphorous than aluminum and suggested that reproductive effects were caused by reduced levels of calcium and phosphorus. No adverse effects were found at 5,000 µg/g aluminum. In our study, aluminum concentrations in pondweed, chironomids, and corixids ranged over two orders of magnitude, with over 55% of the pondweed samples exceeding 5,000 µg/g and 30% exceeding the 10,000 µg/g avian dietary effect concentration. Nearly all chironomid samples contained >5,000 µg/g aluminum and 72% had>10,000 µg/g, whereas only 23% of the corixid samples had >5,000 µg/g, with only 7% >10,000 µg/g.

Effect or concern concentrations for aluminum in whole fish were not located during a review of available literature. No fish samples exceeded the avian dietary effect concentration.

Avian Eggs and Liver Tissue

Aluminum was rarely detected in avian eggs and in slightly less than 50% of the avian juvenile liver samples. No infonnation was found on effect concentrations avian eggs and livers. Aluminum toxicosis in birds may be attributed to the fonnation of insoluble phosphates in the gastrointestinal tract and tissues and the interference of phosphate metabolism (Sparling 1990; Sparling 1995; Miles et at. 1993). In this case, aluminum in tissues may not be expected to be elevated.

Correlations. Trends. and Management Implications

Aluminum concentrations in water were not significantly correlated to specific conductance when wetlands were combined ($P = 0.261$; $r^2 = 0.018$; $n = 73$) or when Dry Lake was examined separately ($P = 0.361$; $r^2 = 0.027$; $n = 33$). Aluminum in water was not correlated to aluminum concentrations in other sample matrices when all units were considered or when Dry Lake was considered separately. Aluminum concentrations in sediment were only correlated with concentrations in chironomids when all units were considered ($P = 0.004$; $r^2 = 0.147$; $n = 54$) and when Dry Lake was considered separately ($P = 0.005$; $r^2 = 0.341$; $n = 21$).

Aluminum concentrations in all matrices varied considerably over the course of the study (Table 5); however, no trends were apparent in any sample matrices within study units. No consistent declines were found in aluminum concentrations water, sediment, pondweed, and corixids collected from Dry Lake over the course of the study. Aluminum concentrations in water decreased following short-tenn (6 month) desiccation of Dry Lake (Table 5). However, concentrations rebounded by the next sampling period. Concentrations in water from Lead Lake decreased following a longer period (3 years) of desiccation, but levels in sediment did not decrease.

Previous studies have found aluminum at concern concentrations throughout Lahontan Valley and background sites. Hoffinan et al. (1990) found that the mean concentration of aluminum in dipteran larvae at background sites was 11,000 µg/g ($n = 3$), whereas the mean concentration at sites affected by agricultural drainwater was 8,150 µg/g ($n = 9$). Similarly, the mean concentration in algae collected from agricultural drains the Carson Division of the Newlands Project was 9,993 µg/g ($n = 48$) whereas the mean concentration at sites not impacted

Table 5. Aluminum concentrations in unfiltered water, sediment, pondweed, and aquatic invertebrates collected from four wetlands on Stillwater National Wildlife Refuge, 1990-93.

Wetland and Sampling Period	Sample Matrix[a] Water (µg/L)	Sediment (µg/g)	Pondweed (µg/g)	Chironomid (µg/g)	Corixid (µg/g)
Lake					
Spring 1990	12,190 (4)[b] [2,850]	37,280 (5) [11,480]		29,240 (5) [4,375]	3,196 (5) [2334]
Fall 1990	15,250 (5) [6,610]	28,600 (5) [3,050]	4,495 (2) [265]	9,060 (1) [-]	3,650 (5) [662]
Fall 1991	3,080 (5) [680]	27,480 (5) [3,660]	5,785 (2) [3,445]	9,070 (3) [2,312]	9,120 (4) [710]
Spring 1992	16,810 (5) [4,790]	22,020 (5) [2,770]	6,501 (5) [3,273]	14,800 (5) [1,958]	4,530 (5) [2,184]
Fall 1992	12,450 (5) [4,380]	23,790 (5) [1,960]	5,181 (5) [2,126]	4,443 (5) [1,040]	2,907 (5) [1,254]
Spring 1993	10,770 (5) [3,610]	35,060 (5) [2,030]	11,026 (5) [6,021]	15,450 (2) [350]	2,926 (5) [1,745]
Fall 1993	14,440 (5) [3,430]	34,600 (4) [2,342]	5,932 (5) [4,642]		
East Alkali Lake					
Spring 1990	12,960 (5) [6,460]	41,420 (5) [7,830]		21,380 (5) [7,071]	77 (5) [35]
Fall 1990	25,360 (5) [8,070]	30,100 (5) [2,510]	9,528 (5) [4,678]	9,778 (4) [908]	2,755 (4) [1,016]
Spring 1991	8,910 (5) [1,470]	37,260 (5) [3,210]		11,940 (5) [3,090]	8,875 (4) [738]
Swan Lake Check					
Spring 1990	9,390 (5) [2,990]	40,040 (5) [5,546]		15,120 (5) [1,707]	2,396 (5) [1,240]
Fall 1990	4,400 (5) [1,610]	25,320 (5) [2,892]		7,974 (5) [2,101]	2,100(5) [696]
Spring 1991	3,210 (5) [630]	35,980 (5) [1,347]		12,718 (5) [4,466]	11,868 (4) [4,360]
Lead Lake					
Spring 1990	6,480 (5) [1,270]	40,680 (5) [2.737]		12,675 (4) [719]	404 (5) [468]
Fall 1993	196 (5) [41]	35,220 (5) [3,730]			

[a] Concentrations are on a dry weight basis, except water.
[b] Numbers include mean, (number of samples), and [standard deviation].

by drainwater was 8,873 (*n* = 8) (Rowe et al. 1991). This suggests that aluminum on Stillwater NWR is not attributable to agricultural drainage.

At pH greater than 6.0, aluminum solubility is low and most precipitates onto sediment or substrates (Sparling 1995). Comparisons of dissolved aluminum (12 versus total recoverable aluminum (3,600 in water collected in Lahontan Valley suggest that most aluminum is associated with particulate material in water (Hoffman et al. 1990b). Therefore, high pH (8.3 to 9.7) in wetlands on Stillwater NWR may be protective of aluminum toxicity to aquatic organisms. However, aluminum in potential avian dietary items remains a concern. It is not certain if aluminum was incorporated in potential avian dietary items or precipitates adhered to surfaces. Regardless, ingestion of plants or invertebrates would result in potentially hazardous exposure to aluminum. Aluminum toxicosis may partially account for unexplained aquatic bird mortality that has occurred on Stillwater NWR in the past. More information is needed to more fully assess the implications of elevated aluminum to fish and wildlife on Stillwater NWR.

Arsenic

Water

The current Nevada standard for arsenic for the protection of aquatic life is based on species of arsenic. The chronic (96-hour) aquatic life standard for arsenic +3 is 180 Our study only determined total arsenic and did not differentiate between arsenic species. Based on Eh and pH values, Lico (1992) suggested that dissolved arsenic in wetlands of Stillwater NWR would exist in an arsenic +3 state. Regulatory standards prior to 1991 were based on total arsenic and were set at 40 Arsenic has been found to be acutely toxic to aquatic invertebrates at concentrations as low as 810 and to juvenile fish at concentrations as low as 490 (U.S. EPA 1985a). Other studies have found lower levels of mortality (LC_{1}) and malformation of fish and amphibian embryos and larvae at concentrations as low as 40 (U.S. EPA 1985a). Over 93% of the water samples in our study exceeded the 40 water quality standard and aquatic life effect level (Table 6). Twelve percent of the water samples exceeded 180 All samples were below acutely toxic levels.

Sediment

Based on a review of sediment-based bioassays, Long and Morgan (1991) identified a lower biological effect concentration (ER-L; lower 10th percentile of tests with effects) of 33 arsenic in coastal marine and estuarine sediment. A similar review identified a lowest effect concentration guideline (5th percentile of effects) of 6 in aquatic sediment (Persaud et al. 1993). In our study, arsenic concentrations in sediment ranged from 10 to 30 (Table 6) with a mean of 19 No sediment sample exceeded 33 but all samples exceeded 6

Aquatic vegetation and Invertebrates

Mortality (LC_{50}) of mallards occurred at dietary arsenic concentrations of 1,000 after 6 days and 500 after 32 days (National Academy of Sciences 1977). Growth, development, and physiology of mallard ducklings maintained on a diet containing 30 arsenic or greater were adversely affected (Camardese et al. 1990). Growth in salmonids maintained on a diet containing 30 arsenic was reduced (Oladimeji et al. 1984). In our study, 28% of the vegetation samples met or exceeded the 30 avian and fish dietary effect concentration. No

Table 6. Comparison of arsenic concentrations (dry weight, except water) in unfiltered water, sediment, and biological matrices collected from four wetlands on Stillwater National Wildlife Refuge, 1990-93. Numbers include mean, (number of samples'), and [sample range].

Sample Matrix	Total	Wetlands: Dry Lake	East Alkali Lake	SwanLake Check	Lead Lake
Water (µg/L)	117 (74) [34-270]	96 (34) [38-166]	203 (15) [131-270]	108(15) [55-210]	69 (10) [34-104]
Sediment (µg/g)	18.9 (74) [10.1-30.3]	16.3 (34) [10.1-22.2]	16.4(15) [13.0-19.0]	22.2 (15) [20.0-24.9]	26.2 (10) [23.0-30.3]
Pondweed (µg/g)	21.9 (29) [4.6-47.0]	18.8 (24) [4.6-42.0]	36.6 (5) [30.0-47.0]		
Chironomid (µg/g)	14.1 (55) [5.8-24.6]	15.4 (21) [7.6-24.3]	14.2 (15) [7.9-24.6]	13.8 (15) [8.3-24.1]	7.6 (4) [5.8-8.9]
Corixid (µg/g)	7.0 (61) [0.6-20.3]	8.1 (29) [2.1-18.8]	8.1 (13) [0.6-20.3]	5.5 (14) [1.3-16.0]	2.3 (5) [1.4-3.8]
Fish[b] (µg/g)	1.4 (19/16) [<0.5-2.8]	1.4 (19/16) [<0.5-2.8]			
Stilt egg (µg/g)	_[c] (4/0) [<0.3]				_[c] (4/0) [<0.3]
Coot egg (µg/g)	0.4 (28/15) [<0.3-1.4]	0.8 (6/5) [<0.3-1.4]	0.4 (10/9) [<0.3-0.6]	_[c] (2/0) [<0.3]	_[c] (10/1) [<0.3-0.3]
Stilt liver (µg/g)	_[c] (36/1) [<OJ-OJ]	_[c] (10/0) [<OJ]	_[c] (10/1) [<OJ-OJ]	_[c] (5/0) [<OJ]	_[c] (11/0) [<OJ]
Coot liver (µg/g)	0.7 (41/29) [<0.3-3.3]	0.4 (10/7) [<0.3-0.9]	1.4 (II) [0.5-3.3]	0.6 (10/9) [<003-1.0]	_[c] (10/2) [<0.3-0.4]

• If one or more samples were below analytical detection limits, the second number in rarentheses is the number of samples which exceeded detection limits.
Fish include juvenile tui chub, pumpkinseed, and carp.
·c More than 50% of the samples were below detection limits.

pondweed samples contained concentrations associated with lethality. All aquatic invertebrate samples contained concentrations below fish and avian dietary effect levels.

An arsenic concentration of 103 (wet weight) in muscle tissue was associated with decreased growth and survival of bluegill *(Lepomis macrochirus)* juveniles (Gilderhus 1966). Muscle concentrations were approximately 60% of whole fish concentrations. Therefore, a whole fish concentration of 2.1 μg/g (wet weight) could be considered an effect level. Conversion to dry weight (wet weight multiplied by a conversion factor of 3.6; Hoffinan et al. 1990b) yields an effect concentration of 7.1 μg/g. Arsenic concern levels for whole fish (exceeding the 85th percentile of concentrations in a national contaminants monitoring program) ranged from 0.22 to 0.38 μg/g, wet weight (Schmitt and Brumbaugh 1990). Conversion to dry weight yields concern concentrations ranging from 0.79 to 1.37 μg/g. In our study, 21% of the whole fish samples exceeded the 7.1 μg/g effect concentration; however, 68% of the samples exceeded the 0.22 μg/g concern level for whole fish. No whole fish samples exceeded the avian dietary effect concentration.

Avian Eggs and Liver

Arsenic was below detection limits in more than half of the avian eggs. An arsenic concentration 00.6 μg/g in mallard eggs was not associated with decreased hatching success or teratogenesis (Stanley et al. 1994). The highest concentration in our study was below this level. A liver concentration of 4.5 μg/g was associated with reduced growth of mallards (Stanley et al. 1994). Concentrations in all liver samples from Stillwater NWR were below this level.

Antagonistic interactions occurred between arsenic and selenium in waterfowl diets (Stanley et al. 1994). Arsenic alleviated the effect of selenium to hatching success and embryo deformation of mallards. Arsenic also appeared to reduce selenium accumulation in livers and eggs. However, the threshold concentration for such protective effects appeared to occur between 100 and 400 μg/g arsenic in diet. Therefore, arsenic in concentrations found in our study may not afford protection to selenium effects in migratory birds in Lahontan Valley.

Correlations. Trends. and Management Implications

Arsenic concentrations in water were correlated to specific conductance when all units were combined ($P < 0.001$; $r^2 = 0.608$; $n = 73$) and when Dry Lake was considered separately ($P < 0.001$; $r^2 = 0.465$; $n = 33$). When all units were considered, arsenic concentrations in water were correlated to concentrations in corixids, but r^2 was small ($P = 0.003$; $r^2 = 0.143$; $n = 59$). When Dry Lake was considered separately, arsenic concentrations in water were correlated to concentrations in sediment ($P = 0.005$; $r^2 = 0.229$; $n = 33$), chironomids ($P = 0.002$; $r^2 = 0.429$; $n = 19$), and fish ($P = 0.033$; $r^2 = 0.254$; $n = 18$). Arsenic concentrations in sediment were weakly correlated to concentrations in corixids when all units were combined ($P = 0.008$; $r^2 = 0.118$; $n = 61$) and when Dry Lake was considered separately ($P = 0.006$; $r^2 = 0.250$; $n = 29$).

Arsenic in water, sediment, and aquatic invertebrates collected from Dry Lake did not markedly decline during the study (Table 7). Lack of decline is likely attributable to the relationship of arsenic and specific conductance. Water availability during the study did not enable consistent maintenance of low levels of specific conductance. Many arsenic compounds are volatile (Moore et al. 1990). However, arsenic concentrations in sediment, chironomids, and corixids increased or did not significantly decrease following a period of short-term desiccation. Similarly, arsenic concentrations in sediment from Lead Lake did not decrease over a longer period of desiccation.

These fmdings suggest that reductions of arsenic concentrations in water, sediment, and aquatic invertebrates may be achieved through management of specific conductance. Wetland desiccation does not appear to reduce concentrations. During all sampling periods, mean arsenic concentrations in water in Dry Lake, East Alkali Lake, and Swan Lake Check exceeded Nevada arsenic standards for protection of aquatic life and aquatic life concern levels. The lowest

Table 7. Arsenic concentrations in unfiltered water, sediment, pondweed, and aquatic invertebrates collected from four wetlands on Stillwater National Wildlife Refuge, 1990-93.

Wetland and Sampling Period	Sample Matrix' Water (µg/L)	Sediment (µg/g)	Pondweed (µg/g)	Chironomid (µg/g)	Corixid (µg/g)
Dry Lake					
Spring 1990	113.3 (4)[b] [3.6]	20.9 (5) [Ll]		16.8 (5) [3.2]	2.7 (5) [0.6]
Fall 1990	68.0 (5) [1.7]	16.0 (5) [0.6]	41.5 (2) [0.5]	19.0 (1) [-]	7.7 (5) [304]
Fall 1991	8Ll (5) [4.6]	15.1 (5) [004]	21.7 (2) [11.0]	16.5 (3) [104]	13.5 (4) [1.2]
Spring 1992	74.0 (5) [104]	12.6 (5) [1.6]	18.9 (5) [504]	1804 (5) [4.0]	7.8 (5) [2.0]
Fall 1992	157.3 (5) [5.1]	18.2 (5) [1.6]	19.4 (5) [2.2]	9.5 (5) [1.7]	8.7 (5) [5.5]
Spring 1993	43.0 (5) [2.6]	15.6 (5) [1.0]	10.5 (5) [304]	15.5 (2) [3.5]	9.0 (5) [5.0]
Fall 1993	132.2 (5) [ILl]	15.8 (4) [1.8]	16.3 (5) [9.8]		
East Alkali Lake					
Spring 1990	14404 (5) [1004]	16.3 (5) [0.9]		10.2 (5) [104]	0.9 (5) [0.2]
Fall 1990	21404 (5) [15.1]	14.8 (5) [1.0]	36.6 (5) [5.9]	11.2 (5) [104]	10.8 (4) [5.6]
Spring 1991	262.5 (5) [8.3]	18.0 (5) [0.9]		21.3 (5) [3.8]	1404 (4) [2.5]
Swan Lake Check					
Spring 1990	64.8 (5) [6.0]	23.1 (5) [1.4]		9.7 (5) [1.3]	1.8 (5) [0.5]
Fall 1990	74.6 (5) [2.7]	21.4 (5) [1.0]		14.2 (5) [5.1]	3.3 (5) [1.0]
Spring 1991	184.0 (5) [32.6]	22.2 (5) [1.2]		17.4(5) [4.0]	12.8 (4) [3.6]
Lead Lake					
Spring 1990	100.2 (5) [5.2]	2704 (5) [2.4]		7.6 (4) [1.1]	2.3 (5) [0.9]
Fall 1993	38.2 (5) [2.3]	25.0 (5) [104]			

• Concentrations are on a dry weight basis, except water.
b Numbers include mean, (number of samples), and [standard deviation].

observed specific conductance value (1,150 corresponded to an arsenic concentration of *43* μg/L (Figure 4). Therefore, to attain water quality standards, a specific conductance below 1,150 may be needed. Arsenic dissolution from sediments also may occur. In this event, arsenic concentrations in sediments may decline if specific conductance levels remain reduced for extended periods. If dissolution occurs, further reductions in water concentrations may accompany reductions in sediment concentrations.

All exceedances of the avian dietary effect level for arsenic occurred in pondweed collected from Dry Lake. Arsenic in pondweed was not significantly correlated to specific conductance, water, or sediment. Therefore, the possibility of reducing arsenic concentrations in pondweed is uncertain.

Boron

Water

Prior to 1996, the Nevada water quality standard for protection of aquatic life for boron was 550 μg/L (NAC 445.1339). All water samples collected during our study exceeded this standard (Table 8). Invertebrate *(Daphnia magna)* mortality (21-day LC_{50}) has been found from 52,200 to 53,200 μg/L boron as boric acid, with a maximum allowable toxicant concentration (MATC) between 6,000 and 13,000 μg/L (Lewis and Valentine 1981; Gersich 1984). The MATCs were determined by sublethal effects to growth and production. Boron compounds were typically more toxic to fish and amphibians in hard water (200 ppm $CaCO_3$) than in soft water (Birge and Black 1977). Water in Lahontan Valley wetlands typically exceeds 200 ppm $CaCO_3$ (Hoffman et al. 1990b). In hard water, significant mortality (7-day LC_{50}) of cyprinid eggs and larvae occurred at concentrations of 59,000 and 79,000 μg/L boron as boric acid and borax, respectively (Birge and Black 1977). Boron species was not determined in our study. The 7-day LC_1 values for cyprinid eggs and larvae were 200 and 900 μg/L boron as borax and boric acid, respectively. In our study, 25% of the water samples exceeded 6,000 μg/L. Concentrations in water samples collected from Stillwater NWR did not approach those causing substantial mortality *(LC_{50})* to vertebrate eggs and larvae. However, concentrations in all samples exceeded a concern concentration (LC_1).

Aquatic Vegetation and Invertebrates

Hatching success of mallards maintained on a diet supplemented with 1,000 μg/g boron was reduced by almost 50% (Smith and Anders 1989). Body weights of hatchlings of adults maintained on diets supplemented with 300 and 1,000 μg/g boron were also lower. Weight gain of ducklings maintained on diets supplemented with 30, 300, and 1,000 μg/g boron were depressed through 21 days. Productivity (number of ducklings through 21-days) of females maintained on a diet containing 1,000 μg/g boron was reduced. Delayed growth rate and biochemical effects were found in female mallards maintained on diets containing 100 and 400 μg/g (Hoffman et al. 1990a). In our study, all aquatic vegetation samples and 77% of the invertebrate samples exceeded 30 μg/g. Seventy-two percent of the vegetation samples exceeded *300* μg/g and 17% exceeded 1000 μg/g.

Adverse effects of boron were increased in the presence of selenium in diet (Hoffman et al. 1991). These effects became more pronounced when dietary protein was reduced. At a restricted dietary protein level (7%), increased mortality was found in mallard ducklings

Table 8. Comparison of boron concentrations (dry weight, except water) in unfiltered water, sediment, and biological matrices collected from four wetlands on Stillwater National Wildlife Refuge, 1990-93. Numbers include mean, (number of samples"), and [sample range].

Sample Matrix	Wetlands: Total	Dry Lake	East Alkali Lake	SwanLake Check	Lead Lake
Water (µg/L)	4,830 (74) [680-21600]	2,930 (34) [1000-6670]	6,705 (15) [3220-11300]	8,350 (15) [2440-21600]	3,040 (10) [680-5520]
Sediment (µg/g)	107 (74) [21-246]	97 (34) [21-184]	97 (15) [21-174]	147 (15) [70-246]	96 (10) [51-159]
Pondweed (µg/g)	554 (29) [180-1288]	574 (24) [180-1288]	462 (5) [247-566]		
Chironomid (µg/g)	117 (54) [51-266]	107 (21) [66-175]	145 (14) [64-266]	112 (15) [51-230]	84 (4) [79-89]
Corixid (µg/g)	61 (61) [4-200]	59 (29) [9-188]	75 (13) [4-200]	65 (14) [13-170]	22 (5) [7-55]
Fish[b] (µg/g)	22 (19) [2-67]	22 (19) [2-67]			
Stilt egg (µg/g)	2.2 (4) [1.2-2.7]				2.2 (4) [1.2-2.7]
Coot egg (µg/g)	6.4 (28) [2.9-16.2]	10.1 (6) [7.2-16.2]	6.2 (10) [3.3-11.6]	6.2 (2) [5.6-6.8]	4.5 (10) [2.9-7.1]
Stilt liver (µg/g)	1.5 (36/29) [<0.5-6.8]	0.7 (10/6) [<0.5-2.3]	2.3 (10/9) [<0.5-6.8]	1.9 (5) [1.0-5.0]	1.4 (11/9) [<0.5-2.6]
Coot liver (µg/g)	5.1 (41) [0.3-18.9]	5.1 (10) [3.4-6.1]	6.0 (II) [0.3-11.7]	5.6 (10) [0.7-18.9]	3.4 (10) [1.9-5.9]

" If one or more samples were below analytical detection limits, the second number in rarentheses is the number of samples which exceeded detection limits.
Fish include juvenile tui chub, pumpkinseed, and carp.

maintained on a diet supplemented with 15 ppm Se and 1000 ppm B. Selenium concentrations did not approach 15 ppm in our study.

Boron effect or concern concentrations for whole fish were not found in a literature review. However, 21% of the whole fish samples exceeded the 30 µg/g avian dietary effect level.

Avian Eggs and Liver Tissue

A boron concentration of 13 in mallard eggs was associated with reduced hatch weight and 49 was associated with reduced hatch rate and juvenile survival (Smith and Anders 1989). In our study, one coot egg exceeded 13 boron and no eggs approached 49 Seventeen boron in mallard liver was associated with reduced growth and production (Smith and Anders 1989). In our study, one coot liver sample contained >17

Correlations. Trends. and Management Imolications

Boron concentrations in water were significantly correlated to specific conductance when all units were considered ($P < 0.001$; $= 0.800$, $n = 73$) and when Dry Lake was considered separately ($P < 0.001$; $r^2 = 0.932$, $n = 34$). When all units were considered, boron concentrations in water were correlated with concentrations in pondweed ($P < 0.001$; $r^2 = 0.416$, $n = 23$), chironomids ($P < 0.001$; $r^2 = 0.396$, $n = 52$), and corixids ($P < 0.001$; $r^2 = 0.416$, $n = 59$). When Dry Lake was considered separately, boron in water was correlated with boron in sediment ($P < 0.022$; $r^2 = 0.154$, $n = 34$), pondweed ($P < 0.001$; $r^2 = 0.362$, $n = 24$), chironomids ($P < 0.001$; $r^2 = 0.478$, $n = 21$), corixids ($P < 0.001$; $r^2 = 0.659$, $n = 29$), and fish ($P = 0.006$; $r^2 = 0.320$, $n = 19$). In Dry Lake only, boron concentrations in chironomids were correlated to boron concentrations in sediment ($P = 0.002$; $r^2 = 0.372$; $n = 21$).

Boron in sediment in all units declined considerably from the initial sampling period (Table 9). Concentrations in water, aquatic vegetation, and invertebrates increased or did not markedly decline over the course of the study. Again, because of the relationship between specific conductance and boron, changes in concentrations in water, aquatic vegetation, and invertebrates is likely attributable to the quality of available water during each of the sampling periods. Boron concentrations in all sample matrices increased or did not markedly decline following a period of short-term desiccation of Dry Lake. Boron concentrations in sediment from Lead Lake declined considerably between the initial and final sampling periods. However, sediment concentrations in all units also declined from the initial sampling period. Therefore, it is not certain whether declines in Lead Lake resulted from desiccation or some other factor.

In our study, all water samples exceeded the pre-1996 Nevada boron standard for protection of aquatic life (550 and an aquatic life concern concentration (200 Birge and Black 1977). In Dry Lake, the lowest mean boron concentration in water (1,160 Table 9) corresponded to a specific conductance of 1,150 (Table 3). Therefore, attainment of the pre-I996 Nevada boron standard through management of specific conductance in wetlands appears unlikely. Exceedances of an avian dietary boron concentration associated with reduced reproduction and duckling survival (1,000 Smith and Anders 1989) occurred in aquatic vegetation collected from Dry Lake during the fall 1992 sampling period. Specific conductance during this sampling period approached 6,500 Exceedances of an avian dietary boron concentration associated with reduced weight of hatchlings and reduced weight gain of ducklings (300 Smith and Anders 1989) occurred in aquatic vegetation. Specific conductance associated with these exceedances ranged from 1,150 to 6,500 These results suggest that management of specific conductance may enable reduction of potentially lethal levels of boron in avian diet, but may not eliminate potential adverse effects to avian species. Wetland desiccation does not appear to be an effective tool in controlling boron in Lahontan Valley wetlands.

Table 9. Boron concentrations in unfiltered water, sediment, pondweed, and aquatic invertebrates collected from four wetlands on Stillwater National Wildlife Refuge, 1990-93.

Wetland and Sampling Period	Sample Matrix[a]				
	Water (µg/L)	Sediment (µg/g)	Pondweed (µg/g)	Chironomid (µg/g)	Corixid (µg/g)
Dry Lake					
Spring 1990	2,890 (4)[b] [190]	160.0 (5) [22.8]		103.3 (5) [7.2]	17.3 (5) [6.6]
Fall 1990	1,900 (5) [0]	86.8 (5) [8.9]	207.0 (2) [27.0]	110.0 (1) [-]	44.0 (5) [15.7]
Fall 1991	4,010 (5) [235]	90.0 (5) [13.4]	192.5 (2) [8.5]	85.1 (3) [13.9]	92.4 (4) [7.5]
Spring 1992	2,500 (5) [194]	55.4 (5) [23.0]	274.8 (5) [73.6]	95.2 (5) [7.7]	47.2 (5) [6.9]
Fall 1992	6,020 (5) [69]	115.8 (5) [10.7]	1,125.0 (5) [128.1]	142.6 (5) [18.8]	127.4 (5) [35.0]
Spring 1993	1,160 (5) [120]	89.2 (5) [14.2]	385.2 (5) [135.9]	91.5 (2) [5.5]	30.6 (5) [15.1]
Fall 1993	1,910 (5) [73]	79.5 (4) [17.5]	807.8 (5) [222.3]		
East Alkali Lake					
Spring 1990	6,940 (5) [335]	148.6 (5) [30.1]		153.8 (5) [29.2]	4.6 (5) [0.8]
Fall 1990	3,390 (5) [91]	73.2 (5) [8.1]	461.6 (5) [111.6]	81.3 (4) [10.3]	61.3 (4) [8.7]
Spring 1991	9,790 (5) [1,214]	69.8 (5) [42.3]		187.2 (5) [49.9]	175.0 (4) [21.8]
Swan Lake Check					
Spring 1990	2,740 (5) [230]	218.2 (5) [29.4]		95.5 (5) [7.2]	24.9 (5) [14.4]
Fall 1990	2,560 (5) [40]	98.6 (5) [10.5]		65.2 (5) [14.5]	30.2 (5) [10.2]
Spring 1991	19,760 (5) [2,890]	124.0 (5) [32.0]		174.0 (5) [40.3]	60.0 (4) [12.3]
Lead Lake					
Spring 1990	5,220 (5) [160]	133.2 (5) [17.5]		83.9 (4) [3.5]	22.2 (5) [17.4]
Fall 1993	860 (5) [92]	58.8 (5) [4.6]			

[a] Concentrations are on a dry weight basis, except water.
[b] Numbers include mean, (number of samples), and [standard deviation].

Cadmium

Water

The Nevada water quality standard for protection of aquatic life for cadmium is based on water harness (NAC 445.1339). Hoffman et al. (1990b) found a median hardness of 440 mg/L as $CaC0_3$ for wetlands in Lahontan Valley. This equates to a 96-hour average standard of 3.09 µg/L. In our study, cadmium was below detection limits (ranging from 0.5 to 2.0 in 75% of the water samples (Table 10). The cadmium concentration in one of 74 water samples exceeded 3.09

Toxicity of cadmium tends to decrease with increasing water hardness (Eisler 1985). The 96-hour LC_{50} for *Daphnia magna* in moderately hard water (51 mg/L $CaC0_3$) was 9.9 (U.S. EPA 1980a), whereas the 21-day LC_{50} of 5.0 (Biesinger and Christensen 1972). At a hardness of 70 mg/L $CaC0_3$, an LC_{50} of 1.0 was found for striped bass *(Morone saxatilis;* Hughes 1973, as cited in Eisler 1985). In our study, 10% of the water samples met or exceeded the 1.0 effect concentration. However, detection limits exceeded the effect concentration in some cases.

Sediment

Long and Morgan (1991) identified a lower biological effect concentration (ER-L; lower 10th percentile) of 5.0 cadmium in coastal marine and estuarine sediments. No sediment samples in our study exceeded this concentration.

Aquatic Vegetation and Inyertebrates

No mortality occurred in mallards maintained on a diet containing 200 cadmium (White and Finley 1978). Sublethal effects, including histopathological effects, occurred in mallards maintained on a diet containing 20 cadmium (Cain et al. 1983). In our study, cadmium concentrations in aquatic vegetation and invertebrates were well below the 20 concern concentration.

Cadmium concern concentrations for whole fish (exceeding the 85th percentile of concentrations in a national contaminants monitoring program) ranged from 0.05 to 0.11 wet weight (Schmitt and Brumbaugh 1990). Conversion to dry weight yields concern levels ranging from 0.18 to 0.40 In our study, 47% of the whole fish samples contained > 0.18 µg/g.

Correlations. Trends. and Management Implications

Cadmium concentrations in water were below analytical detection limits in > 50% of the water samples overall and from Dry Lake; therefore, statistical examination was not attempted. Detected cadmium concentrations in water did not correspond to elevated specific conductance (Tables 3 and 11). When all units were considered, specific conductance was significantly correlated to cadmium concentrations in chironomids ($P < 0.001$, $r^2 = 0.277$, $n = 48$). When Dry Lake was considered, specific conductance was correlated to cadmium concentrations in sediment ($P = 0.002$, $r^2 = 0.288$, $n = 32$), aquatic vegetation ($P = 0.002$, $r^2 = 0.378$, $n = 23$), and chironomids ($P = 0.017$, $r^2 = 0.291$, $n = 19$). Cadmium in sediment was correlated to cadmium in pondweed ($P < 0.001$, $r^2 = 0.468$, $n = 23$) and chironomids ($P = 0.008$, $r^2 = 0.347$, $n = 19$).

Table 10. Comparison of cadmium concentrations (dry weight, except water) in unfiltered water, sediment, and biological matrices collected from four wetlands on Stillwater National Wildlife Refuge, 1990-93. Numbers include mean, (number of samples'), and [sample range].

Sample Matrix	Total	Wetlands: Dry Lake	East Alkali Lake	Swan Lake Check	Lead Lake
Water (µg/L)	_[b] (74/17) <0.5-11.0]	_[b] (34/16) [<0.5-5.0]	_[b] (15/0) [<2.0]	-b(15/1) [<0.8-11.0]	_[b] (10/0) [<1.0]
Sediment (µg/g)	0.5 (74/41) [<0.2-1.7]	0.6 (34/22) [<0.2-1.5]	0.3 (15/8) [<0.2-0.4]	_[b] (15/6) [<0.2-0.5]	0.9 (10/5) [<0.5-1.7]
Pondweed (µg/g)	0.5 (29/20) [<0.2-1.2]	0.6 (24119) [<0.2-1.2]	_[b] (5/1) [<0.2-0.4]		
Chironomid (µg/g)	1.0 (54/52) [<0.2-2.6]	1.6 (21/20) [<1.0-2.6]	0.6 (14) [0.2-1.1]	0.8 (15) [0.3-1.3]	0.2 (4/3) [<0.2-0.3]
Corixid (µg/g)	1.0 (61/56) [<0.1-3.6]	1.2 (29) [0.5-3.6]	1.3 (13) [0.4-3.3]	0.6 (14113) [<0.1-0.9]	_[b] (5/1) [<0.2-0.2]
Fish[c] (µg/g)	0.3 (19/14) [<0.1-0.6]	0.3 (19/14) [<0.1-0.6]			
Stilt egg (µg/g)	_[b] (4/0) [<0.1]				_[b] (4/0) [<0.1]
Coot egg (µg/g)	_[b] (28/0) [<0.1]	_[b] (6/0) [<0.1]	_[b] (10/0) [<0.1]	_[b] (2/0) [<0.1]	_[b] (10/0) [<0.1]
Stilt liver (µg/g)	0.3 (36/25) [<0.1-4.0]	0.3 (10) [0.2-0.5]	0.3 (10/5) [<0.1-0.8]	_[b] (5/2) [<0.1-0.1]	0.5 (11/8) [<0.1-4.0]
Coot liver (µg/g)	0.2 (41/27) [<0.1-0.7]	0.4 (10) [0.2-0.7]	0.2 (11/9) [<0.1-0.4]	0.2 (10/6) [<0.1-0.3]	_[b] (10/2) [<0.1-0.2]

• If one or more samples were below analytical detection limits, the second number in parentheses is the number of samples which exceeded detection limits.
More than 50% of the samples were below detection limits.
c Fish include juvenile tui chub, pumpkinseed, and carp.

Cadmium concentrations in sediment from Dry Lake increased during our study (Table 11). Changes in other matrices were generally within seasonal sample variability. Concentrations in all matrices did not decline following short-term desiccation of Dry Lake.

These results suggest that limited control of cadmium in some components of Lahontan Valley wetlands may be achieved through management of specific conductance. Wetland desiccation does not appear to be an effective tool in the control of this element.

Table 11. Cadmium concentrations in unfiltered water, sediment, pondweed, and aquatic invertebrates collected from four wetlands on Stillwater National Wildlife Refuge, 1990-93.

Wetland and Sampling Period	Sample Matrix[a] Water (µg/L)	Sediment (µg/g)	Pondweed (µg/g)	Chironomid (µg/g)	Corixid (µg/g)
<u>Dry Lake</u>					
Spring 1990	-[b] (4/0)[c] [-]	0.76 (5/4) [0.30]		1.07 (5/5) [0.06]	0.96 (5/5) [0.05]
Fall 1990	-[b] (5/0) [-]	-[b] (5/1) [-]	0.50 (2/2) [0.10]	-[b] (1/0) [-]	0.60 (5/5) [0.11]
Fall1991	-[b] (5/1) [-]	0.25 [0. 2]	0.80 (2/2) [0.21]	2.02 (3/3) [0.13]	1.63 (4/4) [0.26]
Spring 1992	1.1 (5/4) [0.5]	0.33 (5/4) [0.16]	0.81 (5/5) [0.32]	2.50 (5/5) [0.08]	1.13 (5/5) [0.26]
Fall1992	-[b] (5/1) [-]	-[b] (5/0) [-]	-[b] (5/0) [-]	0.70 (5/5) [0.25]	1.01 (5/5) [0.17]
Spring 1993	0.5 (5/5) [0.2]	1.42 [0. 2]	0.80 (5/5) [0.16]	2.25 (2/2) [0.05]	2.06 (5/5) [1.00]
Fall 1993	0.8 (5/5) [0.4]	1.38 (4/4) [0.11]	0.48 (5/5) [0.24]		
<u>East Alkali Lake</u>					
Spring 1990	-[b] (5/0) [-]	-[b] (5/0) [-]		0.78 (5/5) [0.33]	2.63 (5/5) [0.60]
Fall 1990	-[b] (5/0) [-]	0.28 (5/4) [0.10]	-[b] (5/1) [-]	0.53 (4/4) [0.15]	0.48 (4/4) [0.08]
Spring 1991	-[b] (5/0) [-]	0.27 (5/4) [0.09]		0.41 (5/5) [0.03]	0.46 (4/4) [0.05]
<u>Swan Lake Check</u>					
Spring 1990	-[b] (5/0) [-]	-[b] (5/0) [-]		0.69 (5/5) [0.20]	0.76 (5/5) [0.07]
Fall1990	-[b] (5/1) [-]	-[b] (5/1) [-]		1.04 (5/5) [0.18]	0.48 (5/5) [0.12]
Spring 1991	-[b] (5/0) [-]	0.38 (5/5) [0.12]		0.53 (5/5) [0.12]	0.49 (4/3) [0.31]
<u>Lead Lake</u>					
Spring 1990	-[b] (5/0) [-]	-[b] (5/0) [-]		0.21 (4/3) [0.09]	-[b] (5/1) [-]
Fall1993	-[b] (5/0) [-]	1.60 (5/5) [0.06]			

[a] Concentrations are on a ary weight basis, except for water.
[b] More than 50% of the samples were below detection limits.
[c] Numbers include arithmetic mean, (number of samples/number of samples with detectable quantities), and [standard deviation].

Chromium

Water

Nevada water quality standards for the protection of aquatic life for chromium are based on chemical species (NAC 445.1339). The 96-hour average standard for hexavalent chromium (chromium +6) is 10 Trivalent chromium (chromium +3) standards are based on hardness. Hardness of water for wetlands in Lahontan Valley (Hoffman et al. 1990b) equates to a chromium +3 96-hour average standard of 697 In our study, chromium species was not determined. Total chromium concentrations in water were well below the chromium +3 criterion; however, 37% of the water samples exceeded the chromium +6 criterion (Table 12).

Chromium +6 is generally the more toxic of the two species of chromium (Eisler 1986). Toxicity (LCsos) of chromium +6 ranged from 435 for *Daphnia magna* to > 100,000 for several fish species (Eisler 1986). Toxicity of chromium +3 ranged from 2,000 for *Daphnia magna* to >50,000 for several fish species. Toxicity of both chromium +6 and chromium +3 decreased at high pH or high water hardness. Low mortality (LC$_t$) and teratogenic effects to egg and larval fish occurred at concentrations of 21.5 (Birge et al. 1979a). All water samples collected during our study were well below acutely toxic concentrations, but a limited number of samples approached concern levels.

Sediment

Long and Morgan 1991) identified a lower effect concentration (ER-L; lower 10th percentile) of 80 chromium in coastal marine and estuarine sediments. In our study, concentrations in all sediment samples were well below this concern level.

Aquatic Vegetation and Inyertebrates

Increased mortality occurred in female black ducks and hatchlings maintained on diets containing 50 chromium +3 (Haseltine et al. 1985). Growth patterns were altered in treated groups, but weights in all groups were similar at 10-weeks of age. Fecundity, egg survival, and embryo development were not affected. Sublethal effects, including histopathology, were found in black ducks maintained on a diet containing 10 chromium. In our study, chromium concentrations in 24% of the pondweed samples exceeded 10 and 7% exceeded 50 Approximately 3I% of the chironomid samples and only one of the corixid samples exceeded 10 µg/g.

Tissue residues in fish and wildlife in excess of 4.0 may indicate chromium contamination (Eisler 1986), although the significance of this concentration is uncertain. In our study, 58% of the whole fish samples exceeded 4.0 Three whole fish samples exceeded a 10 avian dietary concern concentration, and one exceeded a 50 avian dietary effect concentration (Haseltine et al. 1985).

Avian Eggs and Liyer Tissue

No information on concern or effect concentrations of chromium to avian eggs was found. The concentration of chromium in black duck livers was independent of dietary concentration (ducks fed diets containing 0,10, and 50 chromium; Haseltine et al. 1985). No liver samples in our study exceeded a 4.0 concern level (Eisler 1986).

Table 12. Comparison of chromium concentrations (dry weight, except water) in unfiltered water, sediment, and biological matrices collected from four wetlands on Stillwater National Wildlife Refuge, 1990-93. Numbers include mean, (number of samples'), and [sample range].

Sample Matrix	Total	Wetlands			
		Dry Lake	East Alkali Lake	SwanLake Check	Lead Lake
Water (Jlg/L)	8 (74/45) [<5-20]	7 (34/27) [<5-18]	10 (1518) [<8-20]	_b (1515) [<8-19]	8 (1015) [<10-12]
Sediment (Jlg/g)	18 (74) [5-32]	17 (34) [9-32]	17 (15) [5-31]	19(15) [12-31]	24 (10) [19-27]
Pondweed (Jlg/g)	II (29) [1-67]	12 (24) [1-67]	6 (5) [3-10]		
Chironomid (Jlg/g)	9 (54) [3-19]	10 (21) [5-19]	9 (14) [5-14]	7 (15) [3-13]	7 (4) [7-8]
Corixid (Jlg/g)	3 (61/52) [<1-11]	4 (29/28) [<2-9]	3 (13/10) [<1-6]	3 (14/9) [<1-11]	2 (5) [1-3]
Fish[c] (Jlg/g)	7 (19/17) [<I-52]	7 (19/17) [<1-52]			
Stilt egg (Jlg/g)	1.1 (4) [0.9-1.2]				1.1 (4) [0.9-1.2]
Coot egg (Jlg/g)	1.1 (28) [0.8-1.6]	1.2 (6) [1.0-1.4]	1.1 (10) [0.8-1.6]	1.0 (2) [1.0]	1.0 (10) [0.8-1.4]
Stilt liver (Jlg/g)	1.4 (36) [0.7-2.4]	1.4 (10) [1.0-2.4]	1.3 (10) [1.0-2.0]	1.9 (5) [1.5-2.4]	1.2 (11) [0.7-1.9]
Coot liver (Jlg/g)	1.3 (41) [0.9-2.0]	1.5 (10) [1.2-2.0]	1.3 (11) [1.1-1.6]	1.2 (10) [0.9-1.5]	1.3 (10) [1.1-1.7]

• If one or more samples were below analytical detection limits, the second number in parentheses is the number of samples which exceeded detection limits.
More than 50% of the samples were below detection limits.
c Fish include juvenile tui chub, pumpkinseed, and carp.

Correlations. Trends. and Management Implications

Specific conductance was not correlated to chromium concentrations in water when all units were considered or when Dry Lake was considered separately. Chromium concentrations in water were correlated to chromium concentrations in sediment when all units were considered (P = 0.001) and when Dry Lake was considered separately (P = 0.015); however, r^2 was small in both cases (r^2 = 0.142 and 0.181, respectively). Chromium concentrations in water were not correlated to concentrations in other sample matrices. Chromium concentrations in sediment were correlated only to concentrations in chironomids when all units were considered (P < 0.001, r^2 = 0.298, n = 54) and when Dry Lake was considered separately (P = 0.006, r^2 = 0.369, n = 21).

Chromium concentrations in sediment from Dry Lake declined slightly during our study (Table 13). Changes in concentrations in other matrices were generally within the range of normal sample variability. Concentrations in all matrices did not decrease following short-term desiccation of Dry Lake. Concentrations in water and sediment in Lead Lake declined following a longer period of desiccation.

These results suggest that management of specific conductance may be used to control chromium levels in chironomids. The possibility of using wetland desiccation as a control strategy should be further assessed.

Copper

Water

The Nevada water quality standard for copper for the protection of aquatic life is based on water hardness (NAC 445.1339). The hardness of water for wetlands in Lahontan Valley (Hoffman et al. I990b) equates to a 96-hour average copper standard of 41.9 Jlg/L. In 96-hour toxicity tests, copper was toxic (LC_{S0}) to rainbow trout *(Sa/rna gairdneri)* eggs and juveniles at a concentration near 400 (Giles and Klaverkamp 1982). In 28-day embryo-larval toxicity tests using rainbow trout, Birge et al. (1979a) identified an LC_{I} of 3.4 and an LC_{S0} of 110 Jlg/L. In our study, copper was below detection limits (ranging from 3 to 8 Jlg/L) in 25% of the water samples. In those samples with detectable copper concentrations, 91% exceeded the 3.4 concern concentration. Copper concentrations in water did not exceed Nevada water quality standards or the 110 effect concentration (Table 14).

Sediment

Long and Morgan (1991) identified a lower effect concentration of 70 for copper in coastal marine and estuarine sediments. Copper concentrations in sediment found in our study did not exceed this concern concentration.

Copper concern concentrations for whole fish ranged from 0.9 to 1.1 wet weight (Schmitt and Brumbaugh 1990). Conversion to dry weight yields concern concentrations of 3.2 to 4.0 Jlg/g. In our study, copper concentrations in 79% of the whole fish samples exceeded the lower concern level.

Correlations, Trends. and Management Implications

Specific conductance was not correlated to copper concentrations in water when all units were considered or Dry Lake was considered separately. Copper concentrations in water were correlated only to concentrations in corixids when Dry Lake was considered (P = 0.008, r^2 = 0.251, n = 27). Copper concentrations in sediment were not correlated to other sample matrices.

Copper concentrations in all sample matrices did not decline during our study (Table IS). Similarly, concentrations in all sample matrices were generally within the range of normal sample variability following short-term desiccation of Dry Lake. Concentrations increased slightly in sediment following a longer period of desiccation of Lead Lake.

Table 13. Chromiwn concentrations in unfiltered water, sediment, pondweed, and aquatic invertebrates collected from four wetlands on Stillwater National Wildlife Refuge, 1990-93.

Wetland and Sampling Period	Sample Matrix[a]				
	Water (µg/L)	Sediment (µg/g)	Pondweed (µg/g)	chironomid (µg/g)	Corixid (µg/g)
Dry Lake					
Spring 1990	13.0 (4/4)[b] [2.0]	27.4 (5/5) [3.5]		15.4 (5/5) [2.4]	3.1 (5/5) [1.1]
Fall 1990	_c (5/1) [-]	15.0 (5/5) [1.1]	3.3 (2/2) [0.3]	6.0 (1/1) [-]	2.0 (5/4) [0.6]
Fall 1991	5.7 (5/4) [1.9]	17.7 (5/5) [2.2]	7.2 (2/2) [3.6]	7.3 (3/3) [1.1]	7.8 (4/4) [0.4]
Spring 1992	12.0 (5/4) [3.9]	15.0 (5/5) [1.3]	37.4 (5/5) [22.1]	11.7 (5/5) [0.9]	4.6 (5/5) [2.2]
Fall 1992	8.5 (5/4) [3.5]	15.5 (5/5) [1.7]	7.2 (5/5) [2.9]	5.5 (5/5) [0.8]	4.2 (5/5) [0.7]
Spring 1993	2.4 (5/5) [1.4]	16.6 (5/5) [3.1]	6.0 (5/5) [3.5]	10.1 (2/2) [1.9]	2.2 (5/5) [1.0]
Fall 1993	4.8 (5/5) [2.3]	13.2 (4/4) [3.7]	2.5 (5/5) [1.5]		
East Alkali Lake					
Spring 1990	10.4 (5/3) [5.4]	26.1 (5/5) [4.1]		12.9 (5/5) [1.2]	1.3 (5/4) [0.4]
Fall 1990	12.0 (5/3) [6.8]	15.0 (5/5) [1.3]	5.6 (5/5) [2.6]	5.8 (4/4) [0.6]	1.3 (4/2) [0.8]
Spring 1991	_c (5/2) [-]	10.3 (5/5) [4.2]		6.9 (5/5) [0.8]	5.5 (4/4) [0.4]
Swan Lake Check					
Spring 1990	13.6 (5/5) [2.8]	26.8 (5/5) [3.8]		9.5 (5/5) [1.9]	2.8 (5/5) [0.4]
Fall 1990	_c (5/0) [-]	14.0 (5/5) [1.4]		4.6 (5/5) [0.9]	_c (5/1) [-]
Spring 1991	_c (5/0) [-]	15.0 (5/5) [1.1]		8.3 (5/5) [0.8]	6.4 (4/3) [4.1]
Lead Lake					
Spring 1990	11.0 (5/5) [0.9]	26.3 (5/5) [0.6]		7.0 (4/4) [0.5]	1.8 (5/5) [0.7]
Fall 1993	_c (5/0) [-]	21.2 (5/5) [1.3]			

• Concentrations are on a dry weight basis except for water.
b Nwnbers include aritlunetic mean, er of samples/nwnber of samples with detectable quantities), and [standard deviation].
c More than 50% of the samples were below detection limits.

Table 14. Comparison of copper concentrations (dry weight, except water) in unfiltered water, sediment, and biological matrices collected from four wetlands on Stillwater National Wildlife Refuge, 1990-93. Numbers include mean, (number of samples[a]), and [sample range].

Sample Matrix	Total	Wetlands: Dry Lake	East Alkali Lake	Swan Lake Check	Lead Lake
Water (µg/L)	12 (74/55) [<2-40]	15 (34/32) [<5-33]	19 (15/12) [<3-40]	_[b] (15/5) [<3-7]	3 (10/6) [<2-8]
Sediment (µg/g)	40 (74) [29-62]	40 (34) [32-50]	38 (15) [29-49]	35 (15) [31-44]	52 (10) [43-62]
Pondweed (µg/g)	22 (29) [8-53]	23 (24) [8-53]	19 (5) [10-28]		
Chironomid (µg/g)	30 (54) [14-49]	35 (21) [14-49]	29 (14) [20-42]	26 (15) [18-33]	26 (4) [26-27]
Corixid (µg/g)	27 (61) [13-45]	31 (29) [24-45]	24 (13) [15-36]	23 (14) [14-29]	19 (5) [13-25]
Fish[c] (µg/g)	5 (19) [1-9]	5 (19) [1-9]			
Stilt egg (µg/g)	5 (4) [4-6]				5 (4) [4-6]
Coot egg (µg/g)	7 (28) [1-77]	4 (6) [4-5]	4 (10) [1-4]	5 (2) [5]	11 (10) [3-77]
Stilt liver (µg/g)	20 (36) [14-36]	20 (10) [15-24]	21 (10) [17-36]	18 (5) [16-20]	19 (11) [14-32]
Coot liver (µg/g)	105 (41) [4-227]	99 (10) [31-202]	48 (11) [4-84]	127 (10) [87-168]	151 (10) [71-227]

[a] If one or more samples were below analytical detection limits, the second number in parentheses is the number of samples which exceeded detection limits.
[b] More than 50% of the samples were below detection limits.
[c] Fish include juvenile tui chub, pumpkinseed, and carp.

These results suggest that limited success in the control of copper in wetlands may be achieved through management of specific conductance. However, implications to copper concentrations in whole fish are uncertain. Wetland desiccation does not appear to be an effective tool for controlling copper in Lahontan Valley wetlands.

Table 15. Copper concentrations in unfiltered water, sediment, pondweed, and aquatic invertebrates collected from four wetlands on Stillwater National Wildlife Refuge, 1990-93.

Wetland and Sampling Period	Matrix·				
	Water (µg/L)	Sediment (µg/g)	Pondweed (µg/g)	Chironomid (µg/g)	Corixid (µg/g)
Dry Lake					
Spring 1990	14.8 (4/4)b [5.4]	41.6 (SIS) [7.0]		36.4 (SIS) [3.6]	26.8 (SIS) [1.6]
Fall 1990	20.2 (SIS) [5.5]	38.4 (5/5) [1.7]	21.2 (2/2) [1.5]	32.0 (1/1) [-]	26.2 (5/5) [1.2]
Fall 1991	6.9 (5/4) [2.5]	40.5 (SIS) [3.4]	29.8 (2/2) [12.6]	26.1 (3/3) [8.6]	42.4 (4/4) [2.5]
Spring 1992	23.8 (5/4) [5.6]	36.0 (SIS) [3.0]	32.6 (SIS) [10.5]	47.6 (SIS) [1.2]	30.4 (5/5) [2.3]
Fall 1992	18.5 (SIS) [7.3]	34.1 (SIS) [1.0]	12.4 (SIS) [3.3]	26.4 (SIS) [6.1]	26.9 (5/5) [2.3]
Spring 1993	7.6 (SIS) [1.6]	44.6 (5/5) [2.8]	21.4 (SIS) [6.7]	39.0 (2/2) [1.0]	35.7 (SIS) [5.5]
Fall 1993	13.0 (SIS) [4.5]	44.0 (4/4) [3.5]	22.3 (SIS) [15.9]		
East Alkali Lake					
Spring 1990	12.0 (5/3) [9.2]	39.2 (SIS) [7.2]		35.9 (SIS) [5.0]	31.1 (SIS) [3.5]
Fall 1990	32.0 (SIS) [6.9]	37.3 (SIS) [1.9]	19.4 (5/5) [6.7]	21.5 (4/4) [1.5]	16.3 (4/4) [1.3]
Spring 1991	10.8 (5/4) [5.8]	36.2 (5/5) [1.0]		26.8 (SIS) [2.0]	24.0 (4/4) [1.2]
Swan Lake Check					
Spring 1990	-[c] (5/1) [-]	37.8 (5/5) [4.8]		29.7 (SIS) [2.3]	25.9 (SIS) [1.0]
Fall 1990	4.4 (5/4) [1.8]	33.0 (5/5) [1.7]		22.8 (SIS) [2.6]	19.2 (5/5) [2.7]
Spring 1991	_[c] (5/0) [-]	35.2 (5/5) [1.3]		25.8 (SIS) [3.1]	25.0 (4/4) [2.9]
Lead Lake					
Spring 1990	_[c] (5/2) [-]	44.6 (5/5) [1.5]		26.0 (4/4) [0.5]	18.7 (5/5) [4.2]
Fall 1993	2.2 (5/4) [0.7]	58.8 (5/5) [2.3]			

• Concentrations are on a dry weight basis, except for water.

b Numbers include aritlunetic mean, (number of sampleslnumber of samples with detectable quantities), and [standard deviation].

C More than 50% of the samples were below detection limits.

Iron

Water

The Nevada water quality standard for protection of life for iron is 1,000 (NAC 445.1339). Over 93% of the water samples exceeded the Nevada aquatic life water quality standard (Table 16).

Sediment

Persaud et al. (1993) identified a lowest effect concentration guideline of 21,200 and a severe effect concentration guideline of 43,766 iron. Iron concentrations in sediment did not exceed the severe effect level guideline, but 97% exceeded the lowest effect level guideline.

Correlations. Trends, and Management Implications

Specific conductance was not correlated to iron concentrations in water when all units were considered or Dry lake was considered separately. Iron concentrations in water and sediment were not correlated to each other or other sample matrices.

Iron concentrations in all sample matrices did not decline during our study and were generally within the range of normal sample variability following short-term desiccation of Dry Lake (Table 17). These results indicate that management of specific conductance may be used with limited success to control iron in sediment. Wetland desiccation does not appear to be an effective method to control iron in water or sediment.

Lead

Water

The Nevada water quality standard for protection of aquatic life for lead is based on water hardness (NAC 445.1339). The hardness of water for wetlands in Lahontan Valley (Hoffman et al. 1990b) equates to a 96-hour average lead standard of 5.2 In our study, lead concentrations in 55% of the water samples were below detection limits (Table 18); however, detection limits ranged from 0.5 to 40 All detected concentrations exceeded the Nevada water quality standard. Reproduction of *Daphnia magna* was impaired 10% at a concentration of I and 50% at a concentration of 10 (U.S. EPA 1985b). At higher hardness (151 mg/L $CaCO_3$), lead is toxic to *Daphnia magna* (LC,o) at a concentration of 1,910 Toxicity (LC,o) has been documented in cyprinids at a concentration of 379 (Eisler 1988). However, sublethal effects in cyprinids, such as reduced growth, histopathology (including gonads), and spinal deformation, have been found at concentrations as low as 127 In our study, lead concentrations in water did not reach potentially toxic levels. However, all detected concentrations exceeded invertebrate reproduction effect levels.

Sediment

Long and Morgan 1991) identified a lower biological effect concentration of 35.0 μg/g lead in coastal marine and estuarine sediments. In our study, 19% of the sediment samples exceeded the lower biological effect concentration. No samples exceeded a median effect level of 110 (Long and Morgan 1991).

Table 16. Comparison of iron concentrations (dry weight, except water) in unfiltered water, sediment, and biological matrices collected from four wetlands on Stillwater National Wildlife Refuge, 1990-93. Numbers include mean, (number of samples), and [sample range].

			Wetlands		
Sample Matrix	Total	Dry Lake	East Alkali Lake	Swan Lake Check	Lead Lake
Water (µg/L)	9,290 (74) [120-33200]	10,530 (34) [1410-26300]	14,890 (15) [3490-33200]	5,600 (15) [1800-14300]	2,880 (10) [120-7120]
Sediment (µg/g)	28,486 (74) [20800-38300]	28,108 (34) [21400-33600]	29,353 (15) [20800-35200]	6,127 (15) [20900-30900]	32,010 (10) [25600-38300]
Pondweed (µg/g)	7,213 (29) [1410-16700]	6,810 (24) [1410-16700]	9,150(5) [4240-15600]		
Chironomid (µg/g)	11,643 (54) [5580-18100]	12,700 (21) [5926-18000]	12,349 (14) [8810-18100]	10,409 (15) [5580-16500]	8,253 (4) [7430-9240]
Corixid (µg/g)	3,961 (61) [44-14600]	4,582 (29) [740-11300]	3,432 (13) [139-9200]	4,449 (14) [449-14600]	372 (5) [44-1090]
Fish" (µg/g)	468 (19) [98-2231]	468 (19) [98-2231]			
Stilt egg (µg/g)	125 (4) [109-148]				125 (4) [109-148]
Coot egg (µg/g)	169 (28) [67-1640]	133 (6) [97-174]	111 (10) [67-137]	121 (2) [105-136]	259 (10) [78-1640]
Stilt liver (µg/g)	1,210 (36) [338-2770]	1,247 (10) [537-2770]	1,370 (10) [953-2060]	896 (5) [464-1370]	1,174 (11) [338-1770]
Coot liver (µg/g)	2,421 (41) [45-6840]	2,579 (10) [1300-4020]	2,880 (11) [45-5200]	2,235 (10) [629-6840]	1,944 (10) [100-2800]

" Fish include juvenile tui chub, pumpkinseed, and carp.

Aquatic Vegetation and Invertebrates

Death or pathology did not occur in mallards maintained on a diet containing 25 lead for 12 weeks, although some biochemical effects were found (Finley et al. 1976). Similar results were found in nestling American kestrels *(Falco sparverius)* administered 25 lead for 10 days (Hoffman et al. 1985). Significant sublethal effects, such as reduced growth and abnormal skeletal development were found at 125 in the diet of kestrels, whereas significant mortality (40%) was found at 625 µg/g in the diet. In our study, 11% of the chironomid samples exceeded the 25 dietary level associated with biochemical changes. No samples exceeded the dietary concentrations causing significant sublethal effects or mortality.

Table 17. Iron concentrations in unfiltered water, sediment, pondweed, and aquatic invertebrates collected from four wetlands on Stillwater National Wildlife Refuge, 1990-93.

Wetland and Sampling Period	Sample Matrix[a]				
	Water (µg/L)	Sediment (µg/g)	Pondweed (µg/g)	Chironomid (µg/g)	Corixid (µg/g)
Dry Lake					
Spring 1990	12,830 (4)[b] [2,200]	26,620 (5) [2,573]		16,460 (5) [1,414]	1,844 (5) [1,257]
Fall 1990	14,860 (5) [6,200]	29,260 (5) [1,795]	4,620 (2) [340]	9,790(1) [-]	4,206 (5) [738]
Fall 1991	2,250 (5) [700]	29,000 (5) [2,853]	6,545 (2) [3,995]	10,427 (3) [1,771]	10,240 (4) [922]
Spring 1992	16,120 (5) [4,200]	24,920 (5) [2,642]	7,628 (5) [4,048]	15,960 (5) [1,100]	5,282 (5) [2,673]
Fall 1992	11,540 (5) [4,300]	25,954 (5) [1,228]	5,512 (5) [2,253]	7,225 (5) [1,021]	4,574 (5) [1,923]
Spring 1993	7,790 (5) [2,300]	30,900 (5) [1,885]	9,600 (5) [5,234]	13,700 (2) [200]	2,476 (5) [1,465]
Fall 1993	9,910 (5) [2,400]	30,600 (4) [2,054]	5,480 (5) [3,990]		
East Alkali Lake					
Spring 1990	11,360 (5) [5,500]	30,260 (5) [5,295]		14,260 (5) [2,727]	162 (5) [19]
Fall 1990	24,360 (5) [7,500]	28,760 (5) [1,525]	9,150 (5) [4,179]	10,570 (4) [1,019]	3,188 (4) [1,129]
Spring 1991	7,470 (5) [1,900]	29,040 (5) [338]		11,862 (5) [2,126]	7,765 (4) [1,156]
Swan Lake Check					
Spring 1990	9,630 (5) [2,900]	26,500 (5) [3,949]		9,580 (5) [1,188]	1,470 (5) [692]
Fall 1990	4,440 (5) [1,600]	24,560 (5) [1,863]		9,408 (5) [2,448]	2,400 (5) [773]
Spring 1991	2,730 (5) [600]	27,320 (5) [1,070]		12,240 (5) [3,604]	10,733 (4) [3,896]
Lead Lake					
Spring 1990	5,590 (5) [900]	27,920 (5) [1,405]		8,253 (4) [699]	372 (5) [388]
Fall 1993	166 (5) [26]	36,100 (5) [1,890]			

[a] Concentrations are on a dry weight basis, except for water.
[b] Numbers include arithmetic mean, (number of samples), and [standard deviation].

Table 18. Comparison of lead concentrations (dry weight, except water) in unfiltered water, sediment, and biological matrices collected from four wetlands on Stillwater National Wildlife Refuge, 1990-93. Numbers include mean, (number of samples[a]), and [sample range].

Sample Matrix	Total	Wetlands: Dry Lake	East Alkali Lake	Swan Lake Check	Lead Lake
Water (µg/L)	[b] (74/34) [<5-53]	16 (34/23) [<5-53]	[b] (15/6) [<7-20]	[b] (15/5) [<7-20]	[b] (10/0) [<40]
Sediment (µg/g)	24 (74) [8-61]	23 (34) [9-52]	18 (15) [9-27]	19 (15) [8-28]	41 (10) [24-61]
Pondweed (µg/g)	9 (29/26) [<1-23]	10 (24/21) [<1-23]	8 (5) [6-10]		
Chironomid (µg/g)	13 (54/44) [<5-39]	19 (21/20) [5-39]	10 (14/10) [<5-20]	8 (15/10) [<5-19]	5 (4) [3-6]
Corixid (µg/g)	[b] (61/26) [<3-23]	7 (29/19) [<3-23]	[b] (1314) [<3-10]	[b] (14/3) [<3-17]	[b] (5/0) [<3]
Fish[c] (µg/g)	3 (19/17) [<0.4-9]	3 (19/17) [<0.4-9]			
Stilt egg (µg/g)	[b] (4/0) [<1.5]				[b] (4/0) [<1.5]
Coot egg (µg/g)	[b] (28/0) [<1.5]	[b] (6/0) [<1.5]	[b] (10/0) [<1.5]	[b] (2/0) [<1.5]	[b] (10/0) [<1.5]
Stilt liver (µg/g)	[b] (36/2) [<1.5-3.1]	[b] (10/0) [<1.5]	[b] (10/1) [<1.5-2.4]	[b] (5/0) [<1.5]	[b] (11/1) [<1.5-3.1]
Coot liver (µg/g)	[b] (41/2) [<1.5-2.0]	[b] (10/0) [<1.5]	[b] (11/0) [<1.5]	[b] (10/0) [<1.5]	[b] (10/2) [<1.5-2.0]

[a] If one or more samples were below analytical detection limits, the second number in parentheses is the number of samples which exceeded detection limits.
[b] More than 50% of the samples were below detection limits.
[c] Fish include juvenile tui chub, pumpkinseed, and carp.

Lead concern concentrations for whole fish ranged from 0.22 to 0.44 wet weight (Schmitt and Brumbaugh 1990). Conversion to dry weight yields concern concentrations ranging from 0.79 to 1.44 In our study, lead concentrations in 74% of the whole fish samples exceeded the lower whole fish concern level. Lead concentrations in whole fish samples did not exceed avian dietary concern concentrations.

Avian Eggs and Liver Tissue

Lead concentrations in avian liver exceeding 2 µg/g were considered elevated, with poisoning associated with concentrations exceeding 8 µg/g (Eisler 1988). In our study, lead was below detection limits in all avian eggs and was detected in only 4 of 77 avian liver tissue samples. Concentrations in two samples exceeded 2.0 µg/g.

Correlations. Trends. and Management Implications

Concentrations of lead in water were not correlated to specific conductance when all units were considered or when Dry Lake was considered separately. Similarly, lead concentrations in water were not correlated to lead in sediment or biological matrices, nor were concentrations in sediment correlated to concentrations in biological matrices. Lead concentrations in sediment from Dry Lake increased during our study (Table 19). Concentrations in other matrices were highly variable, and no trends were apparent. Concentrations in all matrices increased following short-term desiccation of Dry Lake. Lead concentrations also increased in Lead Lake sediments following a longer period of desiccation.

These results do not indicate that management of specific conductance or wetland desiccation could be effectively used to control lead in Lahontan Valley wetlands.

Mercury

Water

The Nevada water quality standard for protection of aquatic life for mercury (96-hour average) is 0.012 µg/L (NAC 445.1339). The detection limit for mercury during our sampling was considerably greater than this standard. Mercury was detected in 20 percent of the water samples. However, detection limits varied from 0.1 to 1.1 µg/L. The highest detected concentration was 0.6 µg/L (Table 20). All samples with detected concentrations of mercury exceeded the Nevada aquatic life standards.

Mercury in water has been found to be toxic (LC_{50}) to aquatic invertebrates at concentrations as low as 2.0 µg/L for inorganic mercury or as low as 0.9 µg/L for organic forms (U.S. EPA 1980b). Mercury has been shown to be toxic to salmonids at concentrations below 0.1 µg/L in 28-day flow through tests (Birge et al. 1979b). In similar tests of shorter duration (Birge et al. 1979b), non-salmonid fish were more tolerant of mercury in water, with LC_{50} values ranging from 0.3 to 5.3 µg/L. Growth of larval fathead minnows *(Pimephales promelas)* was reduced at a mercury concentration in water of 0.5 µg/L (Snarski and Olson 1982). In our study, mercury concentrations in 19% of the water samples 0.3 µg/L. The concentration in one water sample exceeded 0.5 µg/L.

Sediment

Long and Morgan (1991) identified a lower effect concentration of 0.15 µg/g and a median effect concentration of 1.3 µg/g for mercury in coastal marine and estuarine sediments. In our study, 87% of the sediment samples exceeded 0.15 µg/g. All samples collected from Lead Lake exceeded 1.3 µg/g.

Table 19. Lead concentrations in unfiltered water, sediment, pondweed, and aquatic invertebrates collected from four wetlands on Stillwater National Wildlife Refuge, 1990-93.

Wetland and Sampling Period	Matrix[a]				
	Water (µg/L)	Sediment (µg/g)	Pondweed (µg/g)	Chironomid (µg/g)	Corixid (µg/g)
Dry Lake					
Spring 1990	_[b] (4/0)[c] [-]	19.9 (5/5) [4.4]		11.4 (5/5) [4.3]	_[b] (5/0) [-]
Fall 1990	_[b] (5/0) [-]	9.4 (5/5) [0.5]	_[b] (2/0) [-]	_[b] (1/0) [-]	_[b] (5/0) [-]
Fall 1991	10.3 (5/4) [4.9]	15.1 (5/5) [2.2]	13.6 (2/2) [9.3]	22.4 (3/3) [2.8]	21.3 (4/4) [1.3]
Spring 1992	26.2 (5/4) [1.7]	12.0 (5/5) [2.5]	14.9 (5/5) [3.9]	36.4 (5/5) [1.4]	10.7 (5/5) [5.9]
Fall 1992	9.4 (5/5) [2.3]	17.3 (5/5) [1.9]	3.1 (5/4) [2.3]	6.3 (5/5) [0.6]	2.3 (5/5) [1.2]
Spring 1993	15.4 (5/5) [6.4]	45.2 (5/5) [2.6]	14.0 (5/5) [6.7]	20.0 (2/2) [1.0]	3.3 (5/5) [2.6]
Fall 1993	24.6 (5/5) [4.7]	45.8 (4/4) [3.7]	8.3 (5/5) [5.7]		
East Alkali Lake					
Spring 1990	_[b] (5/1) [-]	24.3 (5/5) [2.1]		11.8 (5/5) [4.3]	_[b] (5/0) [-]
Fall 1990	_[b] (5/0) [-]	9.0 (5/5) [0.0]	7.6 (5/5) [1.4]	_[b] (4/0) [-]	_[b] (4/0) [-]
Spring 1991	11.2 (5/5) [5.4]	19.6 (5/5) [7.9]		15.0 (5/5) [2.6]	9.1 (4/4) [0.6]
Swan Lake Check					
Spring 1990	_[b] (5/2) [-]	22.4 (5/5) [7.7]		6.8 (5/5) [3.0]	_[b] (5/0) [-]
Fall 1990	_[b] (5/0) [-]	8.4 (5/5) [0.5]		_[b] (5/0) [-]	_[b] (5/0) [-]
Spring 1991	14.0 (5/3) [7.3]	25.4 (5/5) [1.4]		15.0 (5/5) [3.3]	9.4 (4/3) [6.9]
Lead Lake					
Spring 1990	_[b] (5/0) [-]	25.3 (5/5) [1.5]		5.1 (4/4) [1.2]	_[b] (5/0) [-]
Fall 1993	_[b] (5/0) [-]	57.4 (5/5) [4.1]			

a Concentrations are on a dry weight basis, except for water.
b More than 50% of the samples were below detection limits.
c Numbers include arithmetic mean, (number of samples/number of samples with detectable quantities), and [standard deviation].

Table 20. Comparison of mercury concentrations (dry weight, except water) in unfiltered water, sediment, and biological matrices collected from four wetlands on Stillwater National Wildlife Refuge, 1990-93. Numbers include mean, (number of samples'), and [sample range].

Sample Matrix	Total	Wetlands Dry Lake	East Alkali Lake	SwanLake Check	Lead Lake
Water (µg/L)	_b (55/11) [<0.2-0.6]	_b (30/6) [<0.3-0.4]	_b(lO/I) [<0.3-0.6]	_b *(1014)* [<0.3-0.4]	_b (5/0) [<0.2]
Sediment (µg/g)	1.3 (54) [0.1-11.2]	0.3 (29) [0.1-0.5]	0.2 (10) [0.1-0.2]	0.5 (10) [0.5-0.6]	10.7 (5) [10.0-11.2]
Pondweed (µg/g)	0.2 (29/26) [<0.2-0.6]	0.3 (24121) [<0.2-0.6]	0.2 (5) [0.1-0.2]		
Chironomid (µg/g)	0.5 (36) [0.3-1.1]	0.6 (16) [0.4-1.1]	0.4 (10) [0.3-0.5]	0.6 (10) [0.4-0.9]	
Corixid (µg/g)	0.5 (41) [0.2-1.8]	0.6 (24) [0.2-1.8]	0.2 (8) [0.2-0.4]	0.7 (9) [0.6-0.9]	
Fish[c] (µg/g)	1.4 (19) [0.8-2.7]	1.4 (19) [0.8-2.7]			

• If one or more samples were below analytical detection limits, the second number in parentheses is the number of samples which exceeded detection limits.
More than 50% of the samples were below detection limits.
c Fish include juvenile tui chub, pumpkinseed, and carp.

<u>AQuatic Vegetation and Invertebrates</u>

Heinz (1979) found that reproduction was reduced in three generations of mallards maintained on diets containing 0.5 µg/g methyl mercury. The 70-day LC_{S0} for organic mercury administered through diet of ring-necked pheasants *(Phasianus colchicus)* was 12.5 µg/g (Spann et al. 1972). In our study, one aquatic vegetation sample, 47% of the chironomid samples, and 44% of the corixid samples contained> 0.5 µg/g. No samples approached 12.5 µg/g.

Fish mortality was associated with whole fish mercury concentrations between 5 and 7 µg/g wet weight (18 and 25 µg/g dry weight; Armstrong 1979). Fathead minnow reproduction was reduced at a mercuric chloride concentration in water of 0.26 and inhibited at a concentration of 1.02 in water (Snarski and Olson 1982). Whole fish mercury residues at 60 days were 0.62 and 2.64 wet weight, respectively (2.2 and 9.5 µg/g dry weight). Mercury concern concentrations for whole fish ranged from 0.17 to 0.19 µg/g wet weight (0.6 to *0.9* µg/g dry weight; Schmitt and Brumbaugh 1990). In our study, mercury concentrations in 95% of the whole fish samples exceeded the lower concern concentration (Schmitt and Brumbaugh 1990) and 16% exceeded the 2.2 µg/g (dry weight) level associated with reduced reproduction (Snarski and Olson 1982). Mercury in whole fish samples did not approach concentrations associated with mortality (Annstrong 1979). All whole fish samples exceeded a 0.5 avian effect concentration (Heinz 1979)

Ayian Eggs and Liver Tissue

Avian nesting only occurred in our study units during the spring of 1990. Samples collected during this period were not analyzed for mercury.

Correlations. Trends. and Management Implications

Mercury contamination in Lahontan Valley is attributed to mining activities in the Carson River basin above Lahontan Valley in the mid- to late-1800's (Cooper et al. 1985; Hoffinan et al. 1990b). A median mercury concentration of 0.04 µg/g was reported in uncontaminated soils collected in Lahontan Valley (Fernley Wildlife Management Area; Lico 1992). All units examined in our study exceeded this background concentration (Table 21). Mercury was not evenly distributed in wetland sediments in our study. The highest mean mercury concentration (10.68 was found in Lead Lake, followed by Swan Lake Check, Dry Lake, and East Alkali Lake (0.54, 0.28, and 0.15 respectively). Distribution of mercury in Lahontan Valley is related to historical flow patterns prior to Carson River flow regulation (Hoffinan 1994).

Mercury was detected in only 20% of the total water samples and in 21% of the water samples from Dry Lake. Detection limits of mercury in water varied considerably over the course of our study (Appendix A). Therefore, meaningful comparisons of detected mercury concentrations in water with specific conductance or concentrations in other sample matrices were not possible.

When all units were considered, mercury concentrations in all sample matrices were not correlated to specific conductance. When Dry Lake was considered separately, mercury in chironomids was correlated to specific conductance ($P < 0.001$, $r^2 = 0.605$, $n = 16$). No obvious explanation for this is available. In Dry Lake, mercury concentrations in sediment were only correlated to mercury concentrations in chironomids ($P = 0.013$, $r^2 = 0.331$, $n = 16$).

Mercury concentrations in sediment and aquatic vegetation from Dry Lake declined over the course of our study (Table 21). The largest decline occurred after short-term desiccation. Mercury concentrations in aquatic invertebrates did not follow this same pattern.

These results do not indicate that management of specific conductance could be effectively used to control mercury in Lahontan Valley wetlands. The possibility of using desiccation to reduce mercury contamination needs to be further investigated.

Molybdenum

Water

The Nevada water quality standard for the protection of aquatic life for molybdenum is (NAC 445.1339). Adverse effects (LC$_{t}$) to rainbow trout were found at a molybdenum concentration of 28 with substantial mortality (LC$_{S0}$) at a molybdenum concentration of 790 (Birge et al. 1979a). Molybdenum was relatively non-toxic to eggs, alvins, and fry life stages of chinook salmon *(Oncorhynchus tshawytscha)* and coho salmon *(Oncorhynchus kisutch),* with no mortality observed at concentrations up to 1,000 mgIL (Hamilton and BOOl 1990). In our study, concentrations in all water samples exceeded the Nevada aquatic life

Table 21. Mercury concentrations in unfiltered water, sediment, pondweed, and aquatic invertebrates collected from four wetlands on Stillwater National Wildlife Refuge, 1990-93.

Wetland and Sampling Period	Sample Matrix'				
	Water (/lg/L)	Sediment (/lg/g)	Pondweed (/lg/g)	Chironomid (/lg/g)	Corixid (/lg/g)
Dry Lake					
Spring 1990					
Fall 1990	_b (5/1)< [-]	0.41 (SIS) [0.05]	0.49 (2/2) [0.08]	0.42 (1/1) [-]	0.26 (SIS) [0.04]
Fall 1991	_b (5/0) [-]	0.31 (SIS) [0.05]	0.17 (2/1) [0.05]	0.36 (3/3) [0.02]	0.35 (4/4) [0.02]
Spring 1992	_b (5/0) [-]	0.21 (SIS) [0.07]	0.29 (5/4) [0.11]	0.58 (SIS) [0.07]	0.56 (SIS) [0.04]
Fall 1992	0.4 (SIS) [0.08]	0.29 (SIS) [0.06]	0.28 (5/4) [0.09]	0.90 (SIS) [0.17]	0.40 (SIS) [0.06]
Spring 1993	_b (5/0) [-]	0.23 (SIS) [0.05]	0.13 (SIS) [0.04]	0.42 (2/2) [0.00]	1.32 (SIS) [0.47]
Fall 1993	_b (5/0) [-]	0.23 (4/4) [0.05]	0.28 (SIS) [0.10]		
East Alkali Lake					
Spring 1990					
Fall 1990	_b (5/0) [-]	0.18 (SIS) [0.03]	0.17 (SIS) [0.02]	0.45 (SIS) [0.03]	0.29 (4/4) [0.06]
Spring 1991	_b (5/1) [-]	0.13 (4/4) [0.01]		0.41 (SIS) [0.07]	0.20 (4/4) [0.02]
Swan Lake Check					
Spring 1990					
Fall 1990	0.4 (5/4) [0.1]	0.53 (SIS) [0.01]		0.41 (SIS) [0.08]	0.73 (SIS) [0.14]
Spring 1991	_b (510) [-]	0.55 (SIS) [0.02]		0.69 (SIS) [0.10]	0.63 (4/4) [0.06]
Lead Lake					
Spring 1990					
Fall 1993	_b (510) [-]	10.68 (SIS) [0.43]			

• Concentrations are on a dry weight basis, except for water.
b More than 50% of the samples were below detection limits.
< Numbers include arithmetic mean, (number of sampleslnumber of samples with detectable quantities), and [standard deviation].

Table 22. Comparison of molybdenum concentrations (dry weight, except water) in unfiltered water, sediment, and biological matrices collected from four wetlands on Stillwater National Wildlife Refuge, 1990-93. Numbers include mean, (number of samples'), and [sample range].

Sample Matrix	Total	Wetlands: Dry Lake	East Alkali Lake	SwanLake Check	Lead Lake
Water (µg/L)	98 (74) [20-280]	68 (34) [30-146]	92 (15) [73-117]	153 (15) [70-280]	127(10) [20-236]
Sediment (µg/g)	_b (74/31) [<1-17]	_b (34/9) [<1-9]	_b (15/7) [<1-7]	5 (15/11) [<5-10]	_b (10/4) [<1-17]
Pondweed (µg/g)	6 (29/24) [<10-12]	6 (24/19) [<10-12]	5 (5) [3-7]		
Chironomid (µg/g)	3 (54/49) [<1-6]	4 (21/18) [<4-6]	2 (14/13) [<1-4]	3 (15/14) [<4-6]	4 (4) [3-5]
Corixid (µg/g)	3 (61/55) [<1-9]	4 (29) [1-9]	2 (13/10) [<1-5]	3 (14) [2-8]	_b (5/2) [<1-5]
Fish[c] (µg/g)	1 (19/12) [<1-2]	1 (19/12) [<1-2]			
Stilt egg (µg/g)	_b (4/0) [<0.5]				_b (4/0) [<0.5]
Coot egg (µg/g)	0.8 (28/16) [<0.5-3.0]	0.7 (6/4) [<0.5-1.2]	_b (10/4) [<0.5-1.2]	0.6 (2/1) [<0.5-0.9]	1.0 (10/7) [<0.5-3.0]
Stilt liver (µg/g)	2.6 (36) [1.0-5.2]	2.6 (10) [1.5-3.8]	2.6 (10) [1.9-3.1]	3.3 (5) [2.3-5.2]	2.3(11) [1.0-3.0]
Coot liver (µg/g)	5.7 (41/40) [<0.5-1004]	6.1 (10) [4.3-8.2]	5.8 (11110) [<0.5-1004]	5.0 (10) [4.0-6.8]	5.7 (10) [4.2-7.3]

, If one or more samples were below analytical detection limits, the second number in parentheses is the number of samples which exceeded detection limits.
More than 50% of the samples were below detection limits.
c Fish include juvenile tui chub, pumpkinseed, and carp.

standard (Table 22) and 97% exceeded the 28 µg/L concern concentration (Birge et al. 1980). No samples approached concentrations associated with significant mortality.

<u>Aquatic vegetation and Inyertebrates</u>

An avian dietary concern range (reduced growth) was identified between 200 and 300 µg/g molybdenum (Eisler 1989). Reproductive impairment was associated with dietary concentrations exceeding 500 µg/g. Mortality was associated with dietary concentrations greater than 6,000 µg/g. In our study, molybdenum concentrations in aquatic vegetation and invertebrates were well below avian dietary concern or effect levels.

No infonnation was found on whole fish effect concentrations. All whole fish samples were well below avian dietary effect levels (Eisler 1989).

Avian Eggs and Liver Tissue

Molybdenum concentrations of 16 to 20 µg/g in avian eggs was embryotoxic (Friberg et al. 1975). No biological samples in our study approached 16 µg/g.

Correlations. Trends. and Management Implications

Molybdenum concentrations in water were correlated to specific conductance when all units were considered ($P < 0.001$, $r^2 = 0.363$, $n = 73$) and when Dry Lake was considered separately ($P < 0.001$, $r^2 = 0.755$, $n = 33$). Molybdenum concentrations in sediment were correlated to molybdenum concentrations in water when all units were considered ($P < 0.001$, $r^2 = 0.512$, $n = 72$), but not when Dry Lake was considered separately.

Molybdenum concentrations in all sample matrices did not decline during the study (Table 23). Concentrations were within the range of nonnal sample variability following short-tenn desiccation of Dry Lake. Concentrations in water and sediment declined in Lead Lake following a longer period of desiccation.

These results suggest that management of specific conductance may be used with some success to control molybdenum concentrations in water and sediment. The possibility of using long-tenn desiccation to control molybdenum should be investigated further.

Selenium

Water

The Nevada water quality standard for the protection of aquatic life for selenium is 5 µg/L (NAC 445.1339). Reproductive failure and mortality of fish and waterfowl were associated with selenium levels in water between 2 and 5 µg/L (Lemly and Smith 1987). The lowest water-borne concentrations of selenium associated with observed effects to fish and wildlife ranged from 1.0 to 3.0 µg/L (Skorupa and Ohlendorf 1991; Ohlendorf et at. 1993; Skorupa et at. 1996). Similarly, Lemly (1996) reported that a selenium concentration> 2.0 µg/L in filtered water (0.45 µm filter) may affect the health and long-tenn survival of fish and wildlife populations. In our study, selenium was below detection limits in 88% of the water samples (Table 24). However, detection limits ranged from 0.5 to 5.6 µg/L. The highest observed selenium value in water during our study was 1 µg/L.

Sediment

The minimum selenium concentration in sediment associated with adverse effects on avian reproduction was 1.0 µg/g, whereas the minimum concentration associated with effects to fish was 3.0 µg/g (Skorupa 1996). Selenium concentrations ≥4 µg/g in aquatic sediments have been associated with significant adverse effects to fish and wildlife (Skorupa 1996). In our study, concentrations in 50% of the sediment samples were below detection limits; however, detection limits ranged from 0.2 to 5.0 µg/g. Twelve percent of the sediment samples exceeded 1.0 µg/g and no sediment samples exceeded 4.0 µg/g.

Table 23. Molybdenum concentrations in unfiltered water, sediment, pondweed, and aquatic invertebrates collected from four wetlands on Stillwater National Wildlife Refuge, 1990-93.

Wetland and Sampling Period	Matrix'				
	Water (µg/L)	Sediment (µg/g)	Pondweed (µg/g)	Chironomid (µg/g)	Corixid (µg/g)
Dry Lake					
Spring 1990	77.0 (4/4)[b] [7.0]	6.4 (5/4) [2.1]		2.9 (5/5) [0.7]	2.8 (5/5) [0.3]
Fall 1990	47.0 (SIS) [6.0]	1.8 (SIS) [0.4]	4.9 (2/2) [0.1]	_c (1/0) [-]	2.8 (SIS) [0.7]
Fall 1991	70.0 (SIS) [7.0]	_c (5/0) [-]	7.4 (2/2) [2.6]	5.6 (3/3) [0.2]	7.3 (4/4) [0.6]
Spring 1992	37.0 (5/5) [6.0]	_c (5/0) [-]	6.9 (5/5) [2.0]	5.8 (SIS) [0.4]	4.1 (SIS) [1.5]
Fall 1992	132.0 (SIS) [10.0]	_c (5/0) [-]	7.6 (5/5) [0.8]	3.9 (SIS) [1.5]	5.6 (SIS) [1.6]
Spring 1993	35.0 (5/5) [5.0]	_c (5/0) [-]	_c (5/2) [-]	_c (2/0) [-]	2.0 (SIS) [0.9]
Fall 1993	71.0 (SIS) [9.0]	_c (4/0) [-]	5.0 (5/3) [0.2]		
East Alkali Lake					
Spring 1990	100.0 (5/5) [10.0]	_c (5/1) [-]		2.5 (5/5) [0.7]	_c (5/2) [-]
Fall 1990	82.0 (SIS) [2.0]	1.1 (5/4) [0.5]	5.0 (SIS) [1.4]	2.0 (4/4) [0.0]	2.9 [I.]
Spring 1991	94.0 (5/5) [14.0]	_c f5/2) -]		1.7 (5/4) [0.9]	2.8 (4/4) [0.4]
Swan Lake Check					
Spring 1990	88.0 (5/5) [19.0]	_c f51l) -]		2.8 (SIS) [1.1]	2.0 (SIS) [0.5]
Fall 1990	110.0 (5/5) [0.0]	3.1 (SIS) [1.0]		2.6 (5/4) [0.8]	2.0 (5/5) [0.0]
Spring 1991	260.0 (5/5) [35.0]	8.2 (SIS) [1.1]		4.7 (SIS) [1.5]	6.2 (4/4) [1.3]
Lead Lake					
Spring 1990	221.0 (5/5) [11.0]	10.7 (5/4) [5.4]		3.5 (4/4) [0.8]	_c (5/2) [-]
Fall 1993	34.0 (5/5) [7.0]	_c (5/0) [-]			

• Concentrations are on a dry weight basis, except for water.
b Numbers include arithmetic mean, (number of sampleslnumber of samples with detectable quantities), and [standard deviation].
c More than 50% of the samples were below detection limits.

Table 24. Comparison of selenium concentrations (dry weight, except water) in unfiltered water, sediment, and biological matrices collected from four wetlands on Stillwater National Wildlife Refuge, 1990-93. Numbers include mean, (number of samples·), and [sample range].

Sample Matrix	Total	Wetlands: Dry Lake	East Alkali Lake	SwanLake Check	Lead Lake
Water (µg/L)	—[b] (74/9) [<0.2-1.0]	—[b] (34/4) [<0.5-1.0]	—[b] (15/5) [<0.2-1.0]	—[b] (15/0) [<2.0]	—[b] (10/0) [<2.0]
Sediment (µg/g)	0.5 (74/37) [<0.2-2.0]	—[b] (34/12) [<0.2-2.0]	—[b] (15/6) [<0.2-0.5]	0.3 (15/9) [<0.2-0.5]	1.0 (10) [0.6-1.2]
Pondweed (µg/g)	—[b] (29/12) [<0.2-1.6]	—[b] (24/7) [<0.2-1.6]	0.5 (5) [0.4-0.6]		
Chironomid (µg/g)	1.9 (55/53) [<0.3-4.9]	2.3 (21) [0.5-3.3]	1.8 (15/14) [<0.4-4.9]	1.5 (15113) [<0.3-2.8]	2.3 (4) [2.1-2.4]
Corixid (µg/g)	2.0 (61) [0.2-7.6]	1.9 (29) [0.5-3.7]	3.1 (13) [0.4-7.6]	1.3 (14) [0.2-2.6]	1.9 (5) [1.5-2.8]
Fish[c] (µg/g)	2.0 (19) [0.9-2.8]	2.0 (19) [0.9-2.8]			
Stilt egg (µg/g)	1.8 (4) [1.6-2.2]				1.8 (4) [1.6-2.2]
Coot egg (µg/g)	1.6 (28) [1.1-2.3]	1.6 (6) [1.4-2.3]	1.6 (10) [1.1-1.9]	1.6 (2) [1.6]	1.5 (10) [1.2-1.6]
Stilt liver (µg/g)	16.2 (36) [8.4-28.8]	16.7 (10) [11.3-23.9]	19.2 (10) [13.2-28.8]	14.3 (5) [10.6-18.9]	13.8 (11) [8.4-19.1]
Coot liver (µg/g)	8.4 (41) [4.1-14.1]	10.8 (10) [7.9-14.1]	7.5 (11) [4.1-13.0]	7.8 (10) [5.6-10.2]	7.7 (10) [6.4-9.6]

• If one or more samples were below analytical detection limits, the second number in rarentheses is the number of samples which exceeded detection limits.
More than 50% of the samples were below detection limits.
[c] Fish include juvenile tui chub, pumpkinseed, and carp.

<u>Aquatic</u> <u>Vegetation</u> <u>and</u> <u>Invertebrates</u>

Selenium dietary concern concentrations are 5.0 in fish and 3.0 in birds (Lemly and Smith 1987). Skorupa and Ohlendorf (1991) identified a critical avian dietary threshold of 5.0 µg/g. Avian reproductive impairment has been associated with diets containing from 2.5 to 8 selenium, whereas sublethal physiological responses and immunotoxic effects have been reported at dietary exposures 2 and 5.5 respectively (Skorupa et al. 1996). These researchers also identified a chronic lethal exposure threshold for nonbreeding birds subject to winter stress between 10 to 15 In our study, concentrations in 21% of the chironomid samples and 11% of the corixid samples exceeded 3.0 and 8% of the corixid samples exceeded 5.0 The highest selenium concentration observed in potential avian dietary items was 7.6 in a corixid sample collected from East Alkali Lake.

Concern and effect concentrations of 4.0 and 10.0 have been identified for selenium in whole fish (Lillebo et al. 1988). Concentrations in whole fish did not exceed these concentrations, or avian and fish dietary concern concentrations.

Avian Eggs and Liver Tissue

The embryo is the most sensitive avian life stage to selenium poisoning (Heinz 1996). Sensitivity to selenium varies among species (Skorupa et al. 1996). The embryotoxic and teratogenic threshold for selenium in avian eggs is between 13 and 24 (Skorupa and OWendorf 1991). Hatching success decreased at selenium concentrations above 8 in eggs. A mean concentration of 4 selenium in eggs was associated with increased susceptibility of mallard hatcWings to a duck hepatitis virus. In our study, selenium concentrations were below concentrations associated with teratogenesis or sublethal effects.

Selenium concentrations in liver were not diagnostic of death in mallards (Heinz et al. 1988). An avian liver concentration exceeding 30 was generally associated with adverse effects to aquatic birds (Skorupa et al. 1992; Skorupa et al. 1996). Concentrations below 10 were generally not associated with adverse effects. The possibility of adverse effects between these two levels was dependent on specific circumstances. In our study, all but one of the black-necked stilt liver samples (97%) and 22% of the American coot liver samples contained concentrations that exceeded 10.0 Concentrations did not exceed 30 in any samples.

Correlations. Trends. and Management Implications

Selenium concentrations were below analytical detection limits in $\geq$50% of the water, sediment, and vegetation samples (Table 25). Therefore, correlations of selenium in these matrices was not attempted. Detected concentrations of selenium in water, sediment, and pondweed did not correspond to higher levels of specific conductance. When all units were considered, specific conductance was not correlated to selenium concentrations in chironomids or corixids. When Dry Lake was considered separately, specific conductance was poorly correlated to selenium concentrations in corixids ($P = 0.045$; $r^2 = 0.140$; $n = 29$). Five corixid samples exceeded a fish dietary concern concentration (5.0 Lemly and Smith 1987) or avian dietary threshold (5.0 Skorupa and OWendorf 1991). All were collected from East Alkali Lake during the initial (spring 1990) sampling period.

In Dry Lake, selenium concentrations in chironomids and corixids did not decline during our study (Table 25). Concentrations in both increased following a period of short-term desiccation. Concentrations in sediment from Lead Lake declined over a longer period of desiccation. However, we are uncertain as to the significance. These results do not indicate that management of specific conductance could be effectively used to control selenium in Lahontan Valley wetlands. The possibility of using wetland desiccation to control selenium should be further assessed.

Table 25. Selenium concentrations in unfiltered water, sediment, pondweed, and aquatic invertebrates collected from four wetlands on Stillwater National Wildlife Refuge, 1990-93.

Wetland and Sampling Period	Sample Matrix[a]				
	Water (µg/L)	Sediment (µg/g)	Pondweed (µg/g)	Chironomid (µg/g)	Corixid (µg/g)
Lake					
Spring 1990	-[b] (4/0)[c] [-]	-[b] (5/0) [-]		1.88 (SIS) [0.91]	2.97 (SIS) [0.39]
Fall 1990	-[b] (5/1) [-]	0.24 (5/3) [0.15]	0.50 (2/2) [0.1]	0.50 (1/1) [-]	0.66 (5/5) [0.14]
Fall 1991	-[b] (5/0) [-]	-[b] (5/0) [-]	-[b] (2/0) [-]	3.01 (3/3) [0.27]	1.69 (4/4) [0.29]
Spring 1992	-[b] (5/0) [-]	-[b] (5/0) [-]	-[b] (5/0) [-]	2.60 (5/5) [0.44]	1.91 (SIS) [0.27]
Fall 1992	-[b] (5/0) [-]	1.84 (5/5) [0.09]	-[b] (5/1) [-]	1.89 (SIS) [0.40]	2.27 (SIS) [0.30]
Spring 1993	-[b] (5/1) [-]	0.26 (5/3) [0.19]	-[b] (5/1) [-]	3.00 (2/2) [0.00]	2.10 (5/5) [0.52]
Fall 1993	-[b] (5/2) [-]	-[b] (4/1) [-]	0.27 (5/3) [0.21]		
East Alkali Lake					
Spring 1990	-[b] (5/0) [-]	-[b] (5/0) [-]		3.88 (5/5) [0.69]	6.78 (5/5) [0.84]
Fall 1990	0.8 (SIS) [0.2]	0.30 (5/4) [0.13]	0.49 (SIS) [0.11]	0.56 (5/4) [0.27]	0.78 (4/4) [0.17]
Spring 1991	-[b] (5/0) [-]	-[b] (5/0) [-]		1.04 (SIS) [0.13]	0.70 (4/4) [0.25]
Swan Lake Check					
Spring 1990	-[b] (5/0) [-]	-[b] (5/0) [-]		2.23 (5/5) [0.26]	2.22 (SIS) [0.21]
Fall 1990	-[b] (5/0) [-]	0.38 (5/5) [0.07]		0.51 (5/3) [0.27]	0.52 (SIS) [0.25]
Spring 1991	-[b] (5/0) [-]	0.22 (5/4) [0.07]		1.72 (5/5) [0.78]	1.07 (4/4) [0.21]
Lead Lake					
Spring 1990	-[b] (5/0) [-]	1.05 (SIS) [0.14]		2.26 (4/4) [0.12]	1.85 (5/5) [0.47]
Fall 1993	-[b] (5/0) [-]	0.87 (SIS) [0.15]			

[a] Concentrations are on a dry weight basis, except for water.
[b] More than 50% of the samples were below detection limits.
[c] Numbers include arithmetic mean, (number of samp1eslnumber of samples with detectable quantities), and [standard deviation].

Zinc

Water

The Nevada aquatic life water quality standard for zinc is based on water hardness (NAC 445.1339). The hardness of water for wetlands in Lahontan Valley (Hoffinan et al. 1990b) equates to a 96-hour average zinc standard of 372 µg/L. Zinc in water has been found to be acutely toxic (96-hour LC,o) to invertebrates at a concentration as low as 32 µg/L (U.S. EPA 1987). Lethal and sublethal effects have been found in fish at concentrations between 50 and 235 µg/L (Eisler 1993). In our study, concentrations in 40% of the water samples exceeded 32 µg/L and 14% exceeded 50 µg/L (Table 26).

Sediment

Long and Morgan (1991) identified a lower effect concentration of 120 µg/g for zinc in coastal marine and estuarine sediments. In our study, zinc concentrations in sediment were below this concern concentration.

Aquatic Vegetation and InYertebrates

Reduced survival of mallards was found at dietary concentrations of zinc exceeding 3,000 µg/g (Gasaway and Buss 1972). Sublethal effects (immunosuppression) in domestic chickens have been found at a dietary level of 178 µg/g (Stahl et al. 1989 as cited in Eisler 1993). In our study, concentrations in 3% of the invertebrate samples exceeded 178 µg/L. No samples approached concentrations associated with reduced survival.

Zinc concern concentrations for whole fish ranged from 34.2 to 46.3 µg/g wet weight (123 to 167 µg/g dry weight; Schmitt and Brumbaugh 1990). In our study, concentrations in 63% of the fish samples exceeded the lowest concern level. Concentrations in approximately 26% of the whole fish samples exceeded an avian dietary effect level of 178 µg/g.

Avian Eggs and Liyer Tissue

A liver concentration of 401 µg/g in mallards was associated with reduced survival (Gasaway and Buss 1972). No mortality of mallards was associated with a zinc concentration of 217 µg/g in liver (French et al. 1987). In our study, concentrations did not exceed 401 µg/g in liver samples; however, several samples approached this concentration. No information was found on concern or effect concentrations for zinc in avian eggs.

Correlations. Trends. and Management Implications

Zinc concentrations in water were not correlated with specific conductance when all units were considered or Dry Lake was considered separately. Zinc concentrations in water were only weakly correlated with concentrations in pondweed (P = 0.025; r^2 = 0.178, n = 28). Zinc concentrations in all sample matrices did not decline during our study and were generally within the range of normal sample variability following short-term desiccation of Dry Lake (Table 27). These results indicate that management of specific conductance or wetland desiccation could not be effectively used to control zinc in water, sediment, or chironomids in Lahontan Valley wetlands.

Table 26. Comparison of zinc concentrations (dry weight, except water) in unfiltered water, sediment, and biological matrices collected from four wetlands on Stillwater National Wildlife Refuge, 1990-93. Numbers include mean, (number of samples"), and [sample range].

Sample Matrix	Total	Wetlands: Dry Lake	East Alkali Lake	SwanLake Check	Lead Lake
Water (μg/L)	28 (74/61) [<5-97]	34 (34/32) [<10-71]	43 (15/14) [<10-97]	14 (15/10) [<10-43]	10 (lOIS) [<5-34]
Sediment (μg/g)	84 (74) [59-119]	86 (34) [67-119]	86 (15) [67-112]	76 (15) [59-101]	90 (10) [76-98]
Pondweed (μg/g)	51 (29) [19-115]	55 (24) [22-115]	33 (5) [19-49]		
Chironomid (μg/g)	82 (54) [45-469]	101 (21) [58-469]	70 (14) [55-93]	68 (15) [45-83]	72 (4) [69-76]
Corixid (μg/g)	130 (61) [87-196]	137 (29) [119-160]	128 (13) [87-196]	113 (14) [87-143]	139 (5) [114-163]
Fish[b] (μg/g)	153 (19) [92-262]	153 (19) [92-262]			
Stilt egg (μg/g)	54 (4) [48-61]				54 (4) [48-61]
Coot egg (μg/g)	71 (28) [53-212]	70 (6) [56-90]	64 (10) [53-75]	71 (2) [70-73]	80 (10) [56-212]
Stilt liver (μg/g)	98 (36) [55-133]	91 (10) [55-107]	102 (10) [89-133]	110 (5) [86-133]	96 (11) [83-113]
Coot liver (μg/g)	220 (41) [86-391]	224 (10) [137-306]	167 (11) [86-245]	227 (10) [196-258]	267 (10) [147-391]

" If one or more samples were below analytical detection limits, the second number in parentheses is the number of samples which exceeded detection limits.
Fish include juvenile tui chub, pumpkinseed, and carp.

Table 27. Zinc concentrations in unfiltered water, sediment, pondweed, and aquatic invertebrates collected from four wetlands on Stillwater National Wildlife Refuge, 1990-93.

Wetland and Sampling Period	Matrix' Water (µg/L)	Sediment (µg/g)	Pondweed (µg/g)	Chironomid (µg/g)	Corixid (µg/g)
Dry Lake					
Spring 1990	*37.0* (4/4)[b] [4.0]	*103.5* (5/5) [10.8]		163.7 (5/5) [1S2.7]	IS0.2 (5/5) [7.1]
Fall 1990	*41.0 (SIS)* [16.0]	*80.9* (5/5) [4.6]	2S.8 (2/2) [3.8]	87.8 (1/1) [-]	*138.2* (5/5) [II.6]
Fall 1991	*12.0 (S/4)* [4.0]	*94.1 (SIS)* [11.2]	*89.0* (2/2) [8.5]	72.S (3/3) [3.0]	*137.0* [10.]
Spring 1992	SI.O *(SIS)* [11.0]	*80.1 (SIS)* [10.4]	*83.0 (SIS)* [19.4]	*90.2* (5/5) [4.0]	137.2 (5/5) [10.3]
Fall 1992	*42.0 (SIS)* [14.0]	73.5 (5/5) [2.4]	*38.4 (SIS)* [11.4]	6S.3 (5/5) [8.3]	*124.9 (SIS)* [6.2]
Spring 1993	*24.0* (5/5) [13.0]	*83.1* (5/5) [6.4]	*49.0 (SIS)* [12.3]	104.S (2/2) [4.5]	13S.2 (5/5) [9.3]
Fall 1993	*37.0 (SIS)* [13.0]	83.S (4/4) [7.4]	*47.2* (5/5) [12.8]		
East Alkali Lake					
Spring 1990	*29.0* (5/4) [16.0]	*93.8 (SIS)* [IS.6]		8S.1 *(SIS)* [6.6]	*162.6* (5/5) [22.0]
Fall 1990	*70.0 (SIS)* [21.0]	*80.7* (5/5) [3.9]	*32.6 (SIS)* [10.4]	*64.9 (4/4)* [3.S]	124.S (4/4) [3.5]
Spring 1991	*28.0* (5/5) [7.0]	*84.4* (5/5) [I.6]		S9.4 *(SIS)* [3.7]	89.S (4/4) [2.1]
Swan Lake Check					
Spring 1990	*28.0* (5/5) [10.0]	*86.2* (5/5) [14.8]		*76.1 (SIS)* [S.5]	*130.6* (5/5) [S.2]
Fall 1990	*9.0 (SIS)* [S.O]	*64.8* (5/5) [4.2]		*71.7 (SIS)* [S.6]	*112.7* (5/5) [17.0]
Spring 1991	_c (5/0) [-]	*76.2 (SIS)* [3.4]		S7.6 *(SIS)* [IS.6]	*90.8* (4/4) [3.3]
Lead Lake					
Spring 1990	*18.0* (5/5) [10.0]	*86.4* (5/5) [6.7]		*72.1 (4/4)* [1.8]	*139.0 (SIS)* [17.6]
Fall 1993	_c (5/0) [-]	*93.6* (5/5) [3.7]			

a Concentrations are on a dry weight basis, except for water.
b Numbers include arithmetic mean, (number of sampleslnumber of samples with detectable quantities), and [standard deviation].
c More than SO% of the samples were below detection limits.

HUMAN **HEALTH** CONCERNS

Human health risks associated with consumption of fish and wildlife obtained at Stillwater NWR are difficult to discern. Regulatory limits for some trace elements exist for commercial food products, but such standards generally do not apply to fish and wildlife obtained by individuals engaged in recreational activities or products that are not sold commercially. Additionally, relative risk associated with the consumption of contaminated fish or wildlife products is a function of rate of consumption. Consumption of waterfowl is expected to be low and limited to a small portion of the year (Le., hunting season). Therefore, regulatory limits for fishery products given below must be used with caution when applied to consumption of waterfowl. A recreational fishery was not present in wetlands that were sampled during our study. The potential for future development is uncertain.

Concentrations of selected trace elements in edible tissues of waterfowl and fish collected from Stillwater NWR are presented in Table 28. The following discussions include trace elements for which some regulatory limits exist in the United States or other countries. Concentrations of trace elements in the following discussion are given on a wet weight basis.

Arsenic

Most samples did not contain detectable arsenic concentrations; however, detection limits ranged from 0.02 to 0.15 µg/g. The Food and Drug Administration has established a tolerance level of 0.5 µg/g arsenic in poultry and poultry products for commercial sale (21 CFR 556.60, 1988). The United States does not have a regulatory limit for arsenic in fish or fishery products (U.S. EPA 1989). Fourteen countries have established limits (0.12 to 10 µg/g) for arsenic in fish and fishery products (U.S. EPA 1989). In our study, arsenic exceeded 0.5 µg/g in one waterfowl muscle sample.

Cadmium

Cadmium concentrations in a large majority of waterfowl muscle samples were below detection limits, which ranged from 0.02 to 0.2 µg/g. No information on tolerance levels for cadmium in poultry or poultry products was found. The United States does not have a regulatory limit for cadmium in fish or fishery products (U.S. EPA 1989). Eight countries have established limits for cadmium in fish and fishery products (U.S. EPA 1989); limits range from 0 to 5.5 µg/g, with a median of 2.25 µg/g. The cadmium concentration in one northern shoveler liver sample exceeded the median regulatory limit for fishery products.

Chromium

Chromium concentrations in the majority of waterfowl muscle and liver samples were below detection limits, which ranged from 0.12 to 1.0 µg/g. No information on tolerance levels for chromium in poultry or poultry products was found. The United States does not have a regulatory limit for chromium in fish or fishery products (U.S. EPA 1989). Only one country, Hong Kong, has a regulatory limit (1.0 µg/g) for chromium in fish or fishery products (U.S. EPA 1989). Concentrations in only one northern shoveler muscle sample and one green-winged teal liver sample exceeded 1.0 µg/g chromium; however, concentrations in 42% of the whole fish samples exceeded 1.0 µg/g.

Table 28. Concentrations (µg/g wet weight) of selected trace elements in edible tissues of waterfowl and fish collected from Stillwater National Wildlife Refuge, 1990-93. Numbers include mean, (number of samples"), and [sample range].

Trace Element	MUSCLE Green-winged teal	MUSCLE Northern shoveler	LIVER Green-winged teal	LIVER Northern shoveler	Whole fish[b]
Arsenic	_[c] (45/14) [<0.02-0.7]	_[c] (30113) [<0.02-0.3]	_[c] (45/14) [<0.02-0.2]	_[c] (3118) [<0.02-0.2]	0.2 (19/16) [<0.1-0.4]
Cadmium	_[c] (45/4) [<0.02-0.1]	_[c] (30/4) [<0.02-0.1]	0.5 (45/41) [<0.02-2.0]	0.8 (31) [<0.02-3.7]	(19/14) [<0.01-0.08]
Chromium	_[c] (45/13) [<0.1-0.6]	_[c] (30/6) [<0.1-1.2]	.[c] (45/22) [<0.1-2.2]	_[c] (31/8) [<0.1-0.3]	1.4 (19/17) [<0.1-10.9]
Copper	13.4 (45) [3.0-60.6]	17.9 (30) [5.2-37.7]	39.9 (45) [2.0-147.0]	35.9 (31) [8.1-136.0]	0.7 (19) [0.2-1.1]
Lead	_[c] (45/3)[d] [<0.1-710]	_[c] (28/0) [<4.0]	.[c] (45/18) [e] [<0.1-31]	_[c] (31113) [<0.1-1.6]	0.5 (19/17) [<0.1·1.7]
Mercury	0.25 (45) [0.02-1.0]	1.4 (30) [0.1-4.4]	0.98 (45) [0.08-12.0]	6.5 (31) [0.5-15.0]	0.2 (19) [0.1-0.5]
Selenium	0.5 (45) [0.2-1.4]	0.5 (30/28) [<0.1-1.0]	1.6 (45) [0.6-3.6]	2.5 (31) [1.3-4.2]	0.3 (19) [0.1-0.6]
Zinc	17 (45) [6-35]	24 (30) [8-47]	71 (45) [10-213]	87 (31) [23-394]	22 (19) [16-34]

• If one or more samples were below analytical detection limits, the second number in parentheses is the number of samples which exceeded detection limits.

Fish include juvenile tui chub, pumpkinseed, and carp.

[c] More than 50% of the samples were below detection limits.

[d] One green-winged teal muscle sample, with a lead concentration of 71O µg/g, may have been contaminated with a metallic lead fragment. The next highest lead concentration in green-winged teal muscle was 0.3 µg/g.

[e] One green-winged teal, with a lead concentration of 31 µg/g in the liver, may have been suffering from lead poisoning. The next highest lead concentration in green-winged teal liver was 1.1 µg/g.

Copper

No information on tolerance levels for copper in poultry or poultry products was found. The United States does not have a regulatory limit for copper in fish or fishery products (U.S. EPA 1989). Ten countries have established limits (10 to 100 µg/g) for copper in fish and fishery products (U.S. EPA 1989). The regulatory limit in six of these countries is 10 µg/g. In our study, concentrations in 42% of the green-winged teal and 57% of the northern shoveler muscle samples exceeded 10 µg/g. Copper concentrations in whole fish were well below 10 µg/g.

Lead

No information on tolerance levels for lead in poultry or poultry products was found. The United States does not have a regulatory limit for lead in fish or fishery products (U.S. EPA 1989). Nineteen countries have established limits (0.5 to 10 μg/g) for lead in fish and fishery products (U.S. EPA 1989). Eisler (1985b) recommended that dietary concentrations should not exceed 0.3 μg/g. In our study, lead was not detected in northern shoveler muscle and was detected in 9% of the green-winged teal muscle samples. However, detection limits for both species ranged from 0.12 to 4.0 μg/g. A lead concentration of 710 μg/g was found in a green-winged teal muscle sample collected in the fall 1990. The liver concentration from this bird was <4.0 μg/g; therefore, sample contamination, possibly lead shot, was suspected. Lead concentrations in waterfowl livers ranged from <0.1 to 31 μg/g, with 40% of the green-winged teal and 42% of the northern shoveler samples exceeding 0.3 μg/g. One green-winged teal had a lead concentration of 31 μg/g in the liver. Although a necropsy was not performed on the teal, the elevated lead concentration in the liver suggests lead poisoning. The next highest lead concentration in green-winged teal liver samples was 1.1 μg/g.

Lead concentrations in 74% of the whole fish samples exceeded 0.3 μg/g.

Mercury

The State of Nevada has established a public health criterion of 1.0 μg/g mercury for edible bird tissues. The United States has a regulatory limit of 1.0 μg/g for mercury in fish and fishery products (U.S. EPA 1989). Additionally, 25 other countries have regulatory limits (0.1 to 1.0 μg/g) for mercury in fish and fishery products. In our study, one green-winged teal muscle sample met the 1.0 μg/g Nevada public health criterion and 53% of the northern shoveler samples exceeded this criterion. Concentrations and public health criterion exceedances were greater in liver samples than in muscle. Concentrations in whole fish did not exceeded the criterion for fish or fishery products. However, only juvenile (young-of-year) fish were included in samples.

Selenium

No information on tolerance levels for selenium in poultry or poultry products was found. Nevada has established a public health criterion of 2.0 μg/g for edible bird tissues. The United States does not have a regulatory limit for selenium in fish or fishery products (U.S. EPA 1989). Three countries have established limits (0.05 to 2.0 μg/g) for selenium in fish and fishery products (U.S. EPA 1989). In our study, no waterfowl muscle samples exceeded the Nevada public health criterion, but concentrations in 22% of the green-winged teal and 77% of the northern shoveler liver samples exceeded this criterion. Concentrations in whole fish were below the Nevada public health criterion.

Zinc

No information on tolerance levels for zinc in poultry or poultry products was found. The United States does not have a regulatory limit for zinc in fish or fishery products (U.S. EPA 1989). Seven countries have established limits (30 to 1,000 μg/g) for zinc in fish and fishery products (U.S. EPA 1989). The limit in only one of these countries exceeds 100 μg/g. In our study, zinc concentrations in waterfowl muscle and whole fish samples did not exceed 100 μg/g. Zinc concentrations in 11% of the green-winged teal, 50% of northern shoveler muscle samples, and 11% of the whole fish exceeded 30 μg/g.

SUMMARY AND MANAGEMENT IMPLICATIONS

The quantity and quality of wetlands in Lahontan Valley have declined since the initiation of large-scale agriculture in Lahontan Valley in the early 1900's. Recent investigations have also documented concentrations of inorganic contaminants in water, sediment, and biological samples in excess of concentrations associated with adverse effects to fish and wildlife. Under the auspices of the Truckee-Carson-Pyramid Lake Water Settlement Agreement, the Department of the Interior has implemented a program to acquire rights for water to restore and maintain a portion of the historic wetlands in Lahontan Valley. Although inflow to wetlands will be partially restored, the effects of increased inflow to wetland contamination is uncertain. In 1990, the Nevada State Office and Stillwater NWR instituted a monitoring program to determine if the acquisition of water would alleviate the impacts of inorganic contaminants to fish and wildlife dependent on Stillwater NWR wetlands. This program entailed the measurement of water quality parameters and the determination of trace element concentrations in water, sediment, and biological matrices. Dissolved solids, as measured by specific conductance, and 12 of 19 chemical elements monitored in the study exceeded levels of concern in 10% or more of the water, sediment and/or biological samples (Table 29). Total dissolved solids, and the elements aluminum, arsenic, boron, mercury, and zinc frequently exceeded concentrations associated with adverse biological effects. Chromium, copper, iron, lead, molybdenum, and selenium were found at lower levels of concern. Mercury, which exceeded recommended levels for human consumption in fish and waterfowl, was the only element that appeared to pose a direct threat to human health.

Several factors may contribute to the wetland degradation and the accumulation of inorganic contaminants in the arid wetlands. The reduction of inflows to wetland systems along with other modifications of hydrologic regimes are key factors (Lemly et al. 1993). In agricultural systems, major and trace elements may be mobilized from soil and groundwater during irrigation or drainage of shallow ground water from agricultural fields. These materials may be transported to wetlands via agricultural drainage, and concentrated in wetlands through evaporative processes. Wetlands in closed hydrologic basins are more prone to the manifestation of wildlife toxicity problems from dissolved solids and trace elements (Lemly et al. 1993; Seiler 1995). Loss of water through surface evaporation and evapotranspiration concentrate dissolved constituents in water. The lack of flushing of wetlands may cause long-term accumulation of these materials.

Reduction of inflows to wetlands and modification of hydrologic regimes have contributed to the degradation of wetlands in Lahontan Valley (Hoffman et al. 1990b; Kerley et al. 1993). Upstream water diversion has substantially reduced the amount of water entering wetlands, and subsequently the area of wetlands in Lahontan Valley. The discharge of agricultural drainage to wetlands has increased the concentration of dissolved solids in the remaining wetlands. Although the load of dissolved solids has not substantially changed, the reduction of wetland area has concentrated these loads in smaller wetland areas.

Reduction of flows to wetlands has also modified other hydrologic characteristics. Historically, water flowed through sequential wetlands toward the Carson Sink (Kerley et al. 1993; Hoffman 1994). This flow-through characteristic probably moved dissolved solids through the system, eventually to Carson Sink. Through evaporative water losses, dissolved solids would be increasingly concentrated in sequential wetlands. This process controlled community development and habitat characteristics, which promoted shifts in species

Table 29. Chemical elements of concern in water, sediment, and biological matrices collected from four wetlands on Stillwater NWR, 1990-1993. "S" indicates that >10% of the samples exceeded water quality standards or criteria; "E" indicates that >10% of the samples exceeded a level associated with an adverse biological effect; "C" indicates >10% of the samples exceeded a biological concern level; "N" indicates that <10% of the samples exceeded water quality standards or biological effect or concern levels; and "-" indicates that data were not available.

				Avian Diet					
Element	Water	Sediment	Whole Fish	Pond-weed	Chiro-nomid	Corixid	Fish	Avian Egg	Avian Liver
Aluminum	S, E			E	E	E	N		
Arsenic	S, E	N	C	E	N	N	N	N	N
Boron	S, C			E	C	C	C	N	N
Cadmium	N		C	N	N	N	N		
Chromium		N	C	C	C	N			N
Copper	N	N	C						
Iron	S	C							
Lead	S, C	C	C	N	N	N	N	N	N
Mercury	S, E	E	E	E	E	E	E		
Molybdenum	S, C			N	N	N	N	N	
Selenium	N	N	N	N	C	C	N	N	C
Zinc	E	N	C	N	N	N	C		C

composition and community structure in sequential wetlands. However, reduction of flows to wetlands caused the hydrologic isolation of several wetlands and reduced the amount of water flowing though others, which promoted the accumulation and concentration of dissolved and suspended solids. Additionally, diking and regulation of water flow on Stillwater NWR, in an attempt to maximize the beneficial use of reduced volumes of water entering Stillwater NWR, increased the retention time of water in wetland units (U.S. Fish and Wildlife Service 1995). This increase in hydrologic retention may have promoted additional concentration and accumulation of water-borne contaminants in wetlands.

The decrease of inflow, modification of hydrologic characteristics, modification of geochemical processes, and the concentration of dissolved solids and trace elements in Lahontan Valley wetlands have, or have the potential to affect fish, wildlife, and habitat quality. Our results, along with previous investigations (Hoffman et al. 1990b; Rowe et al. 1991; Hallock and Hallock 1993) identified several trace elements in water, sediment, food chains, and tissue at levels that may adversely affect fish, wildlife, and habitat quality. Several trace elements in water exceeded toxic thresholds of sensitive species. Water in some wetlands was also found to be toxic to aquatic organisms (Dwyer et al. 1990; Dwyer et al. 1992; Finger et al. 1993). Additionally, combinations of certain trace elements may produce a greater toxic response. Dwyer et al. (1990, 1992) found that water containing arsenic, boron, copper, lithium, molybdenum, and strontium, at concentrations similar to those found in Stillwater NWR wetlands was acutely toxic to striped bass. When each element was tested individually, no mortality occurred. Similarly, Finger et al. (1993) found appreciable mortality of fish and invertebrates exposed to water collected from wetlands on Stillwater NWR. Again, toxicity was not attributed to a single element, but it was noted that arsenic, boron, lithium, and molybdenum were elevated in water where appreciable mortality occurred. The combined effect of elevated trace elements in water and diet to migratory birds or other terrestrial wildlife is not known. However, dissolved solids and several of the trace elements monitored during our study existed at levels that may affect plants, invertebrates, and bird survival and productivity. The degradation of water quality entering Stillwater NWR, coupled with water management practices restricted habitat variability and quality. On a community level, the combined effects of these factors have caused declines in marsh productivity, habitat variability, species diversity, and avian production (Hoffman et al. 1990b; Kerley et al. 1993).

Lemly et al. (1993) identified two measures that are needed to reverse the degradation of arid wetlands: 1) restoration of high quality water inflow to wetlands, and 2) reduction of loading of contaminants to wetlands. Under the accords of the Truckee-Carson-Pyramid Lake Water Settlement Agreement, the volume of high quality water flowing into Stillwater NWR will be increased. However, the agreement also stipulates that Stillwater NWR must continue to receive agricultural drainage from the Newlands Project. Because of dissolved solids loads associated with agricultural drainwater and irrigation quality water, the load of dissolved solids entering wetlands of Stillwater NWR is not expected to be reduced (U.S. Fish and Wildlife Service 1995). Under some water rights acquisition scenarios, loading may actually increase. Our study indicates that increasing inflow of good quality water alone will not eliminate the potential for adverse effects of dissolved solids or trace elements to fish and wildlife in Stillwater NWR. Therefore, other measures to control dissolved solids and trace element contamination are needed. Refinement of existing water management practices may hold the greatest promise for reducing and managing contamination.

Our study identified statistical correlations between concentrations of a number of trace elements in unfiltered water and concentrations in sediment and/or biological matrices. This would suggest that a reduction of certain trace element concentrations in sediment and food chains could be achieved through control of trace element concentrations in water. Correlations between specific conductance and concentrations of some elements in unfiltered water were also identified. Relationships were strongest for arsenic, boron, and molybdenum, although there were weaker correlations for copper and zinc. This would suggest that a reduction of total dissolved solids in water would contribute to a reduction of concentrations of certain trace elements in water, and therefore, in sediment and food chains. Again, the greatest reductions may be realized in the concentrations of arsenic, boron, and molybdenum. Additionally, elevated arsenic, molybdenum, and copper have been associated with toxicity of water in Lahontan Valley. Therefore, control of specific conductance has the potential to reduce toxicity of water to aquatic organisms. The lack of correlations between specific conductance and concentrations of aluminum and mercury in unfiltered water indicate that control of dissolved solids would be ineffective in controlling concerns with these elements.

Little information on the management of dissolved or suspended solids in arid wetlands is available. Because of their terminal nature, wetlands in hydrologically isolated basins are susceptible to accumulation and concentration of dissolved and suspended solids (Lemly et al. 1993; Seiler 1995). In such basins, water is lost through evaporation, but dissolved and suspended solids remain. Over time, these materials accumulate and become concentrated. To alleviate long-term accumulation of water soluble contaminants, regular flushing of wetlands is needed. Water acquired under the Truckee-Carson-Pyramid Lake Water Settlement Agreement may provide sufficient water for flushing wetlands higher on the hydrologic gradient. Flushing of these wetlands could be enhanced by delivering water to lower gradient wetlands through sequential wetlands as opposed to delivering water via canals. Flushing of lower gradient wetlands would be limited to infrequent periods of flooding.

Hydrologic retention time will affect evaporative rates. In general, larger wetlands with a longer hydrologic retention times lose proportionally more water than smaller systems with shorter retention times (Herron 1986). This indicates that reducing the size of impoundments on Stillwater NWR would enable more effective management and maintenance of water quality.

Chemical species, and subsequently mobility, toxicity, and biological availability of trace elements in marsh systems is influenced by redox potential, pH, organic matter, and water hardness (Giblin 1985; Mitsch and Gosselink 1986; Masscheleyn and Patrick 1993). Desiccation and flooding of wetlands will alter these parameters and may affect the biological availability or toxicity of certain elements. Desiccation of marshes has been used to control availability of certain trace elements in western Nevada and elsewhere (Lori Carpenter, Huffman and Associates, 1996, pers. comm.). However, success of these measures to control trace elements may depend on timing of desiccation and reflooding of wetlands. In our study, concentrations of trace elements in water, sediment, or biological samples were typically within normal sample variability following short-term desiccation of Dry Lake. Insufficient data were available to discern trends following long-term desiccation of other wetlands.

RECOMMENDATIONS

1) To the extent possible and practicable, flow-through management of marshes should be restored. Such management would promote the movement of waterbome contaminants through sequential wetlands. Because contaminant mobility, concentration, and accumulation are related to water volume and hydrologic retention time, the best results may be attained through the movement of larger volumes of water. Therefore, delivery of water through wetland units, as opposed to delivery via canals running parallel to natural flow gradients, should be considered. Additionally, the degradation of water quality in sequential wetlands would promote restoration of habitat variability. Therefore, management of water in this manner may further the objective of Stillwater NWR management, as defined in the Truckee-Carson-Pyramid Lake Water Settlement Agreement, to "restore and protect natural biological diversity."

2) Hydrologic retention time in wetlands will affect concentration of dissolved solids, and, therefore, concentrations of some agricultural drainage-related trace elements. Measures to reduce water retention time in wetlands should be considered. One method of reducing hydrologic retention time is the reduction of impoundment sizes.

3) The relationship between specific conductance and some agricultural drainage-related trace elements indicates that specific conductance may be used as a cost-effective tool for the monitoring and management of dissolved solids and some trace elements. Stillwater NWR should consider instituting a specific conductance monitoring program. Monitoring data would be necessary to define and refine management practices.

4) Mercury in Stillwater NWR wetlands may present the greatest risk to migratory birds and humans. Our study did not indicate that mercury impacts could be reduced through increasing flows through wetlands, reduction of impoundment sizes, or managing specific conductance. Desiccation of wetlands to volatilize mercury should be further evaluated. Because mercury contamination on Stillwater NWR appears to be confined to a few wetlands, permanent desiccation of mercury-contaminated wetlands should also be considered.

REFERENCES

Annstrong, F.A.J. 1979. Effects of mercury compounds on fish. Pages 657-670 in J.O. Nriagu (ed.). The biogeochemistry of mercury in the envirorunent. ElsevierlNorth-Hollarid Biomedical Press, New York.

Bailey, V. 1898. Physiography, Nevada: Carson Lake .PM6 (Wadsworth, Ragtown and Stillwater). Unpublished manuscript, Smithsonian Institution Archive, Record Unit 7176, U.S. Fish and Wildlife Service, 1860-1961, Field Reports.

Biesinger, K.E., and G.M. Christensen. 1972. Effects of various metals on survival, growth, reproduction, and metabolism of *Daphnia magna*. 1 Fish. Res. Board Canada 29:1691-1700.

Birge, W.J., and J.A Black. 1977. Sensitivity of vertebrate embryos to boron compounds. U.S. Envirorunental Protection Agency, EPA 560/1-76-008.

_____, _____, AG. Westerman, and J.E. Hudson. 1979a. Aquatic toxicity tests on inorganic elements occurring in oil shale. Pages 519-534 in C. Gale (ed.). Oil shale symposium: Sampling, analysis, and quality assurance. U.S. Envirorunental Protection Agency, EPA 600/9-80-022.

_____, _____, _____, and _____. 1979b. The effect of mercury on reproduction offish and amphibians. Pages 629-655 in 10. Nriagu (ed.). The biogeochemistry of mercury in the envirorunent. ElsevierlNorth Holland Biomedical Press, New York.

Cain, B.W., L. Sileo, J.C. Franson, and J. Moore. 1983. Effects of dietary cadmium on mallard ducklings. Environ. Res. 32:286-297.

Camardese, M.B., D.J. Hoffman, L.J. LeCaptain, and G.W. Pendleton. 1990. Effects of arsenate on growth and physiology in mallard ducklings. Environ. Toxico!. Chern. 9:785-795.

Cooper, lJ., R.O. Thomas, and S.M. Reed. 1985. Total mercury in sediment, water, and fishes in the Carson River drainage, west-central Nevada. Nevada Division of Envirorunental Protection, Carson City, Nevada. 96 pp.

DeLonay, A.l, E.B. Little, D.F. Woodward, W.G. Brumbaugh, A.M. Farag, and C.F. Rabeni. 1993. Sensitivity of early-life-stage golden trout to low pH and elevated aluminum. Environ. Toxico!. Chern. 12:1223-1232.

Dwyer, F.J., S.A Burch, and C.G. Ingersol!. 1990. Investigations of the combined toxicity of trace elements and salinity to aquatic organisms at Stillwater Wildlife Management Area. U.S. Fish and Wildlife Service, Columbia, Missouri. 47 pp.

____-, ____-, ____-" and J.B. Hunn. 1992. Toxicity of trace element and salinity mixtures to striped bass *(Marone saxatilis)* and *Daphnia magna*. Environ. Toxico!. Chern. 11:513-520.

Eisler, R. 1985. Cadmium hazards to fish, wildlife, and invertebrates: A synoptic review. U.S. Fish and Wildlife Service Biological Report 85(1.2). 46 pp.

_____. 1986. Chromium hazards to fish, wildlife, and invertebrates: A synoptic review. U.S. Fish and Wildlife Service Biological Report 85(1.6). 60 pp.

_____. 1988. Lead hazards to fish, wildlife, and invertebrates: A synoptic review. U.S. Fish and Wildlife Service Biological Report 85(1.14). 134 pp.

_____. 1989. Molybdenum hazards to fish, wildlife, and invertebrates: A synoptic review. U.S. Fish and Wildlife Service Biological Report 85(1.19). 61 pp.

_____. 1993. Zinc hazards to fish, wildlife, and invertebrates: A synoptic review. U.S. Fish and Wildlife Service Biological Report 10. 106 pp.

Farag, A.M., D.F. Woodward, E.E. Little, B. Steadman, and F.A. Vertucci. 1993. The effects of low pH and elevated aluminum on Yellowstone cutthroat trout *(Oncorhynchus clarki bouvieri)*. Environ. Toxico!. Chern. 12:719-731.

Finger, S.E., S.J. Olson, and A.C. Livingstone. 1993. Toxicity of irrigation drainage and its effect on aquatic organisms. Pages 21-37 *in*, R.I. Hallock and L.L. Hallock (eds.). Detailed study of irrigation drainage in and near wildlife management areas, west-central Nevada, 1987-90: Part B. Effect on biota in Stillwater and Fernley Wildlife Management Areas and other nearby wetlands. U.S. Geological Survey Water-Resources Investigations Report 92-4024B. 84 pp.

Finley, M.T., M.P. Dieter, and L.N. Locke. 1976. Sublethal effects of chronic lead ingestion in mallard ducks. J. Toxico!. Environ. Health 1:929-937.

French, M.C., C.W. Haines, and J. Cooper. 1987. Investigation into the effects of ingestion of zinc shot by mallard ducks *(Anas platyrhynchos)*. Environ. Pollut. 47:305-314.

Friberg, L., P. Boston, G. Nordberg., M. Piscator, and K.H. Robert. 1975. Molybdenum - a toxicological appraisal. U.S. Environmental Protection Agency, EPA 600/1-75-004.

Gasaway, W.C., and 1.0. Buss. 1972. Zinc toxicity in the mallard duck. J. Wild!. Manage. 36:1107-1117.

Gersich, F.M. 1984. Evaluation of static renewal chronic toxicity test method for *Daphnia magna* Straus using boric acid. Environ. Toxico!. Chern. 3:89-94.

Giblin, A.E. 1985. Comparisons of the processing of elements by ecosystems, II, metals. Pages *158-179 in* P.J. Godfrey, E.R. Kaynor, S. Pelczarski, and J. Benforado (eds.). Ecological considerations in wetland treatment of municipal wastewaters. Van Nostrand Rienhold Company, New York, New York. 473 pp.

Gilderhus, P.A. 1966. Some effects of sublethal concentrations of sodium arsenite on bluegills and the aquatic environment. Trans. Am. Fish Soc. 95:289-296.

Giles, M.A., and J.F. Klaverkamp. 1982. The acute toxicity of vanadium and copper to eyed eggs of rainbow trout *(Salmo gairdneri)*. Water Research 16:885-889.

Hall, L.W., S.J. Bushong, M.C. Ziegenfuss, W.S. Hall, and R.L. Herriman. 1988. Concurrent mobile and *in-situ* striped bass contaminant and water quality studies in the Choptank River and upper Chesapeake Bay. Environ. Toxico!. Chern. 7:815-830.

Hallock, R.J., and L.L. Hallock, (eds.). 1993. Detailed study of irrigation drainage in and near wildlife management areas, west-central Nevada, 1987-90: Part B. Effect on biota in Stillwater and Fernley Wildlife Management Areas and other nearby wetlands. U.S. Geological Survey Water- Resources Investigations Report 92-4024B. 84 pp.

Hamilton, S.I., and K.J. Buhl. 1990. Acute toxicity of boron, molybdenum, and selenium to fry of chinook salmon and coho salmon. Arch. Environ. Contam. Toxico!. 19:366-373.

Haseltine, S.D., L. Sileo, D.I. Hoffman, and B.D. Mulhern. 1985. Effects of chromium on reproduction and growth of black ducks. U.S. Fish and Wildlife Service, Patuxent Wildlife Research Center, Laurel, Maryland. Unpub!. Report. 25 pp.

Heinz, G.H. 1979. Methylmercury: Reproductive and behavioral effects on three generations of mallard ducks. J. Wild!. Manage. 43:394-401.

_____. 1996. Selenium in birds. Pages 447-458 *in* W.N. Beyer, G.H. Heinz, and A.W. Redmon-Norwood (eds.), Environmental Contaminants in Wildlife; Interpreting Tissue Concentrations. Lewis Publishers, Boca Raton, Florida.

_____' D.J. Hoffman, and L.G. Gold. 1988. Toxicity of organic and inorganic selenium to mallard ducklings. Arch. Environ. Contam. Toxico!. 17:561-568.

Herron, G.B., C.A Mortimore, and M.S. Rawlings. 1985. Nevada raptors. Nevada Department of Wildlife Biological Bulletin No.8. Nevada Department of Wildlife, Reno, Nevada. 114 pp.

Herron, RC. 1986. Wetlands and water quality in Great Basin/desert and montane region of the United States. Pages 60-114 *in* lH. Sather and J.Low (eds.) Great Basin desert and montane regional wetlands functions. The Environmental Institute, University of Massachusetts at Amherst Publication Number 90-4.

Hoffman, D.J., M.B. Camardese, L.J. LeCaptain, and G.W. Pendleton. 1990a. Effects of boron on growth and physiology in mallard ducklings. Environ. Toxicol. Chern. 9:335-346.

_____' J.C. Franson, O.H. Pattee, C.M. Bunck, and A Anderson. 1985. Survival, growth, and accumulation of ingested lead in nestling American kestrels *(Falco sparverius).* Arch. Environ. Contam. Toxicol. 14:89-94.

_____, CJ. Sanderson, L.l LeCaptain, E. Cromartie, and G.W. Pendleton. 1991. Interactive effects of boron, selenium, and dietary protein on survival, growth, and physiology in mallard ducklings. Arch. Environ. Contam. Toxicol. 20:288-294.

Hoffman, RJ. 1994. Detailed study of irrigation drainage in and near wildlife management areas, west-central Nevada, 1987-90: Part C. Sununary of irrigation-drainage effects on water quality, bottom sediment, and biota. U.S. Geological Survey Water-Resources Investigations Report 92-4024C. 32 pp.

_____, R.J. Hallock, T.G. Rowe, M.S. Lico, H.L. Burge, and S.P. Thompson. 1990b. Reconnaissance investigation of water quality, bottom sediment, and biota associated with irrigation drainage in and near Stillwater Wildlife Management Area, Churchill County, Nevada, 1986-87. U.S. Geological Survey Water-Resources Investigations Report 89-4105. 150 pp.

Hughes, J.S. 1973. Acute toxicity of thirty chemicals to striped bass *(Marone saxatilis).* Proc. Western Assoc. State Game Fish Comm. 53:399-413. (reference not seen).

Kania, HJ., and J.O. O'Hara. 1974. Behavioral alterations in a simple predator-prey system due to sublethal exposure to mercury. Trans. Am. Fish. Soc. 103:134-136.

Kerley, L.L., G.A. Ekechukwu, and R.l Hallock. 1993. Estimated historical conditions of the lower Carson River wetlands. Pages 7-20 *in,* RJ. Hallock and L.L. Hallock (eds.). Detailed study of irrigation drainage in and near wildlife management areas, west-central Nevada, 1987-90: Part B. Effect on biota in Stillwater and Fernley Wildlife Management Areas and other nearby wetlands. U.S. Geological Survey Water-Resources Investigations Report 92-4024B. 84 pp.

Lernly, AD. 1996. Selenium in aquatic organisms. Pages 427-455 *in* W.N. Beyer, G.H. Heinz, and A.W. Redmon-Norwood (eds.), Environmental Contaminants in Wildlife; Interpreting Tissue Concentrations. Lewis Publishers, Boca Raton, Florida. 494 pp.

_____, S.E. Finger, and M.K. Nelson. 1993. Sources and impacts of irrigation drainwater contaminants in arid wetlands. Environ. Toxicol. Chern. 12:2265-2279.

_____, and GJ. Smith. 1987. Aquatic cycling of selenium: Implications for fish and wildlife. U.S. Fish and Wildlife Service, Fish and Wildlife Leaflet 12. 10 pp.

Lewis, M.A., and L.C. Valentine. 1981. Acute and chronic toxicities of boric acid to *Daphnia magna* Straus. Bull. Environ. Contam. Toxicol. 27:309-315.

Lieo, M.S. 1992. Detailed study of irrigation drainage in and near wildlife management areas, west-central Nevada, 1987-90: Part A. Water quality, sediment composition, and hydrogeochemical processes in Stillwater and Fernley Wildlife Management Areas. U.S. Geological Survey Water-Resources Investigations Report 92-4024A 65 pp.

Lillebo, P.H., S. Shaner, P. Carlson, N. Richard, and P. Dubarry. 1988. Regulation of agricultural drainage to the San Joaquin River. Califomia State Water Resources Control Board Report W.O. 85-1. 151 pp.

Long, E.R., and L.G. Morgan. 1991. The potential for biological effects of sediment-sorbed contaminants tested in the National Status and Trends Program. National Oceanic and Atmospheric Administration Technical Memorandum NOS OMA 52. 175 pp. plus appendices.

Masscheleyn, P.H., and W.H. Patrick, Jr. 1993. Biogeochemical processes affecting selenium cycling in wetlands. Environ. Toxicol. Chern. 12:2235-2243.

Miles, AK., C.E. Grue, G.W. Pendleton, and J.H. Soares, Jr. 1993. Effects of dietary aluminum, calcium, and phosphorus on egg and bone of European starlings. Arch. Environ. Contam. Toxicol. 24:206-212.

Mitcham, S.A., and G. Wobeser. 1988. Toxic effects of natural saline waters on mallard ducklings. J. Wildl. Dis. 24:45-50.

Mitsch, W.I., and J.G. Gosselink. 1986. Wetlands. Van Nostrand Reinhold, New York, New York. 539 pp.

Moore, S.B., J. Winckel, S.J. Detwiler, S.A. Klasing, P.A Gual, N.R. Kanim, B.E. Kesser, A.B. DeBevec, K. Beardsley, and L.K. Puckett. 1990. Fish and wildlife resources and agricultural drainage in the San Joaquin Valley, California. San Joaquin Valley Drainage Program, Sacramento, California.

Morgan, D.S. 1982. Hydrogeology of the Stillwater area, Churchill County, Nevada. U.S. Geological Survey Open File Report 82-345. 95 pp.

National Academy of Sciences. 1977. Arsenic. National Academy of Sciences, Washington D.C. 332pp.

Nicholson, J.K., and D. Osborn. 1984. Kidney lesions in juvenile starlings Stumus vulgaris fed on a mercury-contaminated synthetic diet. Environ. Pollut. 33A:195-206.

Nyholm, N.E.I. 1981. Evidence of involvement of aluminum in causation of defective formation of eggshells and of impaired breeding in wild passerine birds. Environ. Res. 26:363-371.

OWendorf, H.M., J.P. Skorupa, M.K. Saiki, and D.A Barnum. 1993. Food-chain transfer of trace elements to wildlife. Pages 596-603 *in* R.G. Allen and C.M.U. Neale (eds.) Management ofIrrigation and Drainage Systems: Integrated Perspectives. Proceedings of the 1993 National Conference on Irrigation and Drainage Engineering. American Society of Civil Engineers, New York, N.Y.

Oladimeji, AA., S.U. Qadri, and AS.W. deFreitas. 1984. Long-term effects of arsenic accumulation in rainbow trout, *Salmo gairdneri*. Bull. Environ. Contam. Toxicol. 32:732-741.

Patuxent Analytical Control Facility. 1990. Reference Manual: U.S. Fish and Wildlife Service, Patuxent Analytical Control Facility, Laurel, Maryland. 89 pp. plus appendices.

Persaud, D., R. Jaagumagi, and A Hayton. 1993. Guidelines for the protection and management of aquatic sediment quality in Ontario. Ontario Ministry of the Environment and Energy, Ontario.

Rowe, T.G., M.S. Lico, R.I. Hallock, AS. Maest, and R.I. Hoffman. 1991. Physical, chemical, and biological data for detailed study of irrigation drainage in and near Stillwater,

Fernley, and Humboldt Wildlife Management Areas and Carson Lake, west-central Nevada, 1987-89. U.S. Geological Survey Open-File Report 91-185. 199 pp.
Russell,I.C. 1885. Geological history of Lake Lahontan, A quartenary lake in northwestern Nevada. U.S. Geological Survey Monograph, Vo!. II, Washington D. C. 288 pp.
Schmitt, C.J., and W.G. Brumbaugh. 1990. National Contaminants Biomonitoring Program: Concentrations of arsenic, cadmium, copper, lead, mercury, selenium, and zinc in U.S. freshwater fish, 1976-1984. Arch. Environ. Contam. Toxico!. 19:731-747.
Schmitt, D.N. 1988. Mammalian Fauna. Pages 262-290 *in* C. Raven and RG. Elston, R.G. (eds.). Preliminary investigations in Stillwater Marsh; Human Prehistory and Geoarcheology, U.S. Fish and Wildlife Service, Portland, Oregon.
Schroeder, R.A., D.U. Palawski, and J.P. Skorupa. 1988. Reconnaissance investigation of water quality, bottom sediment, and biota associated with irrigation drainage in the Tulare Lake bed area, southern San Joaquin Valley, California, 1986-87. U.S. Geological Survey Water-Resources Investigations Report 88-4001. 86 pp.
Seiler, RL. 1995. Prediction of areas where irrigation drainage may induce selenium contamination of water. J. of Environmental Quality 24:973-979.
Setmire, J.G., J.C. Wolfe, and R.K. Stroud. 1990. Reconnaissance investigation of water quality, bottom sediment, and biota associated with irrigation drainage in the Salton Sea area, California, 1986-87. U.S. Geological Survey Water-Resources Investigations Report 88-4102. 68 pp.
Simpson, J.H. 1876. Report of explorations across the Great Basin of the Territory of Utah for a direct wagon-route from Camp Floyd to Genoa, in Carson Valley in 1859 (Reissued by University of Nevada Press, 1983). 518 pp.
Skorupa, J.P., S.P. Morman, and J.S. Sefchick-Edwards. 1996. Guidelines for interpreting selenium exposures of biota associated with nonmarine aquatic habitats. U.S. Fish and Wildlife Service, Sacramento, California.
_____' and H.M. Ohlendorf. 1991. Contaminants in drainage water and avian risk thresholds. Pages 345-368 *in* A. Dinar and D. Zilberman (eds.) The economics and management of water and drainage in agriculture. Kluwer Academic Publishers, Boston, Mass.
_____, , and RL. Hothem. 1992. Interpretive guidelines for field studies of selenium-exposed waterbirds. U.S. Fish and Wildlife Service, Davis, Calif. Unpub!. manuscript.
Smith, GJ., and V.P. Anders. 1989. Toxic effects of boron on mallard reproduction. Environ. Toxico!. Chern. 8:943-950.
Snarski, V.M., and G.F. Olson. 1982. Chronic toxicity and bioaccumulation of mercuric chloride in fathead minnow *(Pimephales promelas)*. Aquatic Toxico!. 2:143-156.
Spann, J.W., RG. Heath, J.F. Kreitzer, and L.N. Locke. 1972. Ethyl mercury p-toluene sulfonanilide: Lethal and reproductive effects on pheasants. Science 175:328-331.
Sparling, D.W. 1990. Acid precipitation and food quality: Inhibition of growth and survival in black ducks and mallards by dietary aluminum, calcium, and phosphorus. Arch. Environ. Contam. Toxico!. 19:457-463.
_____. 1995. Acid deposition: A review of biological effects. Pages 301-329 *in* DJ. Hoffman, B.A. Rattner, G.A. Burton Jr., and 1. Cairns Jr. (eds.). Handbook of Ecotoxicology. Lewis Publishers, Boca Raton, Florida.
Stahl, J.L., M.E. Cook, M.L. Sunde, and J.L. Greger. 1989. Enhanced humoral immunity in progeny chicks fed practical diets supplemented with zinc. Applied Agricultural Res. 4:86-89. (reference not seen).

Stanley, T.R Jr., J.W. Spann, G.J. Smith, and R Rosscoe. 1994. Main and interactive effects of arsenic and selenium on mallard reproduction and duckling growth and survival. Arch. Environ. Contam. Toxico!. 26:444-451.

Thompson, S.P., and K.L. Merritt. 1988. Western Nevada wetlands - history and current status. Pages 40-45 in RE. Blesse and P. Goin (eds.). Nevada Public Affairs Review, University of Nevada, Reno.

U.S. Bureau of Reclamation. 1990. Newlands Project Water Measurement, Management and Conservation Program 1990 Annual Report. U.S. Bureau of Reclamation Mid Pacific Region, Carson City, Nevada.

U.S. Environmental Protection Agency. 1980a. Ambient water quality criteria for cadmium. EPA 440/5-80-025.

_____ 1980b. Ambient water quality criteria for mercury. EPA 440/5-80-058.

_____ 1985a. Ambient water quality criteria for arsenic-1984. EPA 440/5-84-033.

_____ 1985b. Ambient water quality criteria for lead - 1984. EPA 440/5-84-027.

_____ 1987. Ambient water quality criteria for zinc - 1987. EPA 440/5-87-003.

_____ 1988. Ambient water quality criteria for aluminum - 1988. EPA440/5-86-008.

_____ 1989. Assessing human health risks from chemically contaminated fish and shellfish: A guidance manual. EPA 503/8-89-002.

U.S. Fish and Wildlife Service. 1995. Draft Environmental Impact Statement, Water Rights Acquisiton for Lahontan Valley Wetlands, Churchill County, Nevada. U.S. Department of the Interior, Fish and Wildlife Service, Region 1, Portland, Oregon.

White, D.H., and M.P. Dieter. 1978. Effects of dietary vanadium in mallard ducks. J. Toxicol. Environ. Health 4:43-50.

_____, and M.T. Finley. 1978. Uptake and retention of dietary cadmium in mallard ducks. Environ. Res. 17:53-59.

Woodward, D.F., A.M. Farag, E.E. Little, B. Steadman, and R Yancik. 1991. Sensitivity of greenback cutthroat trout to acidic pH and elevated aluminum. Trans. Am. Fish. Soc. 120:34-42.

Wong, P.T.S., Y.K. Chau, O. Kramar, and G.A. Bengert. 1981. Accumulation and depuration of tetramethyllead by rainbow trout. Water Res. 15:621-625.