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Assessment of Goshen Warm Springs as a Potential Hatchery Site For June suckers (*Chasmistes liorus*) Focusing on Selenium Bioaccumulation and Fish Growth Rates

Final Report To
Utah Reclamation Mitigation and Conservation Commission

U.S. Geological Survey
Columbia Environmental Research Center

March 2003

U.S. Department of the Interior
U.S. Geological Survey

Cover photo of June suckers courtesy of Jameson Weston, Utah’s Hogle Zoo

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Final Report

To

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Executive Summary

A study was conducted to determine the feasibility of establishing an interim hatchery and rearing facility for the propagation of the endangered June sucker (*Chasmistes liorus*) at Goshen Warm Springs, Utah. The study had three objectives: 1) determine the growth rates of juvenile June sucker in Goshen Warm Springs and Utah Lake; 2) determine the uptake rates of selenium in juvenile June sucker in Goshen Warm Springs and Utah Lake; and 3) evaluate the limnological factors that influence growth and selenium accumulation in juvenile June sucker in Goshen Warm Springs and Utah Lake.

Juvenile June suckers were placed in cages at three locations (July 11, 2001): Goshen Warm Springs - Lily Pond; Goshen Warm Springs - North Pond, and Provo Bay, Utah Lake. Four cages constructed of plastic mesh were anchored to the sediment to give fish free access to sediment and to allow constant water and plankton movement. Juvenile suckers were fed a commercial diet twice daily. Fish also had access to natural sources of zooplankton. Selenium concentrations in June sucker, water, sediment, and zooplankton were determined monthly. Growth, survival, and selenium uptake were determined in fish on a biweekly to monthly basis over 87 d. Water quality (i.e., dissolved oxygen, pH, hardness, alkalinity, conductivity, chlorophyll a, ammonia, and particulate organic carbon) was monitored monthly. Temperature was recorded hourly. After 87 d (October 16, 2001), the fish were transferred to the U.S. Geological Survey, Columbia Environmental Research Center, Columbia, Missouri for selenium depuration.

Fish growth was significant at all sites. June suckers were significantly larger at Utah Lake and North Pond compared to Lily Pond. Low levels of dissolved oxygen may have affected growth of June suckers at Lily Pond.

Selenium uptake in June sucker was statistically significant at Lily Pond (0.005 μg/g/d), North Pond (0.010 μg/g/d) and Utah Lake (0.003 μg/g/d). Average whole-body selenium concentration in June suckers at day 0 was 1.13 μg/g dry weight. Final average whole-body concentrations of selenium in June sucker at the end of the 87-d uptake phase of the study were 1.62 (43% increase), 1.90 (68% increase), and 1.32 (17% increase) μg/g dry weight in Lily Pond, North Pond, and Utah Lake, respectively. Final concentrations were approximately 35% of selenium concentrations known to cause toxicity.

Significant selenium depuration occurred following transfer of June sucker to clean water. Whole-body selenium concentrations in June sucker declined over the 57-d depuration period to 1.50 (7% decrease), 1.18 (38% decrease), and 1.17 (11% decrease) μg/g dry weight in fish from Lily Pond, North Pond, and Utah Lake, respectively.

Results indicated that selenium concentrations in June suckers increased significantly at all sites. However, these concentrations are not likely to be of concern since they are not at levels known to cause acute or chronic toxicity. Depuration experiments indicated juvenile June sucker reared at Goshen Warm Springs and stocked into Utah Lake would reduce bioaccumulated selenium with three months; therefore if fish were stocked into Utah Lake at 360 d, and recruited to reproductive age in Utah Lake, their reproduction should not be impacted by selenium. Anecdotal evidence suggests that the commercial diet fed to fish in Goshen Warm Springs is not optimum for growth of juvenile June sucker. Fish appeared to grow rapidly in Utah Lake due to an abundance of natural plankton even though temperature fluctuations were greater. Further evaluation of Utah Lake as an interim rearing site for June sucker should be considered.

Introduction

The June sucker (*Chasmistes liorus*) was federally listed as endangered in 1986, and is endemic to Utah Lake, Utah and the Provo River (USFWS 1999). Utah Lake covers approximately 38,400 ha and is located in a sedimentary drainage basin. It is highly eutrophic due to nutrient inputs from agriculture and industry. Utah Lake is relatively shallow (average depth of 2.8 m; maximum depth of 4.2 m); frequent winds prevent stratification and contribute to the high turbidity of the lake. Blue-green algal blooms are common in late summer. The lake typically loses 50% of its surface area in the summer due to evaporation.

The June sucker is a long-lived (up to 45 years) lake-dwelling planktivore that uses the Provo River as its primary spawning habitat. June sucker become reproductively mature between 5 - 10 years (Belk 1998). Total length at maturation ranges between 440 - 490 mm (Shirley 1983). June sucker can grow to total lengths of greater than 600 mm (Whitney and Belk 2000). The June sucker provided an important commercial fishery in Utah Lake until the 1950's when common carp (*Cypinus carpio*) increased in number and largely displaced the June sucker (USFWS 1999). June sucker populations have declined substantially since 1950 due to numerous factors including flow alteration, channelization, water quality degradation, predation, and competition with non-native invasive fish (USFWS 1999, Whitney and Belk 2000). Collectively, these factors have resulted in reduced recruitment of June suckers, which is further exacerbated by the small native range of the species.

Refugia populations of June sucker have been established at several sites, including the Fisheries Experiment Station (FES) in Logan, Utah, which serves as the interim hatchery for spawning and rearing June suckers. Similar programs have been established for other fish species, which has limited available culture facilities at FES for the spawning and

rearing June suckers. Thus, alternative hatchery sites are being sought. Goshen Warm Springs, located in the Utah Lake watershed south of Provo, Utah, has been identified as a potential site for a warm-water hatchery for June sucker. This location is advantageous for two reasons: 1) it contains a thermal warm spring that maintains a year-round temperature near 21 °C, thus potentially offering a bioenergetic advantage for rapid growth, and 2) it is located within the historic range of the species.

Previous studies (FishPro 1996, SWCA 1999, FishPro 2000) have indicated that selenium concentrations in water at Goshen Warm Springs approach the current Utah water quality criterion for selenium of 5 μg/L (UDEQ 2003). Selenium is a metalloid that is widely distributed across the western United States. Soils in northeast Utah are seleniferous because the area lies above a geological formation comprised of Mancos shale, which typically has selenium concentrations of approximately 1.1 mg/kg (Stephens et al. 1992). Coal-fired power plants and industrial point sources also serve as significant sources of selenium to the environment (DeForest et al. 1999).

Selenium is found in many forms, both inorganic and organic, in the environment. The organic forms of selenium, such as selenomethionine and selenocysteine, are the primary forms of ecological concern because they are known to be substituted into protein matrices and enzymes that have altered biological function (Lemly 1995). Various biological, physical and chemical processes can influence the state of selenium in the environment (Lemly and Smith 1987). Lentic conditions tend to promote the production of the more toxic organoselenium forms because of higher productivity and lower flushing rates (Lemly and Smith 1987, DeForest et al. 1999). As a result, selenium is more bioavailable in lentic systems due to increased efficiency of trophic transfer of selenium between water, sediments, invertebrates, and fish (Lemly and Smith 1987, DeForest et al. 1999). Food-borne selenium,

in the form of selenium-contaminated zooplankton, macroinvertebrates and forage fish in lentic systems, represent the primary ecotoxicological concern for higher trophic-level consumers such as fish and birds.

The U.S. Environmental Protection Agency (USEPA) is currently considering a revision of the existing water quality criterion for selenium, which may recommend the use of tissue-based criterion in addition to the traditional water-based concentration (Hamilton and Palace 2001, Hamilton 2002). The USEPA proposed whole-body criterion for fish tissue is 7.9 μg/g dry weight (USEPA 2002). The pending revision of selenium criteria has led to several re-evaluations of the ecotoxicological database for selenium regarding thresholds of concern for various physical and biological matrices including tissue, food, sediment, and water. Although new criteria for selenium have not been finalized, there is a range of proposed tissue thresholds (Lemly 1993, DeForest et al. 1999) for applied risk assessment of the risk of growing June sucker in Goshen Warm Springs for ultimate stocking in Utah Lake.

In this study, we evaluate the concentrations and dynamics of selenium in Goshen Warm Springs and Utah Lake in order to determine the feasibility of establishing an interim hatchery and rearing facility that minimizes the ecotoxicological risks of selenium. The study had three objectives: 1) determine the growth rates of juvenile June sucker in Goshen Warm Springs and Utah Lake; 2) determine the uptake rates of selenium in juvenile June sucker in Goshen Warm Springs and Utah Lake in relation to proposed selenium thresholds; and 3) evaluate the limnological factors that influence growth and selenium accumulation in June sucker in Goshen Warm Springs and Utah Lake.

Materials and Methods

Experimental Design

Two sites in Goshen Warm Springs were selected below two major seep inflows (Figure 1). North Pond (approximately 77 x 29 m; GPS coordinates 39° 57’ 20.44” N, 111° 51’ 23.97” W) was located at the northern end of the warm springs complex. It had little vegetation. Lily Pond (approximately 30 x 23 m; GPS coordinates 39° 57’ 11.65” N, 111° 51’ 22.67” W) was located at the southern end, where additional inflow enters the warm springs complex. Lily Pond was shallower than North Pond and was extensively covered with water lilies (*Nymphaea sp.*). Both sites had high water clarity and similar substrates, although Lily Pond had a large number of snail shells mixed into the sediment. The third site was selected in Provo Bay of Utah Lake (GPS coordinates 40° 11’ 55.57” N, 41° 43’ 7.29” W) because June sucker have historically used this area. Utah Lake had high turbidity and fine substrate. Water depth at the start of the study was similar to that in North Pond, however, as the study progressed, it declined significantly. June suckers (approximately 360-d old; average length = 49.2 mm; average weight = 0.82 g) were obtained from the FES, Logan, Utah (Lot Number 000601SKJNPR07).

Three treatments were examined: North Pond (downstream of inflow), Lily Pond (downstream of additional inflow), and Provo Bay, Utah Lake. All three locations had identical cages with access to sediment and received the same commercially prepared feed. Each treatment was replicated four times, with each cage considered to be an experimental unit. Cages were constructed with 91.4-cm (36") fiberglass rings (top and bottom) and two [0.33-cm (1/8") and 0.63-cm (¼")] plastic meshes. Cage height was approximately 91.4 cm (36"). The top panel had a 61-cm (24") door. The cages on the substrate were anchored using PVC pipe to give fish access to sediment and benthic invertebrates.

Fish were initially fed BioKyowa® Fry Feed Kyowa B, FFKB-700 (Lot Number 0040211) (BioKyowa, Inc., Chesterfield, MO). They were switched to BioKyowa® Fry Feed Kyowa C, FFKC-1000 (Lot Number 147060) during the last month of the study. Food was delivered using a Koi® Café Feeder® (Sweeney Inc., Beeville, TX). Fish were fed twice daily (0800 and 1400 hr) at a minimal rate of 5% of estimated body weight per day.

Field Sampling

One 10-g sample of each food type was collected at the beginning of the study (day 0) for selenium analysis. A second sample of the FFKB-700 diet was taken on day 14. Samples were placed in scintillation vials and kept frozen until analysis.

June suckers were collected from each cage using a clean (e.g., soap-washed and distilled water rinsed) dipnet. Fish were euthanized with tricaine methanesulfonate (TMS or MS-222). Composite samples of June suckers were taken initially to obtain a minimum 2-g wet weight for selenium analysis; a minimum 4-g wet weight for trace metal and mercury analysis, and tissue quality assurance (QA) samples (i.e., duplicates and spikes). We later determined that we could have selenium concentrations in small samples analyzed at the University of Missouri Research Reactor (MURR) by instrumental neutron activation analysis, which would allow measurement of selenium in individual fish. Therefore, early sampling events have greater number of observations per cage for whole-body selenium concentration in June suckers. Total lengths (mm) and weights (0.1 g) of fish were measured in the field. Fish were wrapped in Saran® wrap and placed in a labeled Ziplock® freezer bag. Samples were placed on dry ice and shipped overnight to the Columbia Environmental Research Center (CERC). See Tables 1 and 2 for quality a summary of quality assurance (QA) requirements and QA sampling frequencies.

Wild fish from each location were also collected to determine background concentrations of selenium and other metals in free-ranging fish populations. Wild fish were collected using either a small cast net or dipnet. Collections were made until enough fish were collected for either individual (e.g., fish > 5 g) or composite chemical analysis. Every effort was made to collect Utah chub (*Gila atraria*) and western mosquitofish (*Gambusia affinis*) at each location. However, we found no Utah chub in Utah Lake. Wild fish were processed following the same procedures as those of June suckers.

Several water quality characteristics (e.g., temperature, conductivity, salinity, pH and dissolved oxygen) were measured in situ on each date and at each site. Two 1-L grab samples were taken from each site and analyzed in the laboratory for alkalinity, hardness, turbidity, ammonia, chlorophyll a (chl-a), and particulate organic carbon (POC). Samples were put on ice and shipped overnight to CERC for analysis. A Hydrolab® DataSonde Water Quality Unit was deployed once (e.g., 24 - 48 hrs) at all sites to measure diurnal cycles of water quality (e.g., temperature, conductivity, pH and dissolved oxygen) to determine if any parameter could limit growth or survival of June suckers. Temperature was also measured hourly at each site using Onset® Tidbits (Bourne, MA).

Samples for POC and chl-a were filtered using a 47-mm A/E glass fiber filters within 24-hr of collection. Water samples for chl-a were extracted in 90% buffered acetone and analyzed by fluorometry within 48-hr of collection (APHA 1998). After filtration, POC samples were frozen until analysis. We analyzed for POC using a Coulometrics® Model 5020 Total Carbon Analyzer (Joliet, IL). Samples for ammonia analysis were filtered using 47-mm 0.45 μm polycarbonate filters, preserved with sulfuric acid (pH < 2), and stored at 4 °C until analysis using a Technicon® Autoanalyzer and colorimetric detection (Tarrytown,

NY). All other water quality analyses (i.e., alkalinity, hardness) were conducted within 48-hr of collection (APHA 1998).

Two grab samples of water for selenium analysis were taken at each site using pre-cleaned 3.78-L (1-gal) carboys. One filtered sample (0.45-μm polypropylene filter) and one unfiltered sample (i.e., total) were collected from each carboy for selenium analysis. Samples were placed on ice, shipped overnight to CERC, and acidified to pH < 2 with Ultrex® nitric acid. Reagent container blanks were created at the time of sample acidification.

Duplicate sediment samples (i.e., two samples) for selenium analysis were collected monthly from each site using a clean (e.g., soapy-washed and distilled water rinse) plastic scoop. Samples were collected from the top 10 cm of sediment. Several scoops were composited for each sample in a pre-cleaned washtub, and placed in pre-cleaned glass jars. Samples were placed on ice and shipped overnight to CERC. An aliquot of each sediment sample was collected for characterization for particle size (Bouyoucos hydrometer method, ASTM 1963) and carbon analysis (Coulometric® Model 5020 Carbon Analyzer, Joliet, IL).

Three replicate zooplankton samples for selenium analysis were collected monthly at each site using modified 2-L funnel traps. Samplers were placed approximately 0.5 m above the sediment surface, deployed overnight, and retrieved the morning of the sampling event. Water was poured through a 63-μm plankton net to concentrate plankton. In September and October (i.e., days 55 - 87), water levels in Utah Lake were too low to use the zooplankton traps, so we sampled zooplankton by towing the zooplankton net (6-m tow) just below the water surface. Zooplankton was rinsed into glass jars using distilled (DI) water. Jars were shipped on ice overnight to CERC.

Chemical Analysis

Whole-body fish were minced with either a meat cleaver or titanium knives in a polypropylene containment tray; sub-samples were placed in a crystallizing dish and immediately lyophilized. Percent moisture was determined during the lyophilization process. Once dried, the fish material was ground into a coarse powder with a Bamix® mixer/blender. Dried fish powder, fish diet and zooplankton were stored in a dessicator at room temperature until analysis.

Analysis of Selenium by Atomic Absorption

Selenium was quantitatively determined in water, sediment, zooplankton, fish and prepared diet material with 20 ml or 0.25 - 0.5 g of each dried sample subjected to a nitric acid-magnesium nitrate dry-ashing procedure (Brumbaugh and Walther 1989) as preparation for selenium determination by hydride generation. The procedure consisted of three steps: boiling with nitric acid for solubilization and partial oxidation; 500 °C ashing with magnesium nitrate to complete the oxidation and decompose remaining organic matter, and heating with hydrochloric acid to dissolve the ash and reduce selenate to selenite oxidation state required for hydride generation. Following reduction, digestates were diluted to ~100 mL with deionized water, yielding a final acid matrix of 10% HCl. The determination of selenium in all digestates was accomplished by hydride generation atomic absorption spectroscopy with flow injection. The digestate was mixed with a hydrochloric acid carrier solution and then reduced using sodium tetrahydridoborate that had been stabilized with sodium hydroxide. The resulting volatile hydrogen selenide was transferred with argon carrier gas into a heated quartz cell mounted on an atomic absorption spectrophotometer for decomposition and measurement.

Analysis of Selenium by Neutron Activation

Fish and zooplankton samples having limited biomass (< 0.5 g) and aliquots of some of the larger fish and zooplankton samples were prepared for instrumental neutron activation analysis (INAA). Aliquots (50 mg - 500 mg) of the larger samples or the entire biomass of smaller samples were transferred into a small 1.5-ml HDPE (high-density polyethylene) vial provided by the MURR staff. The sample was positioned and pressed flat against the bottom of the vial with a cleaned glass rod. Several aliquots of each of three additional certified reference tissues, IRMM CRM 414 Plankton, IRMM CRM 422 Cod Muscle, and NRCC DORM-2 Dogfish Muscle were prepared for analysis with each set of samples. All vials were left open and placed in the tray chamber of a Virtis Genesis 35EL lyophilizer in "shelf" control and frozen to -75 °C. Once a condenser temperature of -70 °C and vacuum of 300 mTorr was reached, the drying cycle commenced. Samples were lyophilized to a constant weight. Lyophilization greatly reduces ^{19}O in the irradiated sample, and significantly enhances measurement precision. Upon recording of final sample weight, an expandable cleaned polyethylene plug was inserted into the vial against and the vial lid was compressed shut. All samples were transported to MURR for the determination of selenium as the radionuclide ^{77m}Se.

Standards in the range of 0.01 - 5 μg selenium were prepared by pipetting appropriate quantities from a series of selenium stock solutions onto filter pulp paper. The pulp paper was then placed in the bottom of HDPE vials in a comparable geometric configuration to that of the samples. NIST Standard Reference Material (SRM) 1577 Bovine Liver was also analyzed as MURR internal quality control sub-samples. Selenium was analyzed using instrumental neutron activation analysis as the ^{77m}Se nuclide. The ^{77m}Se nuclide is produced from the activation reaction of ^{76}Se (i.e., ^{77m}Se is the product nuclide resulting from neutron

capture). The ^{77m}Se nuclide has a half-life of 17.4 seconds. The activation and decay reactions follow: $^{76}Se_{34}\ ^{1}n_{0} \rightarrow {}^{77m}Se_{34} + \Upsilon$prompt and $^{77m}Se_{34} + {}^{76}Se_{34} + \Upsilon$delayed ($T_{1/2}$ = 17.4 s). It is advantageous to use ^{76}Se with biological samples where much of the bulk matrix is carbon, hydrogen and oxygen, because it can be analyzed very rapidly by looking at the product nuclide ^{77m}Se. In contrast, ^{75}Se would be used for samples with matrices having high concentrations of inorganic constituents (e.g., geological samples), where the half-life is 120 d and long decay times can be used prior to counting, thus avoiding interferences from the matrix. Each standard or sample was placed in the top-center position of a shuttle rabbit and irradiated for five seconds in the Row I position using the pneumatic-tube irradiation facility at MURR. This position has thermal and epithermal neutron flux densities of 8 x 10^{13}n x cm^{-2} x sec^{-1} and 2 x 10^{12}n x cm^{-2} x sec^{-1}. The pneumatic transfer facility employed has a delivery time to the counting station of about four seconds. The returned shuttle rabbit was quickly opened and the sample vial transferred to a special holder that positions the small HDPE vial on the face of the detector. All 25-sec real-time counts were analyzed using a high-resolution gamma-ray spectrometer. The gamma-ray spectrometer included a Tennelec 244 Amplifier coupled to a Nuclear Data 599 Loss-Free Counting Module and a Nuclear Data 581 ADC. Data acquisition and peak extraction were done using a VAX Station 3100, Model 38 with Canberra/ND Application Software. The 161.9 keV gamma-ray from the decay of ^{77m}Se was used to determine selenium concentrations by standard comparison.

Analysis of Metals

A homogenized aliquant (0.25 g) was prepared from composite fish samples and from one individual chub sample from Lily Pond. Each aliquant was heated with 6 ml of HNO_3 in a sealed microwave Teflon® vessel. The digestate liquid was transferred into a 125-ml polyethylene bottle with ultrapure water (> 10 megOhm/cm) to a final weight of 101.5 g (100

ml). A portion of the digestate (30 ml) was transferred to a glass tube to which 0.3 ml HCl was added, giving a final acid matrix of 6% HNO_3-1% HCl for subsequent mercury determination. The 70-ml portion of the digestates was analyzed by inductively-coupled plasma-mass spectrometry (ICP-MS) using a semi-quantitative scan. This scanning mode has a manufacturer's reported accuracy of ± 30 - 50%. Fish digestates were diluted 10X by CETAC ASD-500 Autodiluter as part of the analytical sequence. Internal standards were scandium (10 ppb), rhodium (10 ppb) and thorium (10 ppb), and the external standard consisted of a NIST traceable reference solution (Trace Metals in Drinking Water; High Purity Standards, Charleston, SC) to which five elements (praseodymium, terbium, thulium, tantalum, and gold) were added for improved calibration in the rare earth region of the mass spectral range.

Mercury in composite fish samples was determined by flow injection cold vapor atomic absorption spectroscopy. Mercury vapor was produced by reacting the sample digestate with stannous chloride, with the vapor then being swept into a ~100 ^{o}C quartz cell interfaced to an atomic absorption spectrometer for measurement.

Quality Assurance Calculations

Procedures for calculating QC statistics are as follows:

Percent Relative Standard Deviation (%RSD) = SD/Mean x 100

Relative Percent Difference or %RPD = (D1-D2)/Mean x 100

Percent Spike Recovery = (Total Measured – Background)/Spike Amount x 100

Method Limit of Detection = $3 \times (SD_b^2 + SD_s^2)^{1/2}$ where

SD_b = standard deviation of a blank or low level standard and

SD_s = standard deviation of a low level sample.

Selenium Thresholds

Toxicity thresholds for selenium in the environment have been proposed because of its availability to organisms and its potential for harm to aquatic organisms. We have selected thresholds proposed by Lemly (1993) and DeForest et al. (1999) to assess the potential for selenium toxicity at our study sites. Both reviews suggest that a tissue-based threshold is more appropriate for selenium assessments than water or sediment thresholds due to the potential for selenium to bioaccumulate. Tissue-based analyses are the emerging toxicological application for selenium because it integrates such factors as exposure, metabolism, dietary uptake, and equalibrium dynamics (Hamilton 2002). We, however, will present thresholds for four matrices (e.g., water, sediment, invertebrates, and fish) in order that a complete review of our data can be made.

Lemly (1993, 1995) synthesized toxicity, bioaccumulation and sediment data from both field and laboratory studies in developing toxic effect thresholds for selenium in water, food-chain organisms, and fish tissues. Studies were not limited to dietary exposures, and included studies that examined the teratogenic effects of selenium in natural populations, and residues in food-chain organisms and fish. The most important aspect of residue analysis is that selenium in fish tissues results from dietary selenium not waterborne selenium.

To develop their proposed thresholds, DeForest et al. (1999) reviewed literature that contained laboratory, mesocosm, and field studies, as well as USEPA guidelines, to develop water quality thresholds of concern. They used only data that contained dietary exposures or maternal transfer of selenium (i.e., no data that had water-only exposures); population effects (i.e., reproduction, survival, growth, teratogenesis); geometric means for chronic values of no observable effect concentrations (NOEC) and lowest observable effect concentrations

(LOEC), and estimating EC_{10}, EC_{20} and EC_{50} values where possible. The review was divided into warmwater and coldwater species. We report warmwater thresholds.

Statistical Treatment

Data were analyzed using the Statistical Analysis System (SAS 2000) using a significance level of $p \leq 0.05$ (Snedecor and Cochran 1969). Data were tested for normality prior to statistical comparisons using the Proc Univariate procedure and the Shapiro-Wilk's Statistic. Growth rates, selenium uptake rates and selenium depuration rates of June suckers in each cage were determined using linear regression. The data were analyzed as a split plot in time. The linear statistical model contained the effect of site, cage within site, time, and the site times time interaction. Cage within site was used as the mean square error in the F test to test the effects of site. The residual mean square was used to test time and the interaction of site times time. Fisher's least significant difference (LSD) was used to determine differences between means. Polynomial contrasts were performed to test differences between site response over time.

Results

Growth

Average total length of June suckers was 49 mm at the beginning of our study (i.e., day 0) (Table 3, Figure 2). The average total length of June suckers held at FES was 53 mm or an increase of 7% (E. Hansen, personal communication) at 87 d of the study (i.e., end of the uptake phase). Average total lengths of June suckers in Lily Pond, North Pond and Utah Lake were 62 (27% increase), 75 (52% increase), 79 (61% increase) mm, respectively, at the

end of the uptake phase of the study. Average total lengths of fish at the end of the depuration phase of the study (i.e., 144 d) were 61 (24% increase), 90 (82% increase), and 94 (92% increase) mm for fish exposed in Lily Pond, North Pond, and Utah Lake, respectively (Table 4), while the average total length of June suckers at FES was 67 (27% increase) mm. Final lengths of June suckers from Lily Pond did not increase during the depuration phase of the study. Final average lengths of June suckers from all sites were significantly greater than at day 0 ($p < 0.01$) at the end of the study, and were significantly greater at Utah Lake and North Pond ($p < 0.01$) than Lily Pond. Our data may be biased by the small sample sizes at day 144, however, these data indicate that bioenergetic conditions in our cage study exceeded those under current hatchery conditions.

Average weight of June suckers was 0.82 g at the beginning of the study (Table 3, Figure 3). Fish grew to average weights of 2.38 (190% increase), 5.03 (513% increase), and 4.03 (391% increase) g after 87 d in Lily Pond, North Pond and Utah Lake, respectively. Average weight of June suckers at the end of the depuration phase of the study (day 144) were 1.25 (52% increase), 7.20 (778% increase), 7.39 (801% increase) g in Lily Pond, North Pond and Utah Lake, respectively (Table 4). Final average weights of June suckers at all sites were significantly greater than at day 0 ($p < 0.01$), and were significantly greater at Utah Lake and North Pond ($p < 0.01$) than Lily Pond. Weights of June suckers from Lily Pond did not increase during the depuration phase, but again, our sampling may have been biased by small sample sizes. Several June sucker from all sites were also infected with a parasite, *Learnae sp.*, which also may have decreased feeding and growth.

Growth data were modeled using three models (linear, quadratic, and cubic). We found that for all three sites, the linear model was the best fit ($p < 0.01$), although the quadratic model was significant for North Pond ($p < 0.05$) and Utah Lake ($p < 0.01$). Data

were analyzed separately by cage for all time periods (days 0, 14, 28, 55, 87, 116, 144); uptake (i.e., days 0, 14, 28, 55, 87), and depuration (i.e., days 87, 116, 144) (Table 5). Relative growth rates varied during the 87-d in-situ cage study, however most were significant in all time periods and during the uptake phase. Growth rates of June suckers at CERC were generally not significant during the depuration phase of the study when analyzed by cage.

Growth rates for each site (i.e., cages pooled) were significantly greater than zero during the uptake phase at all sites (Table 6). The pooled growth rate of June suckers from Utah Lake was significant ($p = 0.0020$) during the depuration phase of the study.

Selenium Dynamics

Selenium thresholds (Lemly 1993, DeForest et al. 1999) that we selected to be used in our assessment are listed in Table 7.

Food

The concentration of selenium in the two commercial diets ranged from 2.39 to 3.55 µg/g dry weight (Table 8). Selenium concentrations were higher in the BioKwoya FFKB-700 diet (fed days 0 - 55) than BioKyowa FFKC-1000 (fed days 55 - 144). Prepared diets contained selenium concentrations near the lower end of toxicological concern expressed by Lemly (1993), but well below those expressed by DeForest et al. (1999).

Fish

Concentrations of selenium in June suckers are presented in Tables 9 and 10, and Figure 4. Fish obtained from the hatchery had an average selenium concentration of 1.13 µg/g dry weight. At the end of the uptake phase of the study (day 87), average

concentrations of selenium in June suckers exposed in Lily and North Ponds (Goshen sites) ranged from 1.62 to 1.90 μg/g dry weight, with percent gain in selenium ranging from 43 - 68%. Average concentrations of selenium in June suckers exposed in Utah Lake were 1.32 μg/g dry weight, with percent gain in selenium of only 17%. At 87 d, there was a significantly greater concentration of selenium in whole-body June suckers at Lily ($p = 0.001$) and North Pond ($p < 0.0001$), but not Utah Lake ($p = 0.129$). Concentrations of selenium in June suckers were 1.50, 1.18, and 1.17 from Lily Pond, North Pond and Utah Lake, respectively, at the end of the depuration phase of the study (i.e., day 144). There were significantly greater concentrations of selenium in June suckers at Lily Pond than North Pond or Utah Lake ($p = 0.001$) during the depuration phase of the study. Concentration of selenium in June suckers did not approach either of the whole-body thresholds (4 or 9 μg/g dry weight) proposed by Lemly (1993) or DeForest et al. (1999).

Concentrations of selenium in whole-body wild fish (i.e., Utah chub, western mosquitofish) (range 2.62 - 8.90 μg/g dry weight) collected at the Goshen sites were considerably higher than concentrations in June suckers (Figure 5). Twelve of the fourteen wild fish analyzed were above the threshold of 4 μg/g dry weight for whole-body fish proposed by Lemly (1993), but below the threshold of 9 μg/g dry weight proposed by DeForest et al. (1999). Selenium concentrations (1.07 - 2.61 μg/g dry weight) in wild fish from Utah Lake did not approach either of the proposed thresholds.

Selenium uptake and depuration rates in June suckers were analyzed by cage for all time periods (days 0, 14, 28, 55, 87, 116, 144); the uptake phase (i.e., days 0, 14, 28, 55, 87) and the depuration phase (i.e., days 87, 116, 144) (Table 11). Rates were similar in cages within each site for all time periods and during the uptake and depuration phases of the study. We also found significant uptake rates in all sites (Table 12) when uptakes were pooled.

There was a significantly greater uptake rate in June suckers from North Pond than Lily Pond or Utah Lake ($p = 0.0004$). The depuration rates for North Pond and Utah Lake were significant ($p = 0.01$), however, there was no significant difference in depuration rates between sites. Small sample size decreased the sensitivity of statistical analyses.

Water

Concentrations of selenium in unfiltered ("total") and filtered ("dissolved") water samples from Lily Pond, North Pond, Utah Lake and CERC well water are presented in Table 13 and Figure 6. Selenium concentrations (both total and dissolved) in CERC well water were low and well below the thresholds proposed by Lemly (1993) and DeForest et al. (1999). Concentrations of total and dissolved selenium in water samples from the same location were similar for most samples, with the total concentration usually being slightly higher. Utah Lake water exhibited consistently lower selenium concentrations throughout the collection periods, with total selenium ranging from 0.52 to 1.94 μg/L.

Concentrations of selenium in all water samples from the Goshen Warm Springs sites were above the lower end of the toxicity threshold (2 - 5 μg/L) proposed by Lemly (1993), however they were below the threshold (5- 32 μg/L) proposed by DeForest et al. (1999). Most of the Utah Lake samples were below the threshold by Lemly (1993).

Sediment

Sediment from Lily and North Ponds had average selenium concentrations ranging from 2.82 to 6.30 μg/g dry weight (Table 13 and Figure 7). Concentrations of selenium in sediment from North Pond were above the threshold (4 μg/g dry weight) proposed by Lemly (1993). Selenium concentrations in sediment from Utah Lake were lower, and ranged from 1.56 to 1.75 μg/g dry weight.

Zooplankton

Zooplankton collected from Lily and North Ponds had average selenium concentrations ranging from 2.50 to 5.81 μg/g dry weight (Table 13 and Figure 8). Zooplankton samples from Utah Lake had lower average concentrations of selenium, ranging from 1.71 to 3.53 μg/g dry weight. Selenium concentrations in zooplankton from all three sites remained relatively stable over the 87-d study. There were zooplankton samples from each site that were above the selenium threshold for zooplankton (3 μg/g dry weight) proposed by Lemly (1993), but only one was greater than the threshold (10 μg/g dry weight) proposed by DeForest et al. (1999).

Trophic Relationships in Bioaccumulation

We compared concentrations of selenium in water, sediment, food-chain organisms (i.e., zooplankton) and fish (both June sucker and wild fish) across sites (Figures 9 - 14). We found that the relationship between zooplankton and wild fish ($R^2 = 0.94$) clearly presented the concept of dietary selenium uptake and increased selenium concentrations in fish tissue. The relationship between zooplankton and June suckers ($R^2 = 0.26$) was not as strong due to the influence of low selenium in the commercial diet fed to June suckers. Dissolved selenium concentrations in water were moderately correlated with zooplankton ($R^2 = 0.40$) and sediment ($R^2 = 0.59$).

Metals in Whole-body Fish Residues

Semi-quantitative metals analyses were conducted on June suckers and wild fish are presented in Table 14. Concentrations of most elements were comparable among fish samples between sites. June suckers from FES had lower levels of some elements (i.e., manganese, zinc, arsenic, rubidium, strontium, and barium). Aluminum and barium

concentrations were most elevated in western mosquitofish from Utah Lake. Selenium, rubidium, silver, and cesium concentrations were highest in Utah chub and western mosquitofish composites from Lily and North Ponds. All composites exceeded the 4 μg/g biological effects threshold for selenium (Lemly 1993, 1996).

Water Quality

Overall average temperatures in Lily Pond, North Pond and Utah Lake for the uptake phase of the study were 21 °C, 20 °C, and 16 °C, respectively (Table 15). Monthly averages showed a similar pattern, with Lily Pond having the highest average temperature and lowest variability. There were statistically significantly differences ($p < 0.001$) in temperatures between Lily and North Ponds at all time periods, however these differences were small and probably not biologically significant. Average water temperatures in Utah Lake were significantly lower (and had greatest diurnal change) ($p < 0.001$) at all time periods compared to the Goshen Warm Springs sites. At the end of the study, the average temperature in Utah Lake was also lower than the average water temperature at FES (15.6 °C). Average water temperature of CERC well water (17 °C) is cooler than sites at Goshen Warm Springs, however it is similar to overall average water temperatures at Utah Lake (16.5 °C). Lower water temperature, small numbers of fish, and different light conditions may have reduced growth of fish during the depuration phase (days 88 - 144) compared to growth rates under natural conditions.

Additional water quality data are listed in Tables 16 and 17. Site waters were well buffered (215 - 350 mg/L $CaCO_3$) and very hard (346 - 483 mg/L $CaCO_3$). Utah Lake had the highest pH, which varied between 8.1 and 8.97 diurnally. Conductivity was higher at Lily and

North Pond (2.2 mS/cm) than Utah Lake (1.4 mS/cm). Dissolved oxygen was lowest in Lily Pond (2.8 – 4.1 mg/L). Dissolved oxygen saturation was generally higher in North Pond (3.5 – 5.1 mg/L) and Utah Lake (3.3 – 12.5 mg/L). Total ammonia concentrations were low (< 0.2 mg/L as N) at all sites, except in Utah Lake on day 87, when concentrations increased to 0.89 mg/L. This sample was taken in extremely shallow water (< 10 cm). The sample may represent the ammonia concentrations that our caged fish were exposed to (since they were also in very shallow water), but ammonia concentrations in the lake were probably much closer to concentrations found earlier in the summer. Turbidity, POC, and chl-a were an order of magnitude higher in Utah Lake than Lily or North Ponds. Turbidity, POC, and chl-a declined over time in Utah Lake.

CERC well water is well buffered (256 - 259 mg/L $CaCO_3$) and hard (285 - 289 mg/L $CaCO_3$), which was similar to site waters. Turbidity (0.77 - 1.02) was similar to the Goshen Warm Springs sites. Chlorophyll a and POC concentrations at CERC were an order of magnitude lower than site waters.

Sediment Quality

Composition of sediment varied slightly between sites (Table 18 and 19). Sediments from Lily Pond were comprised mostly of sand (70%) and clay (25%). North Pond sediments were a sand (50%) and clay mixture (40%). Utah Lake sediments were finer, with a mixture of sand (10%), silt (60%) and clay (30%). Average carbon contents of sediments were similar across sites (range of TIC = 3.59 - 6.54%; range of TOC 2.79 - 6.50%) and are considered highly organic.

Discussion

Growth

Belk (1998) estimated annual total length of June sucker using otoliths collected from ten adult June suckers caught in Utah Lake after they had died from unknown causes. Estimated total length of June suckers at 360 d and 720 d were 111 mm and 232 mm, respectively.

Average total length of 360-d June suckers obtained from the FES for this study was approximately 50 mm (E. Hansen, personal communication), which is considerably shorter than those estimated for 360-d wild June suckers by Belk (1998). One of the primary reasons for building a facility at Goshen Warm Springs is its source of warmer water. The average water temperature at FES is 15.6 °C while the average summer water temperature at the Goshen Warm Springs sites during this study was 20.5 °C. The average temperature in Utah Lake during July and August was approximately 19 °C.

Our data indicate that growth rates for June suckers would be higher at all three sites than FES, however, none of the locations would produce June sucker with the target total length of 216 mm (SCWA 2002) within one growing season. At 87 d, average lengths of June suckers at Utah Lake, North Pond and Lily Pond increased 29 mm, 19 mm and 12 mm, respectively. It is unlikely that June suckers would grow at that rate throughout the year in Utah Lake because water temperatures would decline in the winter, however, if June suckers did grow at the rates found in this study, the projected lengths of June suckers after one year in Utah Lake, North Pond and Lily Pond would be 106 mm, 76 mm, and 48 mm, respectively. This projected length of June suckers in Utah Lake is close to that estimated by Belk (1998) for 360-d June suckers (111 mm), whereas the projected length of June sucker in Lily Pond is close to that obtained at FES (50 mm). Higher growth rates at North Pond (1.5X

greater than Lily Pond) did occur. Low dissolved oxygen (i.e., generally below 2 mg/L throughout the study) and poor feeding may have limited growth at Lily Pond.

A limited growing season (i.e., May - September) and declining water levels may limit the use of Utah Lake as a rearing location. We did not move fish cages as water levels dropped in the lake, which exposed June suckers to very shallow water, however, floating cages may be an alternative means of rearing fish, if culturing June suckers in Utah Lake is considered. High water temperatures (> 28 °C) may also interacted with other water quality parameters such as pH and ammonia to impact June sucker growth and survival. The Utah 30-d criterion for total ammonia at temperature = 25 °C and pH = 8.5, is 0.43mg/L as N (UDEQ 2003). Both temperature and pH values are common during the summer in Utah Lake. Total ammonia concentration in Utah Lake on day 87 exceeded the criterion, however, this sample was taken in very shallow water (< 10 cm), and probably does not represent habitats where wild June suckers would have been located. Ammonia concentrations in Utah Lake were probably significantly lower (i.e., comparable to earlier sampling periods).

Selenium

The data from this study suggest that there is a low potential for selenium to bioaccumulate at Goshen Warms Springs. We based our findings on whole-body residue analyses of June suckers which were well below published toxicity thresholds of concern (Lemly 1993, DeForest et al. 1999), despite finding selenium concentrations in some water, sediment, zooplankton and wild-fish samples which exceeded published selenium toxicity thresholds (Lemly 1993, DeForest et al. 1999). Selenium concentrations greater than the proposed thresholds may be hazardous to the health and long-term survival of fish and wildlife populations due to the high potential for food-chain bioaccumulation (Lemly 1993, DeForest et

al. 1999). The Utah 4-d average selenium criterion in water for aquatic wildlife is 5 μg/L (UDEQ 2003).

Selenium concentrations in filtered water collected from Lily and North Ponds exceeded 2 μg/L and approached 5 μg/L, both of which exceed toxicity thresholds for selenium (Lemly 1993, DeForest et al. 1999). Eight out of 16 sediment samples from Lily and North Pond exceeded 4 μg/g dry weight. Eighteen out of 24 zooplankton from the Goshen sites samples exceeded 3 μg/g dry weight, a level in food-chain organisms reported to be potentially lethal to fish and aquatic birds that consume them (Lemly 1993, Hamilton 2002). Only three out of 14 zooplankton samples collected in Utah Lake exceeded the biological effects threshold of 3 μg/g. Only one sample from Lily Pond approached the upper toxicity threshold of 9 μg/g for food-chain organisms (DeForest et al. 1999). As water levels drop, selenium in sediment is more likely to be disturbed and available for uptake by zooplankton (Lemly 1995).

Lemly (1995) presented a method to assess the potential for a selenium hazard at each site. The rating system incorporates five components (e.g., water, sediment, benthic invertebrates, fish eggs and bird eggs). The ranks for each component are summed and given a score based on the degree of hazard. High hazard denotes imminent, persistent effects; moderate hazard indicates a persistent toxic threat to impair but not eliminate reproductive success; low hazard denotes a periodic or ephemeral toxic threat that could marginally affect the reproductive success of some sensitive species, and minimal hazard indicates that no toxic threat is identified but concentrations of selenium are slightly elevated in one or more components. Although we did not analyze fish or bird eggs for selenium, the hazard risk ranged from minimal (Utah Lake) to moderate (North Pond), with Lily Pond exhibiting a low hazard.

Whole-body selenium concentrations in caged June suckers were all below the proposed toxicity thresholds proposed by Lemly (1993) and DeForest et al. (1999), as well as

the proposed USEPA whole-body fish criterion of 7.9 μg/g dry weight (USEPA 2002). Previous studies (Lemly 1982, Besser et al. 1994) have indicated that fish in this study should have reached equilibrium in whole-body selenium concentrations. However, an extended caging experiment lasting for one year, which is the anticipated rearing time that the June suckers will spend in the ponds prior to transfer to Utah Lake, might provide a better estimation of the potential for the North and Lily Ponds environments to facilitate selenium bioaccumulation in June sucker, because the relative exposure mechanism of selenium (food versus water) may vary over an annual cycle. Also, June suckers feeding on naturally occurring zooplankton in North and Lily Ponds, instead of a prepared diet, would presumably bioaccumulate selenium in a more similar fashion to that found in the indigenous planktivores such as juvenile Utah chubs.

Utah Lake has a much lower potential for selenium bioaccumulation than the Goshen Warm Spring sites, based on concentrations of selenium in water, sediment, zooplankton and indigenous fish found in this study. Since June suckers would not reach sexual maturity for 5 - 10 years after spending only one year being reared at Goshen Warm Springs, any accumulated selenium should be largely depurated or mitigated through growth dilution, with no known detrimental effects on long-term reproductive health. Coyle et al. (1993) found reduced survival in bluegill fry of adult bluegill that had been exposed to 10 μg/L water-borne selenium in combination with a dietary exposure of > 15 μg/g dry weight seleno-L-methionine. These concentrations were much higher than levels found at either of our sites.

Our assessment was made using juvenile (> 360-d old) June suckers. The age of fish may be important in determining selenium toxicity (Hamilton 2002) due to increased sensitivity during the early life stages. In addition to mortality, selenium can have sub-lethal effects, such as reduced growth rates and increases in deformities (Hamilton 2002). Factors such as

temperature, nutrition, disease and species sensitivity should all be considered in fully assessing the potential for selenium to impact fish population (Hamilton and Hoffman 2002). Providing adequate nutrition for June suckers will be critical to limiting any impact water-borne selenium at a rearing facility at Goshen Warm Springs.

Metals in Whole-body Fish Residues

Schmitt et al. (1999) summarized elemental contaminant data from the National Contaminant Biomonitoring Program (1976 - 1986), and presented means, maximum concentrations, and the 85th percentile concentrations for seven metals: arsenic, cadmium, copper, mercury, lead, selenium and zinc. Although the 85th percentile concentrations do not represent criteria, they do provide a large contaminant database to compare our metal analyses in June sucker and indigenous fish. We also evaluated tissue residues against published effect concentrations summarized by Jarvinen and Ankley (1999).

Zinc concentrations in wild fish from all sites exceeded the 85th for zinc (159 μg/g dry weight) and arsenic (1.2 μg/g dry weight). Reduced survival and growth in freshwater fish have occurred with whole-body zinc residues of approximately 360 μg/g dry weight (Jarvinen and Ankley 1999) more than twice the concentrations found in wild fish. No fish exceeded the 85th percentile for cadmium (0.20 μg/g dry weight) or mercury (0.9 μg/g dry weight). Concentrations for cadmium of 2 μg/g dry weight indicate contamination, with 5 μg/g dry weight considered life-threatening (Schmitt et al. 1999), which are two order of magnitudes higher than measured concentrations. The threshold concentration for adverse effects of methyl-mercury (typically 90% of mercury in whole fish) ranges from 3.5 - 16.5

μg/g dry weight depending on taxon and endpoints (Jarvinen and Ankley 1999), which is an order of magnitude greater than measured concentrations in this study.

A wild-fish sample from each site exceeded the 85th percentile for lead (1.05 μg/g dry weight), which is near effects concentration for heme synthesis (1 μg/g dry weight) (Schmitt et al. 1999). In addition to effects on heme synethesis, elevated whole-body lead concentrations may result in anemia, spinal deformities, and increased mucus production (Eisler 1988). Dwyer et al. (1987) found decreased erythrocyte enzyme δ-aminolevulinic acid dehydratase (ALA-D) activity and changes in vertebrate biochemical properties (i.e., decrease in phosphorous in bone) in longer sunfish with whole-body lead concentrations of 0.85 μg/g dry weight. Reduced hatchability in brook trout (*Salvelinus fontinalis*) occurred at lead whole-body residues near 2 μg/g dry weight (Jarvinen and Ankley 1999), twice the levels found in our study. Reduced growth occurred in larval and juvenile brook trout at lead whole-body residues greater than 20 μg/g dry weight (Jarvinen and Ankley 1999), an order of magnitude greater than residues in this study. Absorption and retention of lead is a function of age, sex and diet (Eisler 1988). Dietary deficiencies of calcium, zinc, iron, vitamin E, thiamin and magnesium may enhance the toxic effects of lead (Eisler 1988).

Copper concentrations in June suckers exceeded the 85th percentile for copper (8.5 μg/g dry weight). Wild fish from all sites were below the 85th percentile, except for one composite sample of western mosquitofish that exceeded the 85th percentile by four times. Dietary and waterborne exposures both contribute to accumulation of copper in fish tissue. Reduced survival in larval carp (*Cyprinus carpio*) with a whole-body residue of approximately 55 μg/g dry weight (Jarvinen and Ankley 1999).

Selenium concentrations in June suckers were all below the 85th percentile (3.3 μg/g dry weight), however wild fish from Lily and North Ponds exceeded the 85th percentile, and the selenium toxicity threshold for whole-body fish proposed by Lemly (1995).

Collectively, these data indicate that metal concentrations in caged June suckers and wild fish occasionally exceed the 85th percentile concentrations for metals (Schmitt et al. 1999), but they are not at levels to effect growth and survival.

Water Quality

The Goshen Warm Springs sites barely met the criterion for average water temperature of 21 °C (FishPro 2000) during the 87-d study. The average temperature in Utah Lake (17 °C) during the study did not meet that criterion but still resulted in robust growth.

Alkalinity (200 mg/L) and hardness (300 mg/L) criteria set for construction of the hatchery were exceeded at all sites (FishPro 2000). Alkaline conditions do contribute to higher concentrations of selenium in solution (May et al. 2001). Because water quality criteria for several metals (i.e., cadmium, copper, lead, nickel and zinc) are based on hardness (UDEQ 2003), greater water hardness from all sites should be more protective against metal toxicity. For example, lead is less toxic with increasing hardness because lead becomes less bioavailable due to precipitation.

Previous studies (Finger et al. 1994, Hamilton et al. 2002) have shown an inconsistent correlation of several water quality variables (e.g., conductivity, hardness, calcium, magnesium, chloride and sulfate) and selenium concentrations. Several water quality parameters such as particulate organic carbon and chlorophyll a may also influence selenium concentrations and growth rates in zooplankton and June suckers. Although zooplankton in Utah Lake had relatively low concentrations of selenium, the potential for bioaccumulation

may still be high because of the large number of zooplankton available to June suckers. The ability to depurate selenium is a function of chronic exposure of selenium (Coyle et al. 1993), thus if June suckers were feeding continuously on a food source with low levels of selenium, as well as being exposed to selenium in water and sediment, depuration may be slowed. Additional research into the annual cycle of selenium concentrations in zooplankton populations would provide a more complete assessment of the likelihood of food-chain transfer of selenium in Utah Lake.

Sediment Quality

Although there is little support for sediment-based criterion for selenium (DeForest et al. 1999, Hamilton 2002), sediments do act as "sinks" (Lemly and Smith 1987) and are thus a long-term source of selenium in aquatic systems. Selenium is removed from solution and sequestered by adsorption onto clay and organic carbon particles (Lemly and Smith 1987). A large percentage of sediments from all three sites were made up of fine particles (i.e., silt and clay), and were highly organic, which aid in removing selenium from solution and then sequestering it in the sediment. Indeed, all of the sediment samples collected from North Pond had selenium concentrations above the sediment threshold proposed by Lemly (1993). However, these high levels of selenium in sediment did not result in high concentrations of selenium in June suckers.

Mobilization of selenium from sediments is also likely to occur due to biological activity (i.e., microbial transformation), and physical perturbation (i.e., wind and wave action). It is expected that the hatchery and rearing facility will include hardened raceways or ponds that would largely remove the potential of dietary selenium by the ingestion of sediment. In

addition, our studies did not find a strong relationship between sediment and fish concentrations of selenium.

Future Research Needs

Additional research information is needed regarding the propagation and recovery of June sucker populations. Data from this study indicated that the commercial BioKyowa® diet is not an optimum diet for June sucker based on weight gains. This may be due to nutritional inadequacy of the diet or non-preference by June sucker. Additional research regarding better dietary formations is needed.

Growth rates of June sucker in Utah Lake indicated that cage-culture of juvenile fish may be an alternative approach for hatchery propagation of June sucker for population restoration. Our study was isolated to one part of the lake. Other locations may contain better zooplankton food sources and water quality conditions that could further enhance growth. In addition, many species of fish show differential growth rates under various stocking densities. Frequently, growth rates increase with increased density as social interactions that lead to high-energy expenditure are reduced. Thus, further evaluation of cage culture in Utah Lake is warranted.

There is little published information regarding selenium or other metal concentrations in wild adult June suckers in Utah Lake. The use of muscle plugs (Waddell and May 1995) would provide a fairly non-invasive approach to determining the biological significance of selenium and other metals in adult June suckers. Muscle plugs would have significant advantages compared to water or food analysis since it is very difficult to quantify different forms (both inorganic and organic) of selenium in the environment (Besser et al. 1993). Additional research into the availability of selenium compounds in Goshen Warms Springs

and Utah Lake would be valuable in assessing the impact of selenium on June sucker populations.

Lastly, there is considerable concern regarding water quality conditions of Utah Lake. Utah Lake is native habitat for June sucker, and is ultimately where stocking/restoration efforts are directed. Additional research is needed to examine historic, current, and future water quality conditions to ensure the success of June sucker restoration efforts.

Conclusions

Fish growth was significantly greater at Utah Lake and North Pond compared to Lily Pond. Decreased growth of fish at Lily Pond may have resulted from low levels of dissolved oxygen. Selenium uptake in June sucker was statistically significant at all sites. Significant selenium depuration occurred following transfer of June Sucker to clean water.

Although our results indicated that selenium was bioaccumulated to statistically significant levels at both Goshen Warm Springs locations, these concentrations are not likely to be of concern since they are not at levels known to cause chronic toxicity. Depuration experiments indicated juvenile June sucker stocked into Utah Lake would reduce bioaccumulated selenium within three months; therefore, if fish were stocked into Utah Lake between 360 - 720 d, and recruited to reproductive age in Utah Lake, they should not be impacted by selenium accumulated during rearing. Anecdotal evidence suggests that the commercial diet fed to fish in Goshen Warm Springs is not optimum for growth of juvenile June sucker. Fish appeared to grow rapidly in Utah Lake due to an abundance of natural plankton even though temperature fluctuated. Further evaluation of Utah Lake as an interim-rearing site for June sucker should be considered.

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Literature Cited

APHA (American Public Health Association), American Water Works Association and Water Environment Federation. 1998. Standard methods for the examination of water and wastewater. 20th edition. APHA, Washington, D.C.

ASTM (American Society for Testing and Materials). 1963. Standard method for particle size analysis of soil. ASTM document designation D422-63 (Reapproved 1972). ASTM, Philadelphia, Pennsylvania.

Belk, M.C. 1998. Age and growth of June sucker (*Chasmistes liorus*) from otoliths. Great Basin Naturalist 58:390-392.

Besser, J.B., T.J. Canfield, and T.W. La Point. 1993. Bioaccumulation of organic and inorganic selenium in a laboratory food chain. Environmental Toxicology and Chemistry 12:57-72.

Besser, J.B., J.N. Huckins, and R.C. Clark. 1994. Separation of selenium species released from Se-exposed algae. Chemosphere 29:771-780.

Brumbaugh, W.G., and M.J. Walther. 1989. Determination of arsenic and selenium in whole fish by continuous-flow hydride generation atomic absorption spectrophotometry. Journal of the Association of Official Analytical Chemists 72:484-486.

Coyle, J.J., D.R. Buckler, C.G. Ingersoll, J.F. Fairchild, and T.W. May. 1993. Effect of dietary selenium on the reproductive success of bluegills (*Lepomis marochirus*). Environmental Toxicology and Chemistry 12:551-565.

DeForest, D.K., K.V. Brix, and W.J. Adams. 1999. Critical review of proposed residue-based selenium toxicity thresholds for freshwater fish. Human and Ecological Risk Assessment 5:1187-1228.

Dwyer, F.J., C.J. Schmitt, S.E. Finger, and P.M. Mehrle. 1988. Biochemical changes in longear sunfish, *Lepomis megalotis*, associated with lead, cadmium and zinc from mine tailings. J. Fish Biology 33:307-317.

Eisler, R. 1988. Lead hazards to fish, wildlife, and invertebrates: a synoptic review. U.S. Fish Wildlife Service Biol. Rep. 85(1.14).

Finger, S.E., A.C. Allert, S.J. Olson, and E.C. Callahan. 1994. Toxicity of irrigation drainage and associated waters in the middle Green River basin, Utah. Final report to the U.S. Fish and Wildfish Service, Salt Lake City, UT. National Fisheries Contaminant Research Center, Columbia, Missouri.

FishPro, Inc. 1996. Feasibility study for improvements and construction of fish hatcheries. Volume II, Native aquatic species facility: Goshen Warm Springs Site, Gandy Warm Springs Site. FishPro, Inc. Port Orchard, Washington.

FishPro, Inc. 2000. Utah warm water sportfish and native aquatic species hatchery siting study for Utah Division of Wildlife Resources and Utah Reclamation Mitigation and Conservation Commission. FishPro, Inc., Port Orchard, Washington.

Hamilton, S.J. 2002. Rationale for a tissue-based selenium criterion for aquatic life. Aquatic Toxicology 57:85-100.

Hamilton, S.J., and D.J. Hoffman. 2002. Trace element and nutrition interactions in fish and wildlife. Pages 1157-1235 *in* Hoffman, D.J., B.A. Rattner, G.A. Burton, Jr., and J. Cairns, Jr., editors. Handbook of Ecotoxicology, 2nd edition. CRC Press, Baton Ration, Florida.

Hamilton, S.J., K.M. Holley, K.J. Buhl, F.A. Bullard, L.K Weston, and S.F. McDonald. 2002. Impact of selenium and other trace elements on the endangered adult razorback sucker. Environmental Toxicology 17:297-323.

Hamilton, S.J., and V.P. Palace. 2001. Assessment of selenium effects in lotic ecosystems. Ecotoxicology and Environmental Safety 50:161-166.

Jarvinen, A.W., and G.T. Ankley. 1999. Linkage of effects to tissue residues: development of a comprehensive database for aquatic organisms exposed to inorganic and organic chemicals. SETAC Technical Publication Series, Pensacola, Florida.

Lemly, D.A. 1982. Response of juvenile centrarchids to sublethal concentrations of waterborne selenium. I. Uptake, tissue distribution, and retention. Aquatic Toxicology 2:235-252.

Lemly, D.A. 1993. Guidelines for evaluating selenium data from aquatic monitoring and assessment studies. Environmental Monitoring and Assessment 28:83-100.

Lemly, D.A. 1995. A protocol for aquatic hazard assessment of selenium. Ecotoxicology and Environmental Safety 32:280-288.

Lemly, D.A. 1996. Selenium in aquatic organisms. Pages 427-445 *in* Beyer, W.N., G.H. Heinz, and A.W. Redmon, editors. Environmental contaminants in wildlife: interpreting tissue concentrations. Lewis Publishers, Boca Raton, Florida.

Lemly, D.A., and G.J. Smith. 1987. Aquatic cycling of selenium: implications for fish and wildlife. U.S. Fish and Wildlife Service, Fish and Wildlife Leaflet 12.

May, T.M., M.J. Walther, J.D. Petty, J.F. Fairchild, J. Lucero, M. Delvaux, J. Manring, M. Armbruster, and D. Hartman. 2001. An evaluation of selenium concentrations in water, invertebrates, and fish from the Republican River Basin: 1997-1999. Environmental Monitoring and Asessment 72:179-206.

Schmitt, C.J., J.L. Zajicek, T.W. May, and D.F. Cowman. 1999. Organochlorine residues and elemental contaminants in U.S. freshwater fish, 1976-1986: National

Contaminant Biomonitoring Program. Reviews of Environmental Contamination and Toxicology 162:43-104.

Shirley, D.L. 1983. Spawning ecology and larval development of the June sucker. Proceedings of the Bonneville Chapter of the American Fisheries Society (1983):18-36.

Snedecor, G.W., and W.G. Cochran. 1969. Statistical Methods. Iowa State University Press, Ames, Iowa.

Statistical Analysis System Institute (SAS). 2000. SAS/STAT Guide for Personal Computers, Version 6, 4th ed., Volumes 1 & 2. SAS Institute, Cary, North Carolina.

Stephens, D.W., B. Waddell, L.A. Peltz, and J.B. Miller. 1992. Detailed study of selenium and selected elements in water, bottom sediment, and biota associated with irrigation drainage in the middle Green River basin, Utah, 1988-1990. Water-Resources Investigation Report 92-4084. U.S. Geological Survey, Salt Lake City, Utah.

SWCA, Inc. 1999. The effects of selenium on fish, a review of literature for the warmwater sportfish and native aquatic species hatchery in Utah. Technical Report prepared for Utah Division of Wildlife Resources. SWCA, Inc., Logan, Utah.

SWCA, Inc. 2002. Non-native fish control feasibility study to benefit June sucker in Utah Lake. Draft Technical Report prepared for Utah Division of Wildlife Resources. SWCA, Inc., Logan, Utah.

UDEQ (Utah Department of Environmental Quality). 2003. Utah Water Quality Standards. Available: http://www.rules.utah.gov/publicat/code/r317/r317-002.htm#18 (March 2003).

USEPA (U.S. Environmental Protection Agency), Office of Water. 2002. Draft Aquatic life water quality criteria for selenium. USEPA, Washington, D.C.

USFWS (U.S. Fish and Wildlife Service), Region 6. 1999. June sucker (*Chasmistes liorus*) recovery plan. USFWS, Denver, Colorado.

Waddell, B., and T. May. 1995. Selenium concentrations in the razorback sucker (*Xyrauchen texanus*): substitution of non-lethal muscle plugs for muscle tissue in contaminant assessment. Archives of Environmental Contamination and Toxicology 28:321-326.

Whitney, M., and M.C. Belk. 2000. Threatened fishes of the world: *Chasmistes liorus* Jordan 1978 (Catostomidae). Environmental Biology of Fishes 57:362.

Table 1. Requirements for accuracy, precision and detection limits (APHA 1998).

Parameter	Estimated Accuracy for each Matrix	Estimated Precision for each Matrix	Precision Protocol for each Matrix	Estimated Detection Limit
Chemical	Measured values within 95% of CI or 10% of Mean	Replicate values within ± 25%	Analyze duplicate at least once per run	Temperature (0.3°C) pH (0.1 unit) Turbidity (1 NTU) Conductivity (100 μmhos/cm) Dissolved oxygen (0.1 mg/L) Metals (varies) Ammonia (20 μg/L) Chlorophyll a (1 μg/L) Alkalinity/hardness (2 mg/L) Total organic carbon (20 μg/L) GPS (10 m)

Table 2. Proposed quality assurance samples for various matrices.

Type	Matrix	Frequency	Analysis	Rationale
Field Duplicate	Site water	1 per run	Selenium, Hydrolab®, water quality	Measures precision of sample collection and degree of environmental variability
Blank	Distilled water	1 per field samples	Selenium	Monitors procedural contamination
Digestion Blank	Tissue	2 - 4 per digestion group or block	Selenium	Monitors method accuracy
Reference Material	Tissue, water, zooplankton	1 - 2 per digestion group or blank	Selenium	Monitors method accuracy
Digestion Replicate	Tissue, water, zooplankton	2 per digestion group or blank	Selenium	Monitors method precision
Analytical Duplicate	Fish, water, sediment, zooplankton	1 per 20 analyses	Selenium, metals, water quality, particle size analysis, total organic carbon	Monitors instrumental precision
Analytical Spike	Fish, water, sediment, zooplankton	1 per analytical run per matrix	Selenium, metals, ammonia	Monitors instrumental accuracy
Laboratory Control Sample	Fish, water, sediment, zooplankton	2 per analytical run	Selenium, metals, ammonia, total organic carbon	Monitors instrumental accuracy
Calibration Standard	Fish, water, sediment, zooplankton	1 per analytical run	Selenium, metals, Hydrolab® water quality ammonia, total organic carbon	Monitors accuracy

Table 3. Average length (mm) and weight (g) of June sucker during the uptake phase of the study. Superscripts donate significant differences between sampling periods (number) ($P < 0.01$) and sites (letter) ($P < 0.01$). Standard deviations (in parenthesis) and number of samples [in brackets].

Site	Sampling Date	Sampling Period (mon)	Days of Exposure	Length (mm)	Weight (g)
Hatchery	07/11/02	0	0	49.2 (5.83) [9]	0.82 (0.27) [9]
Lily Pond	07/26/01	0.5	14	50.3 (5.71) ^{1A} [13]	1.03 (0.34) ^{1A} [13]
Lily Pond	08/09/01	1	28	55.0 (4.84) ^{1A} [8]	1.14 (0.34) ^{1A} [8]
Lily Pond	09/05/01	2	55	59.5 (7.33) ^{1A} [4]	1.38 (0.22) ^{1A} [4]
Lily Pond	10/08/01	3	87	62.3 (6.85) ^{2A} [4]	2.38 (0.91) ^{2A} [4]
North Pond	07/26/01	0.5	14	50.2 (6.89) ^{1A} [12]	0.98 (0.44) ^{1A} [12]
North Pond	08/09/01	1	28	59.0 (7.66) ^{1A} [7]	1.81 (1.11) ^{1A} [7]
North Pond	09/05/01	2	55	60.3 (8.39) ^{1A} [3]	1.80 (0.78) ^{1A} [3]
North Pond	10/08/01	3	87	74.8 (22.8) ^{2B} [4]	5.03 (3.38) ^{2B} [4]
Utah Lake	07/26/01	0.5	14	49.8 (6.48) ^{1A} [12]	1.15 (0.50) ^{1A} [12]
Utah Lake	08/09/01	1	28	62.0 (10.6) ^{2A} [5]	2.01 (1.17) ^{2A} [5]
Utah Lake	09/05/01	2	55	68.8 (5.38) ^{2A} [4]	2.53 (0.54) ^{2A} [4]
Utah Lake	10/07/01	3	86	79.3 (3.77) ^{2C} [4]	4.03 (0.66) ^{2C} [4]

Table 4. Average length (mm) and weight (g) of June sucker during the depuration phase of the study. Superscripts donate significant differences between sampling periods (number) ($P < 0.01$) and sites (letter) ($P < 0.01$). Standard deviations (in parenthesis) and number of samples [in brackets]. Samples were compared to final concentrations during the uptake phase (i.e., 87 d).

Site	Sampling Date	Sampling Period (mon)	Days of Exposure	Length (mm)	Weight (g)
Lily Pond	10/08/01	3	87	62.3 (6.85) ^{1A} [4]	2.38 (0.91) ^{1A} [4]
Lily Pond	11/05/01	4	116	61.7 (1.53) ^{1A} [3]	2.10 (0.10) ^{1A} [3]
Lily Pond	12/03/01	5	144	61.0 (4.24) ^{1A} [2]	1.25 (0.78) ^{1A} [2]
North Pond	10/08/01	3	87	74.8 (22.8) ^{1B} [4]	5.03 (3.38) ^{1B} [4]
North Pond	11/05/01	4	116	86.7 (3.51) ^{1B} [3]	7.00 (1.04) ^{1B} [3]
North Pond	12/03/01	5	144	89.7 (11.4) ^{1B} [13]	7.20 (2.80) ^{1B} [13]
Utah Lake	10/07/01	3	86	79.3 (3.77) ^{1A} [4]	4.03 (0.66) ^{1B} [4]
Utah Lake	11/05/01	4	116	96.7 (8.96) ^{2B} [3]	7.80 (2.25) ^{1B} [3]
Utah Lake	12/03/01	5	144	94.4 (8.70) ^{2B} [15]	7.39 (2.16) ^{1B} [15]

Table 5. Growth rates (g/d) for June suckers in each cage. Rates were calculated for the entire study (i.e., all time periods), and for each segment of the study (i.e., uptake and depuration).

		All Time Periods		Uptake Phase		Depuration Phase	
		Day 0 - 144		Day 0 - 87		Day 88 - 144	
Site	Cage	Growth Rate	P-Value	Growth Rate	P-Value	Growth Rate	P-Value
Lily Pond	A	0.010	0.0010	0.009	0.098	0.014	0.7761
Lily Pond	B	0.008	0.0158	0.021	< 0.0001	-0.027	0.1703
Lily Pond	C	0.011	< 0.0001	0.011	0.0010	0.014	0.2601
Lily Pond	D	0.009	0.0485	0.009	0.0485	No fish	
North Pond	E	0.045	< 0.0001	0.041	< 0.0001	0.027	0.5470
North Pond	F	0.062	< 0.0001	0.079	< 0.0001	-0.086	CNC [1]
North Pond	G	0.046	< 0.0001	0.033	0.0052	0.058	0.1504
North Pond	H	0.027	0.0051	0.027	0.0051	No cage	
Utah Lake	I	0.033	0.0214	0.033	0.0214	No fish	
Utah Lake	J	0.048	< 0.0001	0.042	< 0.0001	0.045	0.2892
Utah Lake	K	0.052	< 0.0001	0.031	< 0.0001	0.079	0.0230
Utah Lake	L	0.040	< 0.0001	0.031	< 0.0001	0.049	0.1562

[1] Could not calculate due to insufficient sample numbers.

Table 6. Growth rates (g/d) for June suckers at each site (i.e., average of cages). Rates were calculated for each segment of the study (i.e., uptake and depuration). Superscripts denote significant differences ($P < 0.01$) between sites.

	Uptake Phase Day 0 - 87		Depuration Phase Day 88 - 144	
Site	**Growth Rate**	**P-Value**	**Growth Rate**	**P-Value**
Lily Pond	0.0134^{A}	< 0.0001	-0.0116^{A}	0.2856
North Pond	0.0427^{AB}	< 0.0001	0.0377^{A}	0.1318
Utah Lake	0.0332^{B}	< 0.0001	0.0596^{A}	0.0020

Table 7. Proposed chronic toxicity thresholds for selenium based on water, sediment and tissue. Sediment, zooplankton and fish are expressed as dry weight.

Matrix	Lemly (1993)	DeForest et al. (1999)
Water	2-5 μg/L	5-32 μg/L
Sediment	4 μg/g	NP[1]
Invertebrates	3 μg/g	10 μg/g
Whole-body Fish	4 μg/g	9 μg/g

[1] NP = none proposed.

Table 8. Concentration of selenium (μg/g dry wt.) in food fed to June suckers.

Food Type	Sample Day (mon)	Collected	Se (μg/g dry wt.)
BioKyowa B - FFBK700	0	07/11/01	3.40
BioKyowa B- FFBK700	14	07/26/01	3.55
BioKyowa C - FFKC1000	0	07/11/01	2.39

Table 9. Means, standard deviations (in parenthesis) and number of samples [in brackets] of selenium concentrations in June sucker during the uptake phase of the study. Superscripts denote significant differences between sampling dates (number) and sites (letter). Sampling dates were compared to initial concentration (1.13 μg/g dry weight). Differences among sites were compared within given date.

Site	Sample Date	Sampling Period (mon)	Days of Exposure	Se conc. (μg/g dry wt.)	% Gain
Hatchery	07/11/02	0	0	1.13 (0.14)^{1A} [13]	--
Lily Pond	07/26/01	0.5	14	1.14 (0.18)^{1A} [13]	0.88
Lily Pond	08/09/01	1	28	1.07 (0.12)^{1A} [8]	-5.31
Lily Pond	09/05/01	2	55	1.53 (0.21)^{1A} [4]	35.4
Lily Pond	10/07/01	3	87	1.62 (0.20)^{2A} [4]	43.4
North Pond	07/26/01	0.5	14	1.40 (0.18)^{1A} [12]	23.9
North Pond	08/09/01	1	28	1.25 (0.25)^{1A} [7]	10.6
North Pond	09/05/01	2	55	1.94 (0.31)^{1A} [3]	71.7
North Pond	10/07/01	3	87	1.90 (0.43)^{2A} [4]	68.1
Utah Lake	07/26/01	0.5	14	1.31 (0.20)^{1A} [12]	15.9
Utah Lake	08/13/01	1	32	1.21 (0.30)^{1A} [5]	7.08
Utah Lake	09/05/01	2	55	1.39 (0.09)^{1A} [4]	23.0
Utah Lake	10/07/01	3	87	1.32 (0.04)^{1B} [4]	16.8

Table 10. Means, standard deviations (in parenthesis), and number of samples [in brackets] of selenium concentrations in June sucker during the depuration phase of the study. Superscripts denote significant differences between sampling dates (number) and sites (letter). Sampling dates were compared to final selenium concentrations from the uptake phase (i.e., 87 d). Differences among sites were compared within given date.

Site	Sample Date	Sampling Period (mon)	Days of Exposure	Se conc. (µg/g dry wt.)	% Loss
Lily Pond	10/07/01	3	87	1.62 (0.20) ^{1A} [4]	--
Lily Pond	11/05/01	4	116	1.37 (0.07) ^{1A} [3]	-15.4
Lily Pond	12/03/01	5	144	1.50 (0.18) ^{1A} [4]	-7.41
North Pond	10/07/01	3	87	1.90 (0.43) ^{1A} [4]	--
North Pond	11/05/01	4	116	1.47 (0.17) ^{1A} [3]	-22.6
North Pond	12/03/01	5	144	1.18 (0.12) ^{2A} [8]	-37.9
Utah Lake	10/07/01	3	87	1.32 (0.04) ^{1B} [4]	--
Utah Lake	11/05/01	4	116	1.19 (0.07) ^{1B} [3]	-9.85
Utah Lake	12/03/01	5	144	1.17 (0.07) ^{1B} [12]	-11.4

Table 11. Selenium uptake and depuration rates (μg/g/d) for June suckers in each cage. Rates were calculated for the entire study (i.e., all time periods), and for each segment of the study (i.e., uptake and depuration).

		All Time Periods Day 0 - 144		Uptake Data Day 0 - 87		Depuration Data Day 88 - 144	
Site	Cage	Se Uptake Rate	P-Value	Se Uptake Rate	P-Value	Se Depuration Rate	P-Value
Lily Pond	A	0.002	0.0272	0.003	0.0414	-0.003	CNC [1]
Lily Pond	B	0.003	0.0008	0.006	0.0033	0.0003	0.9364
Lily Pond	C	0.005	0.0006	0.007	0.0002	-0.014	0.0695
Lily Pond	D	0.005	0.0950	0.005	0.0950	No fish	
North Pond	E	0.020	0.1293	0.008	0.0005	-0.0041	0.1259
North Pond	F	0.007	0.0004	0.010	0.0002	-0.008	CNC
North Pond	G	0.002	0.0873	0.010	< 0.0001	-0.018	0.0634
North Pond	H	0.011	0.0293	0.011	0.0293	No cage	
Utah Lake	I	0.006	0.0726	0.005	0.0726	No fish	
Utah Lake	J	0.001	0.5770	0.003	0.0844	-0.003	0.0708
Utah Lake	K	0.001	0.4710	0.003	0.0266	-0.004	0.0989
Utah Lake	L	0.001	0.3110	0.002	0.24889	-0.002	0.5695

[1] Could not calculate due to insufficient sample number.

Table 12. Selenium uptake and depuration rates (μg/g/d) for June suckers at each site (i.e., average of cages). Rates were calculated for each segment of the study (i.e., uptake and depuration). Superscripts denote significant differences (P < 0.01) between sites.

	Uptake Data		Depuration Data	
	Day 0 - 87		Day 88 - 144	
Site	Se Uptake Rate	P-Value	Se Depuration Rate	P-Value
Lily Pond	0.0054 [B]	< 0.0001	-0.0029 [A]	0.3376
North Pond	0.0097 [A]	< 0.0001	-0.0126 [A]	0.0051
Utah Lake	0.0030 [B]	0.0005	-0.0027 [A]	0.0055

Table 13. Means and standard deviations (in parenthesis) of selenium concentrations in unfiltered (total) water (N = 2), and filtered (dissolved) water (N = 2), zooplankton (N = 3), and sediment (N = 2).

			Water			
Site	**Sample Date**	**Days of Exposure**	**Total (µg/L)**	**Dissolved (µg/L)**	**Zooplankton (µg/g dry wt.)**	**Sediment (µg/g dry wt.)**
Lily Pond	07/11/02	0	3.7 (.006)	3.70 (0.01)	2.50 (1.63)	2.93 (0.14)
Lily Pond	07/26/01	14	3.86 (0.20)[1]	3.61 (0.01)[1]	3.04 (0.81)[2]	3.14 (0.03)
Lily Pond	08/09/01	28	2.93 (2.30)[1]	4.05 (0.01)[1]	NT[3]	NT
Lily Pond	09/05/01	55	4.05 (0.8)	4.20 (0.25)	5.81 (4.48)	2.82 (0.03)
Lily Pond	10/07/01	87	3.79 (0.01)	3.81 (0.07)	3.60 (0.77)	3.11 (0.34)
North Pond	07/11/02	0	4.07 (0.30)	3.92 (0.11)	5.77 (1.63)	6.14 (0.29)
North Pond	07/26/01	14	3.85 (0.02)	3.97 (0.13)	4.17 (1.01)	6.30 (0.84)
North Pond	08/09/01	28	4.41 (0.21)	3.89 (0.54)	NT	NT
North Pond	09/05/01	55	4.09 (0.24)	4.37 (0.01)	4.51 (1.41)	5.70 (0.49)
North Pond	10/07/01	87	4.18 (0.06)	4.34 (0.13)	4.06 (1.06)	5.74 (0.44)
Utah Lake	07/11/02	0	1.16 (0.20)	0.99 (0.13)	1.91 (0.87)	1.56 (0.11)
Utah Lake	07/26/01	14	1.94 (0.12)	1.30 (0.05)	1.71 (0.50)	1.63 (0.06)
Utah Lake	08/13/01	28	1.72 (0.50)	1.13 (0.16)	NT	NT
Utah Lake	09/05/01	55	0.52 (0.10)	0.37 (0.10)	2.68 (0.62)	1.57 (0.13)
Utah Lake	10/07/01	87	0.63 (0.12)	0.50 (0.08)	3.53 (1.10)	1.75 (0.07)
CERC Well Water	10/10/01	116	< 0.77 (0)	< 0.77 (0)	NT	NT
CERC Well Water	12/03/01	144	< 0.77 (0)	< 0.77 (0)	NT	NT

[1] N = 3.
[2] Average = 3.04; standard deviation = 0.81, if high value (10.9) omitted.
[3] Not taken.

Table 14. Concentrations of metals (μg/g dry weight) in fish samples. Metal concentrations determined by ICP-MS semi-quantitative scan for fish samples, except selenium was determined by flow injection hydride generation atomic absorption spectroscopy, and mercury was determined by cold vapor atomic absorption spectroscopy.

Site	Pre-Exposure	Post-Exposure			Wild Fish					
	Hatchery	Lily Pond	North Pond	Utah Lake	Lily Pond		North Pond		Utah Lake	
Type of Sample	Composite	Composite	Composite	Composite	Single fish	Composite	Composite	Composite	Composite	Composite
Element	June sucker	June sucker	June sucker	June sucker	Utah chub	Western mosquitofish	Utah chub	Western mosquitofish	Unknown minnow	Western mosquitofish
Na	4200	5100	5400	5300	4700	6000	4300	5400	3500	3300
Mg	1300	1400	1200	1600	1500	1800	1700	1700	2100	1800
Al	15	8.5	3.8	6.6	16	40	26	45	320	100
K	15000	15000	12000	18000	18000	14000	18000	15000	21000	16000
Ca	33000	42000	25000	42000	40000	58000	39000	54000	40000	51000
Ti	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	0.22	< 0.1
V	< 0.1	< 0.1	< 0.1	< 0.1	0.19	< 0.1	< 0.1	< 0.1	0.61	< 0.1
Cr	1.6	< 1	< 1	< 1	< 1	< 1	< 1	1.9	< 1	< 1
Mn	2.3	72	3.8	7.9	5.9	16	9.2	16	14	19
Fe	250	390	250	230	230	210	290	270	530	300
Co	0.42	0.6	0.39	0.29	< 0.1	0.17	0.15	0.15	0.12	< 0.1
Ni	< 1	< 1	< 1	3	< 1	< 1	< 1	< 1	< 1	< 1
Cu	15	11	7.8	6.3	6.6	6	6.8	6.8	6	47
Zn	140	460	280	390	190	270	250	180	150	250
Ga	0.21	0.34	0.13	0.35	0.31	0.45	0.31	0.42	0.56	0.41
Ge	< 0.1	< 0.1	< 0.1	< 0.1	0.25	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1

Table 14 (*cont.*).

Site	Pre-Exposure	Post-Exposure			Wild Fish					
	Hatchery	Lily Pond	North Pond	Utah Lake	Lily Pond		North Pond		Utah Lake	
Type of Sample	Composite	Composite	Composite	Composite	Single fish	Composite	Composite	Composite	Composite	Composite
Element	June sucker	June sucker	June sucker	June sucker	Utah chub	Western mosquitofish	Utah chub	Western mosquitofish	Unknown minnow	Western mosquitofish
As	0.15	0.37	0.60	0.62	1.9	1.50	1.6	1.60	1.70	0.54
Se	1.33	1.31	1.41	1.28	5.54	4.38	6.06	5.49	2.61	1.32
Rb	3.6	16	14	13	48	45	41	54	22	19
Sr	30	88	68	230	200	350	170	3320	260	380
Mo	< 0.1	0.11	< 0.1	0.17	0.48	0.28	0.17	0.18	0.17	0.14
Ag	< 0.1	< 0.1	< 0.1	< 0.1	2.3	1.3	0.21	0.51	< 0.1	< 0.1
Cd	< 0.1	< 0.1	0.12	< 0.1	< 0.1	0.14	< 0.1	< 0.1	< 0.1	< 0.1
Sn	0.4	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1
Te	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	0.11	< 0.1	< 0.1	< 0.1
Cs	< 1	< 1	1.2	< 1	2.4	8.2	2.8	6	< 1	< 1
Ba	2.1	3.7	1.7	4.8	4.9	8.8	5	11	17	16
La	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	0.2	< 0.1
Ce	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	0.42	< 0.1
Nd	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	0.21	< 0.1
Hg	0.41	0.43	0.20	0.19	0.17	0.30	0.08	0.06	0.02	0.02
Pb	< 1	< 1	< 1	< 1	< 1	1.4	< 1	2.1	1.2	< 1

List of elements with no detectable concentrations in fish: Pd, Sb, Pr, Sm, Eu, Gd, Tb, Dy, Ho, Er, Tm, Yb, Lu, Hf, Tl, Ta, W, Re, Os, Ir, Pt, Au, Tl (< 0.1 μg/g dry weight); Li, Be, Y, Zr, Nb, Ru, In, Bi, U (< 1 μg/g dry weight).

Table 15. Average temperatures (°C) and standard deviations (in parenthesis) for all sites. Temperature measured with Onset® Tibit Temperature Monitors. Superscripts donate significant differences between sampling intervals (number) ($P < 0.001$) and sites (letter) ($P < 0.001$).

Site	July	August	September/October	Overall
Lily Pond	21.2 (0.54) [1A]	21.2 (0.48) [1A]	20.8 (0.43) [2A]	21.0 (0.51)
	N=462	N=743	N=894	N=2009
North Pond	20.1 (1.08) [1B]	20.0 (0.75) [2B]	19.4 (0.81) [2B]	19.8 (0.91)
	N=464	N=743	N=897	N=2104
Utah Lake	19.0 (2.63) [1C]	18.7 (3.63) [2C]	13.4 (3.55) [3C]	16.5 (4.33)
	N=466	N=743	N=876	N=2087

Table 16. Means and standard deviations (in parenthesis) of field water quality parameters. N = 2 for each sample date, except July 26, 2001, where N = 6.

Site	Sample Date	Days of Exposure	Temperature (°C)	pH	Conductivity (mS/cm)	Dissolved Oxygen (mg/L)
Lily Pond	07/11/02	0	21.2 (0)	7.47 (0.02)	2.26 (0)	3.99 (0.17)
Lily Pond	07/26/01	14	21.2 (0.01)	7.40 (0.04)	2.29 (0.01)	4.15 (0.04)
Lily Pond	08/09/01	28	22.0 (0.02)	6.90 (0.01)	2.27 (0)	4.08 (0.23)
Lily Pond	09/05/01	55	20.8 (0.02)	7.53 (0.03)	2.29 (0.01)	2.80 (0.18)
Lily Pond	10/07/01	87	20.4 (0.01)	7.70 (0.06)	2.24 (0)	3.58 (0.40)
North Pond	07/11/02	0	21.8 (0.02)	7.45 (0.04)	2.16 (0)	5.07 (0.08)
North Pond	07/26/01	14	20.4 (0.14)	7.48 (0.03)	2.22 (0.01)	3.49 (0.21)
North Pond	08/09/01	28	20.9 (0.09)	7.00 (0.01)	2.19 (0)	4.21 (0.10)
North Pond	09/05/01	55	20.3 (0.11)	7.67 (0.01)	2.17 (0)	3.61 (0.11)
North Pond	10/07/01	87	19.4 (0.28)	7.77 (0.11)	2.14 (0.01)	4.43 (0.01)
Utah Lake	07/11/02	0	27.2 (0.55)	8.97 (0.01)	1.34 (0)	12.45 (0.25)
Utah Lake	07/26/01	14	21.6 (0.05)	8.65 (0.03)	1.40 (0.01)	6.16 (0.19)
Utah Lake	08/09/01	28	20.6 (0.03)	8.10 (0.03)	1.40 (0.01)	3.28 (0.04)
Utah Lake	09/05/01	55	22.6 (0.20)	8.32 (0.03)	1.51 (0.01)	6.25 (0.32)
Utah Lake	10/07/01	87	16.4 (1.48)	8.51 (0.03)	1.45 (0.02)	9.11 (0.81)

Table 17. Means and standard deviations (in parenthesis) of laboratory water quality parameters. N = 2 for each sample date. Alk = alkalinity. Hard = hardness. POC = particulate organic carbon.

Site	Sample Date	Days of Exposure	Alk (mg/L $CaCO_3$)	Hard (mg/L $CaCO_3$)	Turbidity (NTU)	Ammonia (mg/L as N)	Chlorophyll a (µg/L)	POC (µg/L)
Lily Pond	07/11/02	0	247 (1.41)	353 (1.41)	8 (0)	0.014 (0.006)	3.67 (2.3)	144 (100)
Lily Pond	07/26/01	14	248 (0)	351 (1.41)	2.55 (3.04)	0.01 (0)	1.06 [3]	2401 (2968)
Lily Pond	08/09/01	28	253 (1.41)	360 (0)	0.55 (0)	0.02 (0)	0.51 (0.40)	240 (26.8)
Lily Pond	09/05/01	55	251 (2.31) [1]	384 (6.00) [1]	0.82 (0.38) [1]	0.012 (0.003)	1.69 (1.74)	380 (218)
Lily Pond	10/7/2001	87	253 (12.7)	365 (7.07)	0.90 (0.84)	NT[2]	2.50 (0.35)	364 (111)
North Pond	07/11/02	0	245 (1.41)	346 (0)	9.5 (2.12)	0.015 (0.007)	2.49 (0.26)	238 (3.09)
North Pond	07/26/01	14	216 (39.6)	347 (4.24)	0.5 (0)	0.005 (0.007)	1.12 (0.22)	227 (35.0)
North Pond	08/09/01	28	252 (2.83)	356 (283)	0.43 (0.02)	0.020 (0)	0.20 (0.01)	159 (48.0)
North Pond	09/05/01	55	249 (1.41) [1]	383 (12.7) [1]	0.55 (0.21) [1]	0.015 (0.007)	0.24 (0.03)	368 (83.2)
North Pond	10/7/2001	87	249 (9.90)	358 (5.66)	1.75 (1.77)	NT[2]	0.93 (0.23)	178 (2.60)
Utah Lake	07/11/02	0	215 (4.24)	346 (8.49)	78.5 (2.12)	0.030 (0)	218 (49.6)	17943 (1786)
Utah Lake	07/26/01	14	350 (14.1)	420 (28.3)	250 (0)	0.02 (0.014)	199 (7.13)	57891 (12029)
Utah Lake	08/09/01	28	276 (22.6)	372 (17.0)	79.5 (4.95)	0.025 (0.007)	30.1 (4.26)	28542 (2589) [4]
Utah Lake	09/05/01	55	289 (4.24) [1]	452 (5.66) [1]	67.5 (3.54) [1]	0.89 (0)	32.0 (5.32)	12600 (3802) [4]
Utah Lake	10/7/2001	87	260 (5.66)	483 (4.24)	55.0 (36.8)	NT[2]	88.0 (3.20)	11382 (6872)
CERC Well Water	10/10/2001	116	256 (0)	285 (1.41)	1.02 (0.11)	NT	0.03 (0.01)	29.8 (35.0) [4]
CERC Well Water	12/3/2001	144	259 (4.24)	289 (4.24)	0.77 (0.01)	NT	0.02 (0.01)	49.6 (36.4) [4]

[1] N = 3; [2] N = Not Tested; [3] N = 1; [4] N = 4.

Table 18. Particle size of sediment.

Site	Station	Duplicate	Date Collected	% Sand	% Silt	% Clay
Lily Pond	East		07/11/01	70.1	17.9	12
Lily Pond	East	dup	07/11/01	63.3	5.9	30.8
Lily Pond	West		07/11/01	76.2	2.6	21.2
North Pond	North		07/11/01	55.4	12.9	31.7
North Pond	South		07/11/01	46.6	13.0	40.4
North Pond	South	dup	07/11/01	49	8.9	42.1
Utah Lake	East		07/11/01	28.9	15.3	55.8
Utah Lake	West		07/11/01	31.5	12.7	55.8
Utah Lake	West	dup	07/11/01	32.9	14.7	52.4

Table 19. Sediment carbon analysis. TC = total carbon per gram of sediment. TIC = inorganic carbon per gram of sediment. TOC = organic carbon per gram of sediment. %TIC = percent of inorganic carbon per each gram of sediment. %TOC = percent of organic carbon perr each gram of sediment.

Site	Location	Date	Rep	TC (µg C/g)	TIC (µg C/g)	TOC (µg C/g)	%TIC	%TOC
Lily Pond	East	07/11/01	1	94188	58949	35238	5.89	3.52
Lily Pond	West	07/11/01	1	125022	60042	64980	6.00	6.50
North Pond	North	07/11/01	1	111859	62146	49714	6.21	4.97
North Pond	North	07/11/01	2	109371	65353	44018	6.54	4.40
North Pond	South	07/11/01	1	110036	63562	46475	6.36	4.65
North Pond	South	07/11/01	2	106232	58517	47715	5.85	4.77
Utah Lake	East	07/11/01	1	67076	39144	27932	3.91	2.79
Utah Lake	East	07/11/01	2	67238	38862	28375	3.89	2.84
Utah Lake	West	07/11/01	1	69019	35916	33103	3.59	3.31
Utah Lake	West	07/11/01	2	86113	36489	49624	3.65	4.96

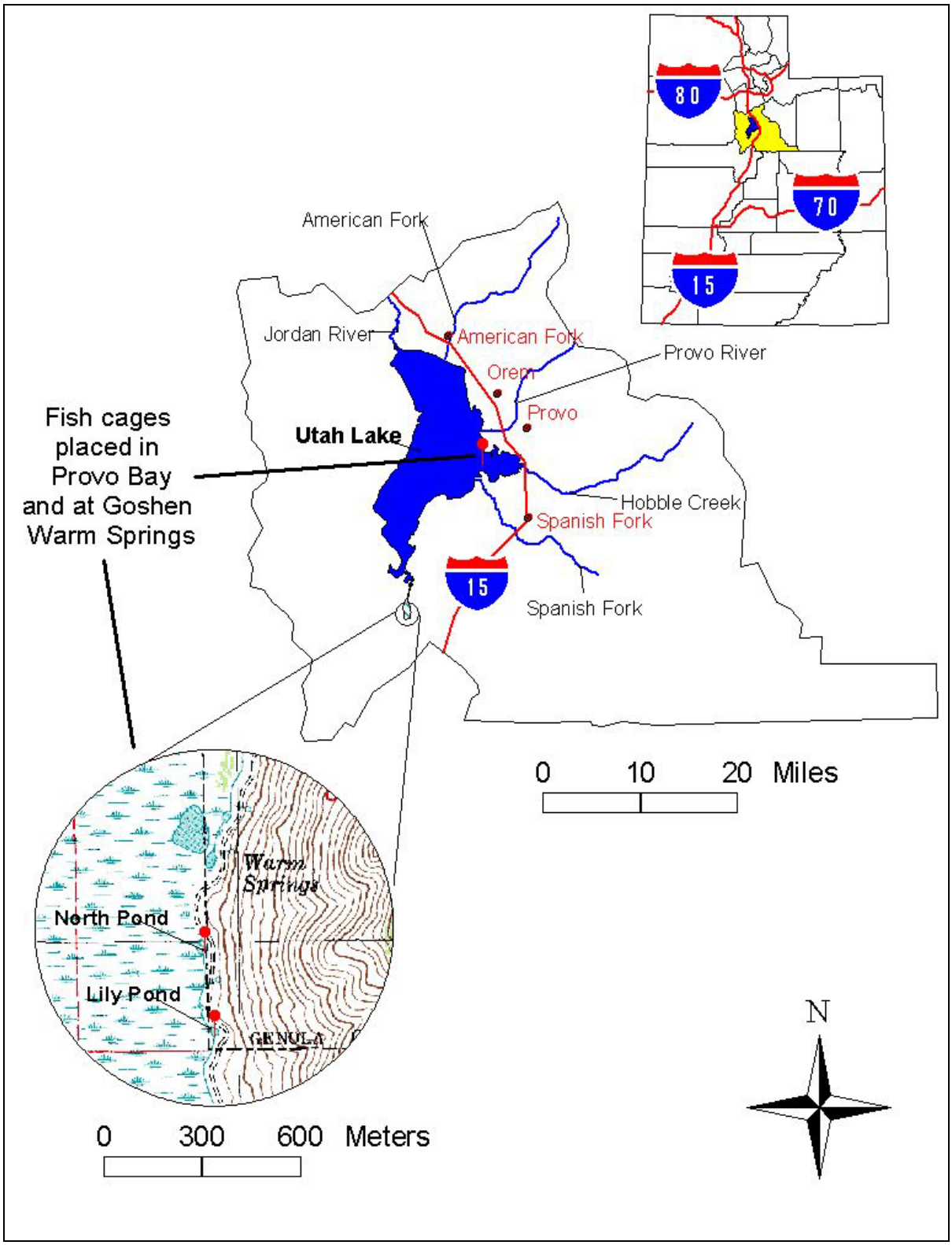

Figure 1. Site locations.

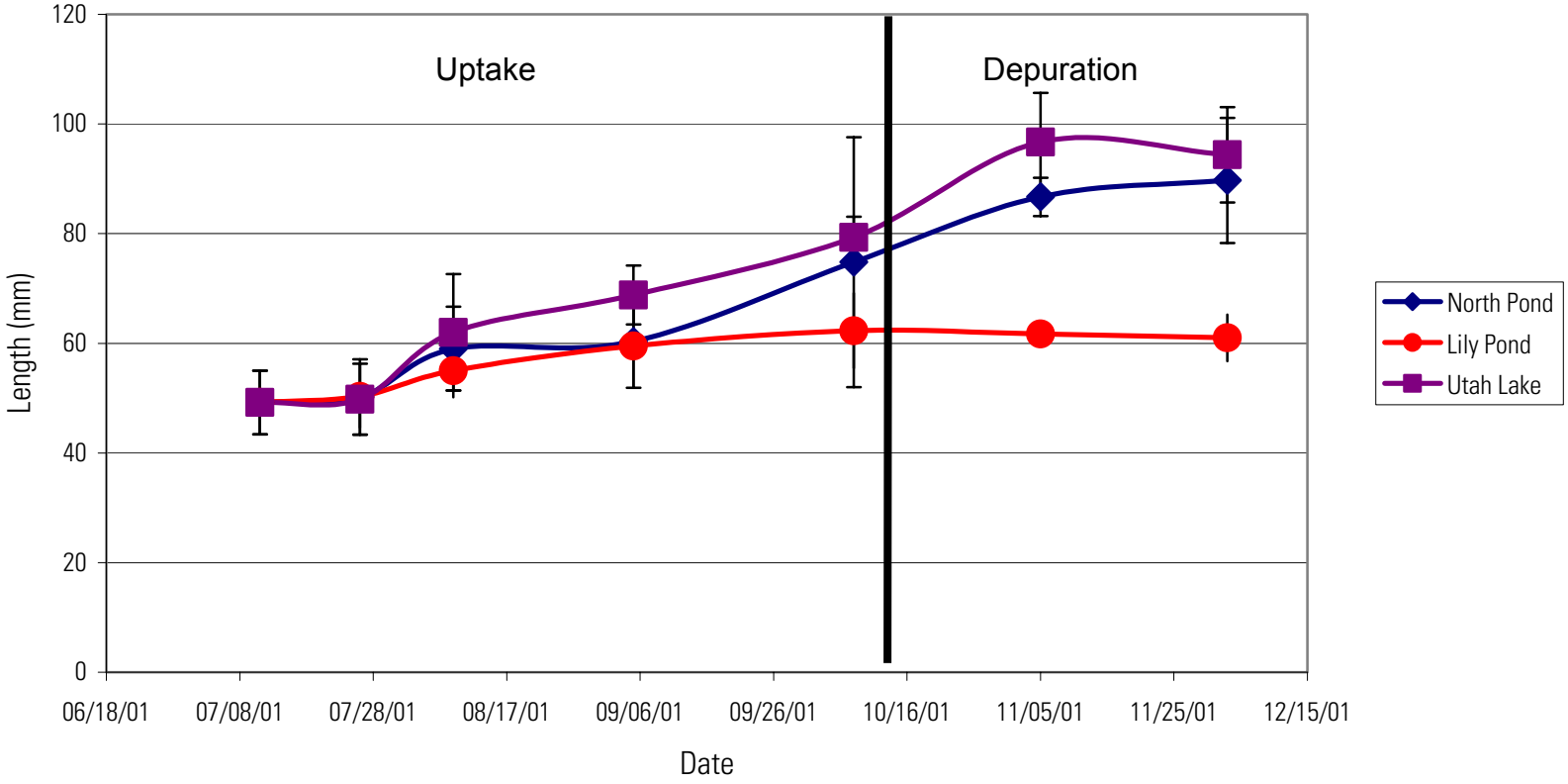

Figure 2. Mean length (mm) of June suckers for each sampling date. See Tables 3 and 4 for number of fish used for each data point. Vertical line represents the division between the uptake (0 - 87 d) and depuration (88 - 144 d) phases of the study. Bars represent standard deviations.

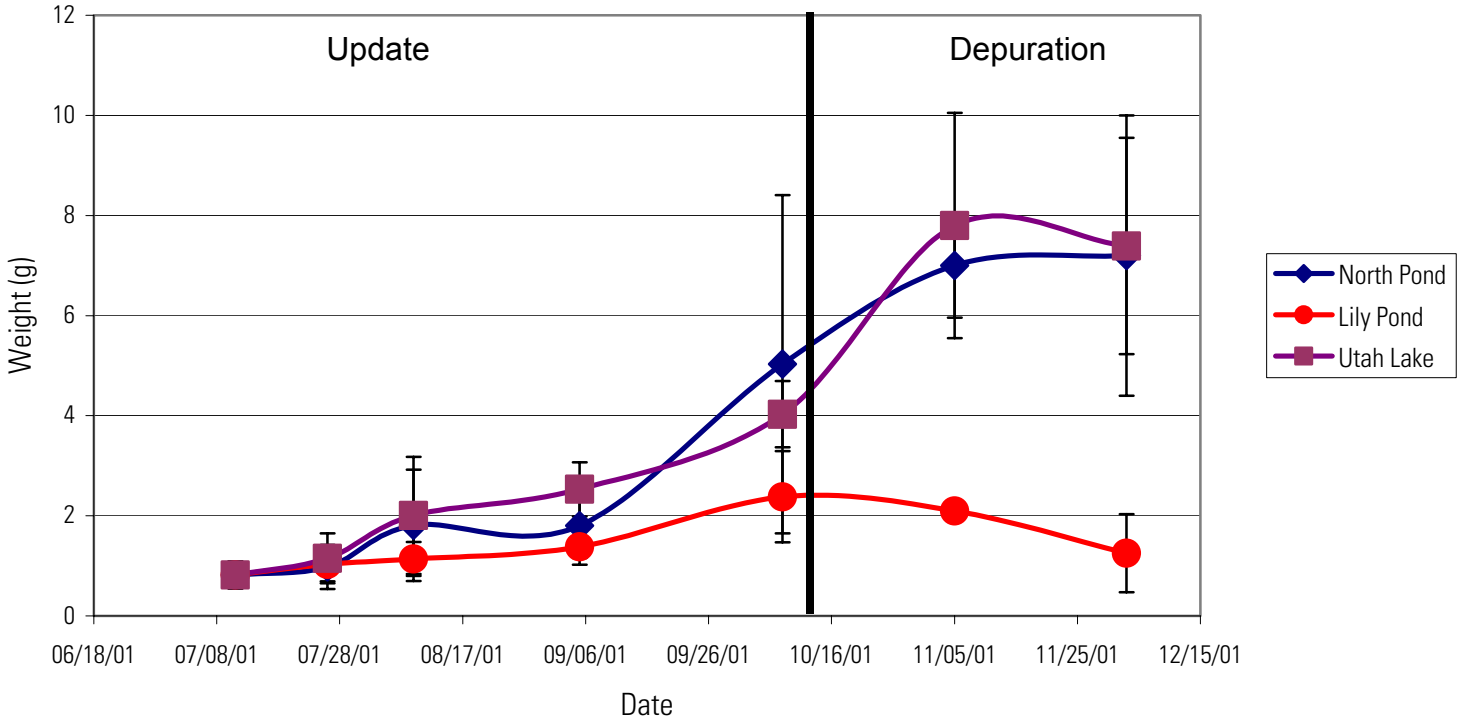

Figure 3. Mean weight (mg) of June suckers for each sampling date. See Tables 3 and 4 for number of fish used for each data point. Vertical line represents the division between the uptake (0 - 87 d) and depuration (88 - 144 d) phases of the study. Bars represent standard deviations.

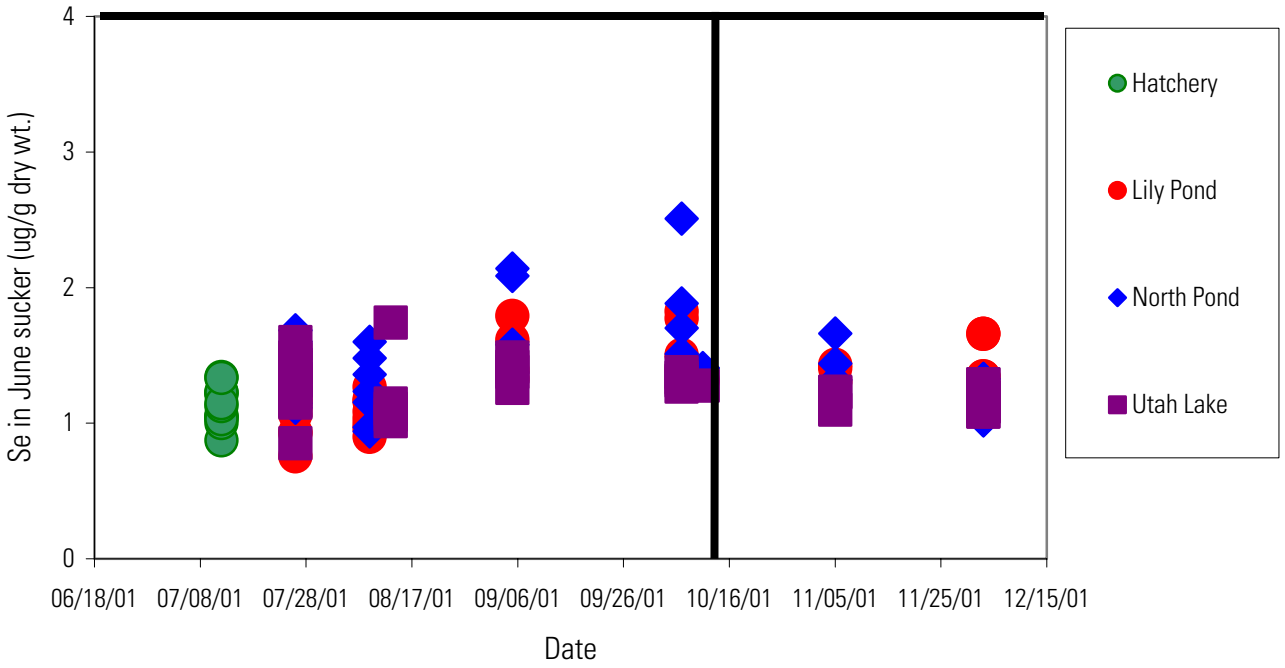

Figure 4. Selenium concentrations measured in June sucker. Solid horizontal line represents selenium threshold for whole-body fish proposed by Lemly (1993). DeForest et al. (1999) proposed a threshold for whole-body fish of 9 μg/g dry weight. Vertical line represents the division between the uptake (0 - 87 d) and depuration (88 – 144 d) phases of the study.

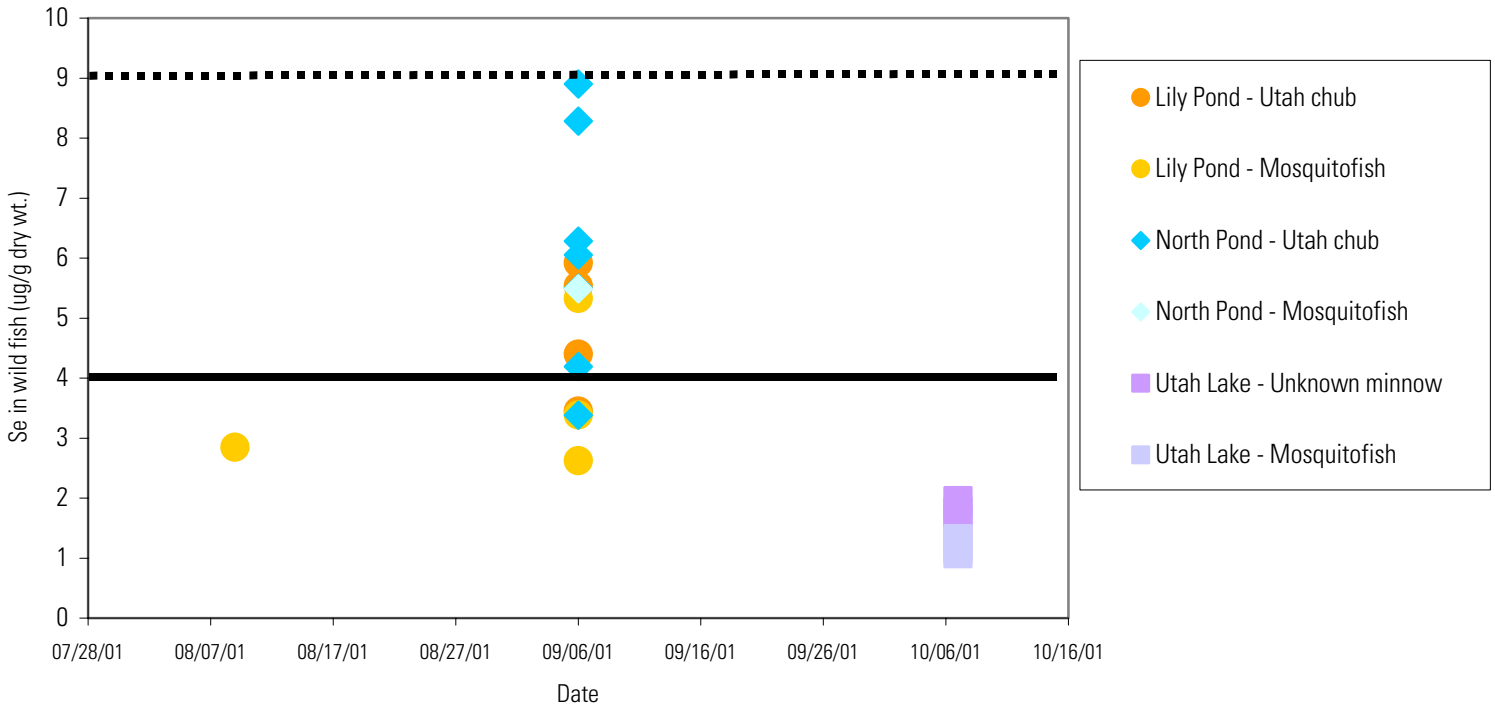

Figure 5. Selenium concentrations (µg/g dry wt.) measured in wild fish. Solid horizontal line represents selenium threshold for whole-body fish proposed by Lemly (1993). The dashed line represents selenium threshold for whole-body fish proposed by DeForest et al. (1999).

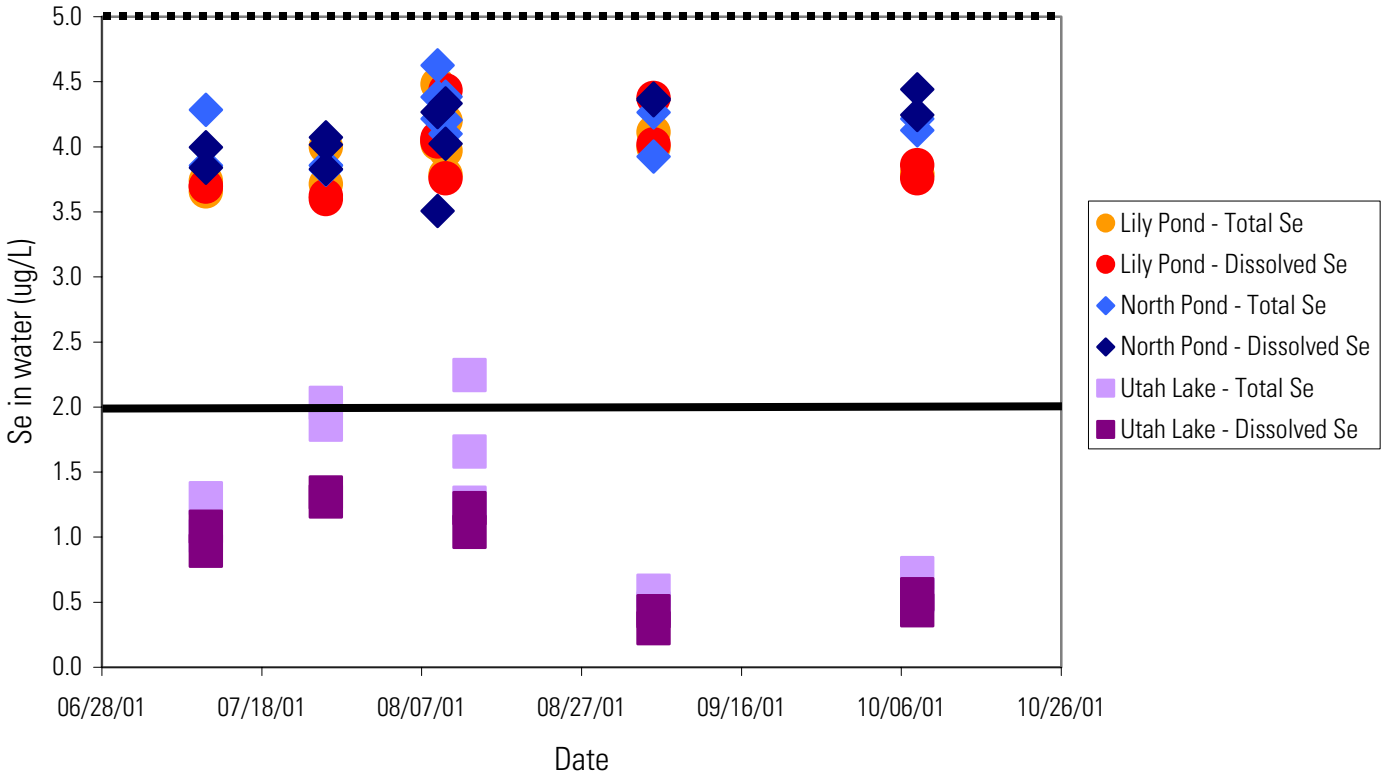

Figure 6. Selenium concentrations (μg/L) measured in water. Solid horizontal line represents the lower limit of the selenium threshold for water (2-5 μg/L) proposed by Lemly (1993). The dashed horizontal line represents lower limit of the selenium threshold for water (5-32 μg/L) proposed by DeForest et al. (1999).

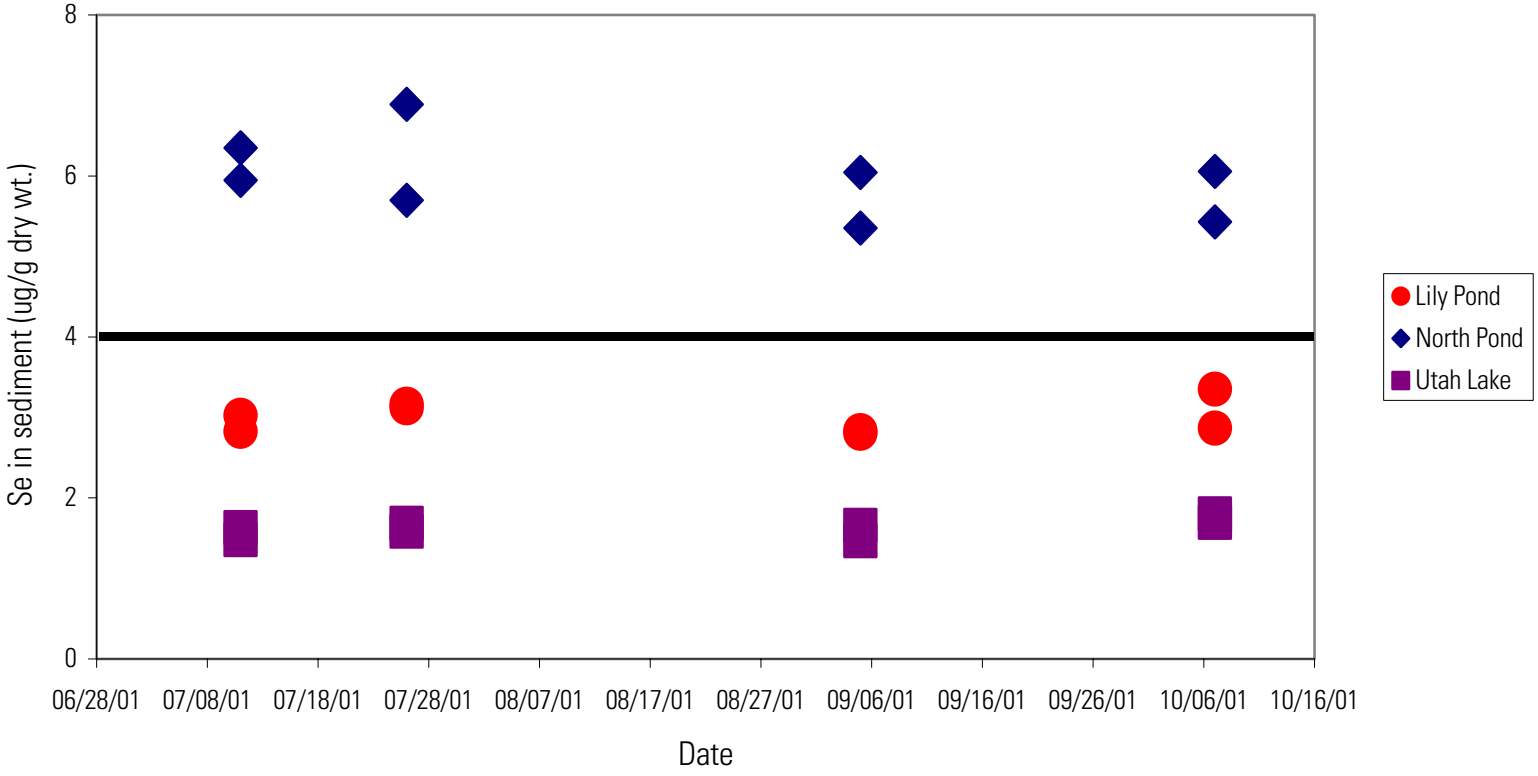

Figure 7. Selenium concentrations (μg/g dry wt.) measured in sediment. Solid horizontal line represents selenium threshold for sediment proposed by Lemly (1993).

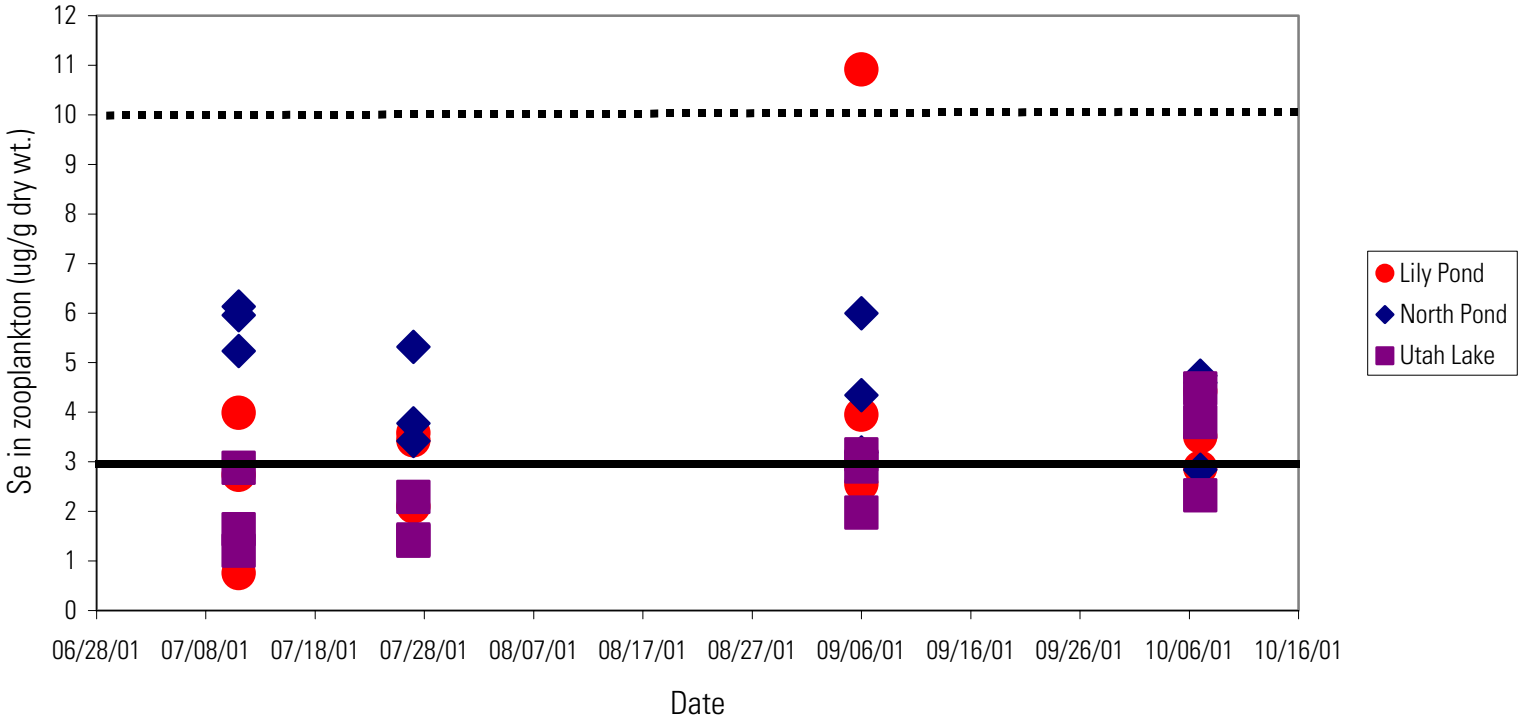

Figure 8. Selenium concentrations (μg/g dry wt.) measured in zooplankton. The solid horizontal line represents the selenium threshold for food-chain organisms proposed by Lemly (1993). The dashed line represents the selenium threshold for food-chain organisms proposed by DeForest et al. (1999).

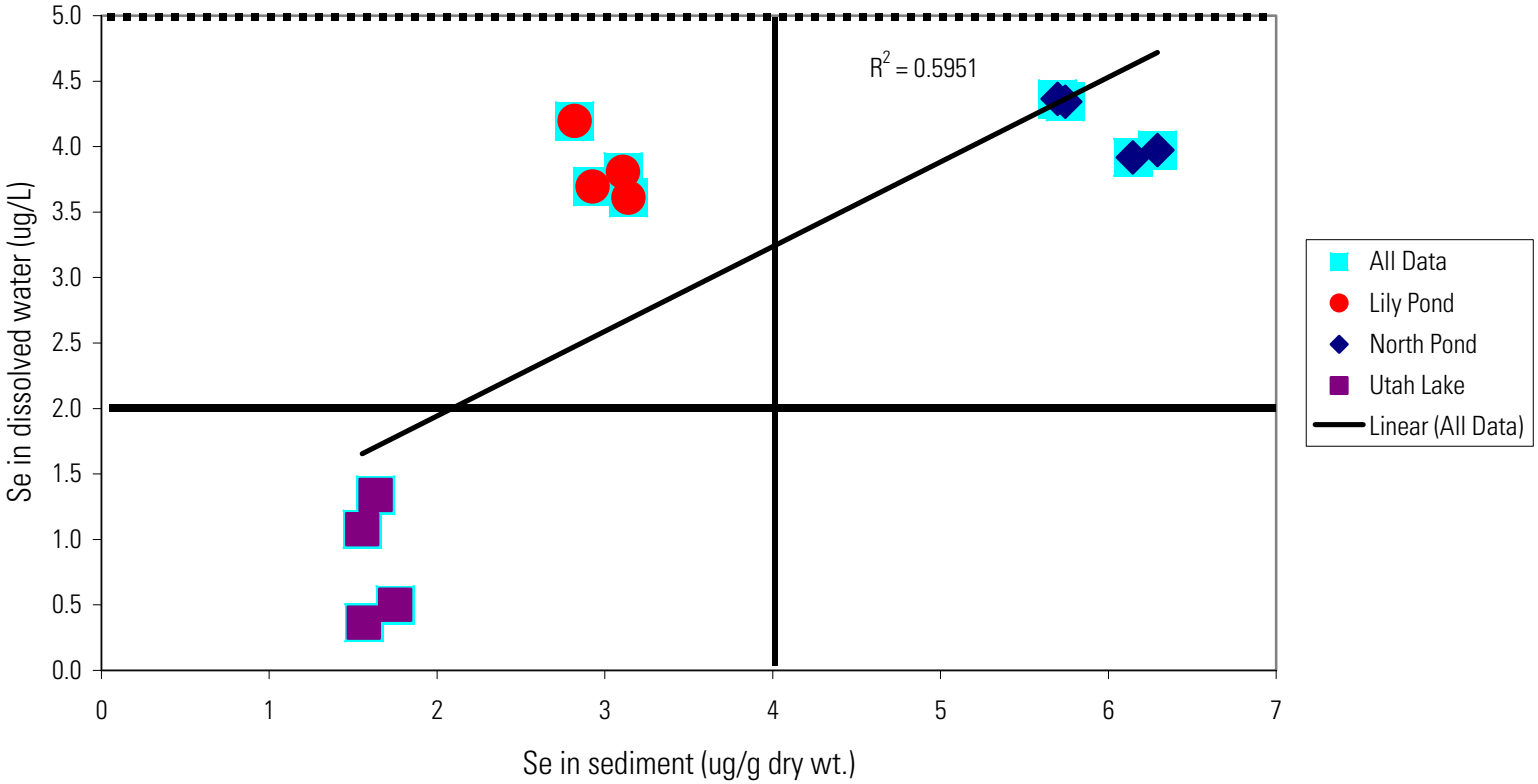

Figure 9. Relationship of selenium concentrations in filtered water (i.e., dissolved) (μg/L) and sediment (μg/g dry wt.). The solid horizontal line represents the lower limit of the selenium threshold for water (2-5 μg/L) proposed by Lemly (1993). The dashed horizontal line represents the lower limit of the selenium threshold (5-32 μg/L) proposed by DeForest et al. (1999). The vertical line represents the selenium threshold for sediment proposed by Lemly (1993).

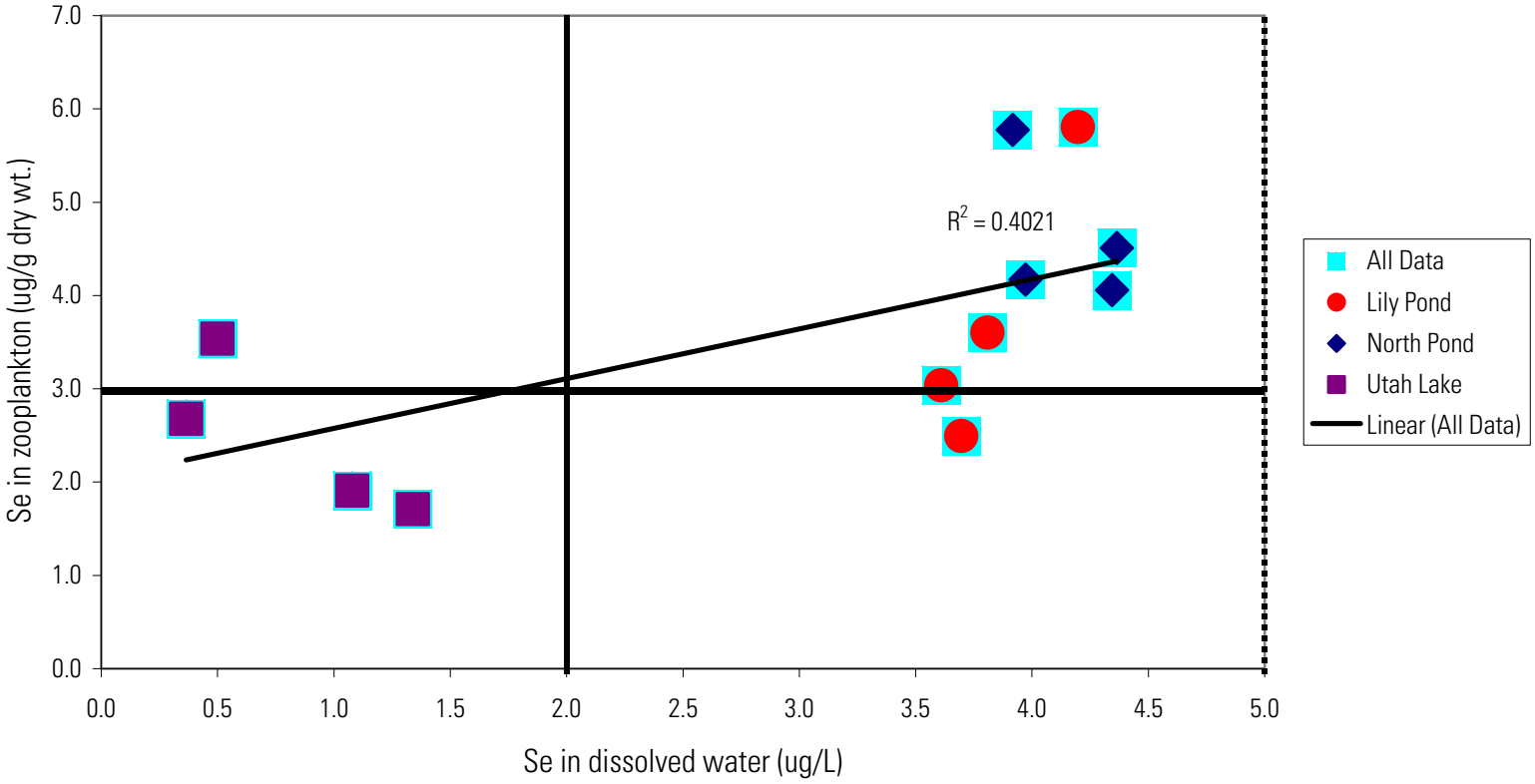

Figure 10. Relationship of selenium concentrations in zooplankton (μg/g dry wt.) and filtered (i.e., dissolved) water (μg/L). The solid horizontal line represents selenium threshold for food-chain organisms proposed by Lemly (1993). The solid vertical line represents lower limit of the selenium threshold for water (2-5 μg/L) proposed by Lemly (1993). The dashed vertical line represents the lower limit of the selenium threshold for water (5-32 μg/L) proposed by DeForest et al. (1999).

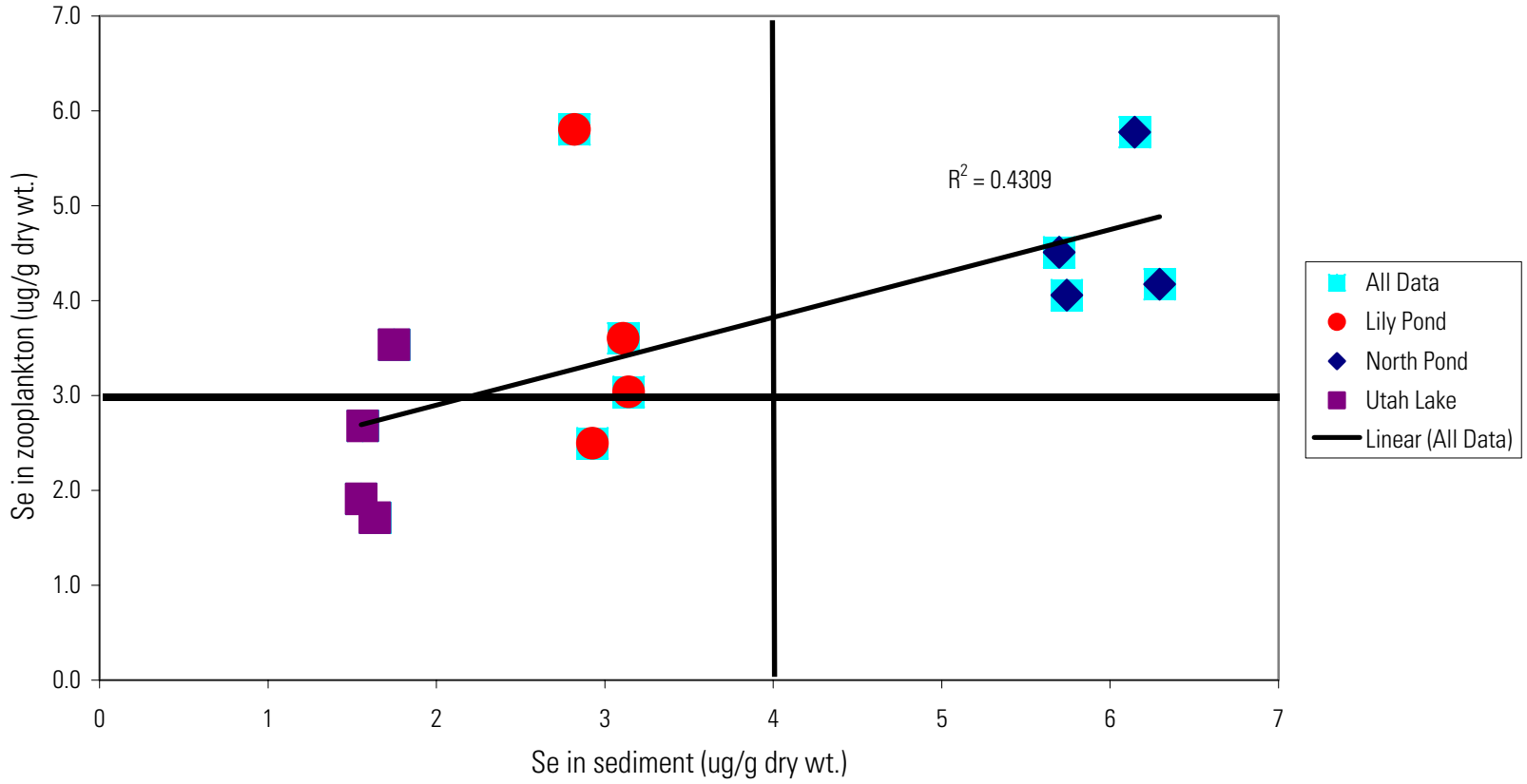

Figure 11. Relationship of selenium concentrations in zooplankton (μg/g dry wt.) and sediment (μg/g dry wt.). The solid horizontal line represents selenium threshold for food-chain organisms proposed by Lemly (1993). The vertical line represents the selenium threshold for sediment proposed by Lemly (1993).

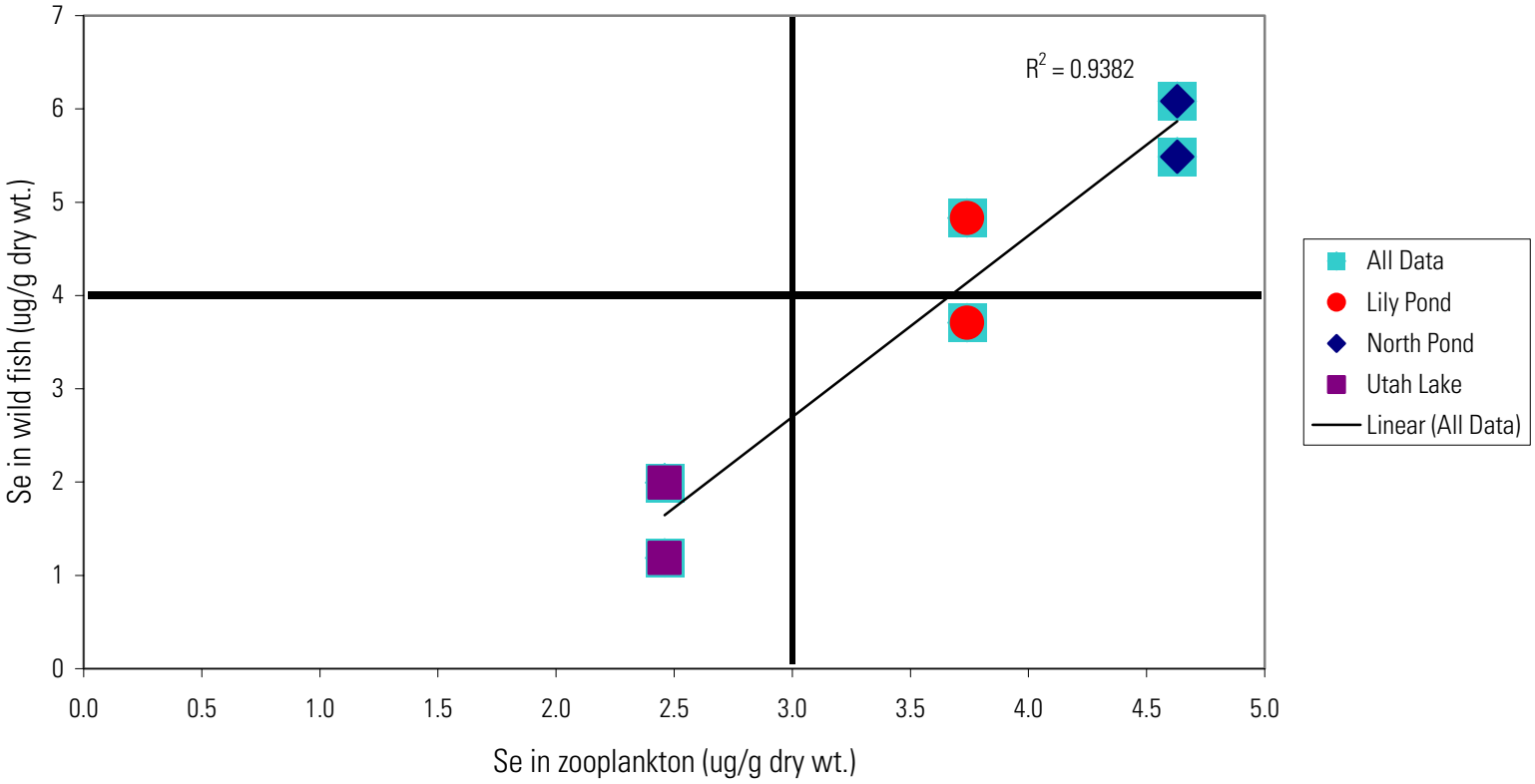

Figure 12. Relationship between selenium concentrations wild fish (μg/g dry wt.) and zooplankton (μg/g dry wt.). The solid horizontal line represents selenium threshold for fish whole-body proposed by Lemly (1993). DeForest et al. (1999) proposed a threshold for warmwater fish of 9 μg/g dry wt. The vertical line represents selenium threshold for food-chain organisms proposed by Lemly (1993). DeForest et al. (1999) proposed a threshold of 10 μg/g dry wt.

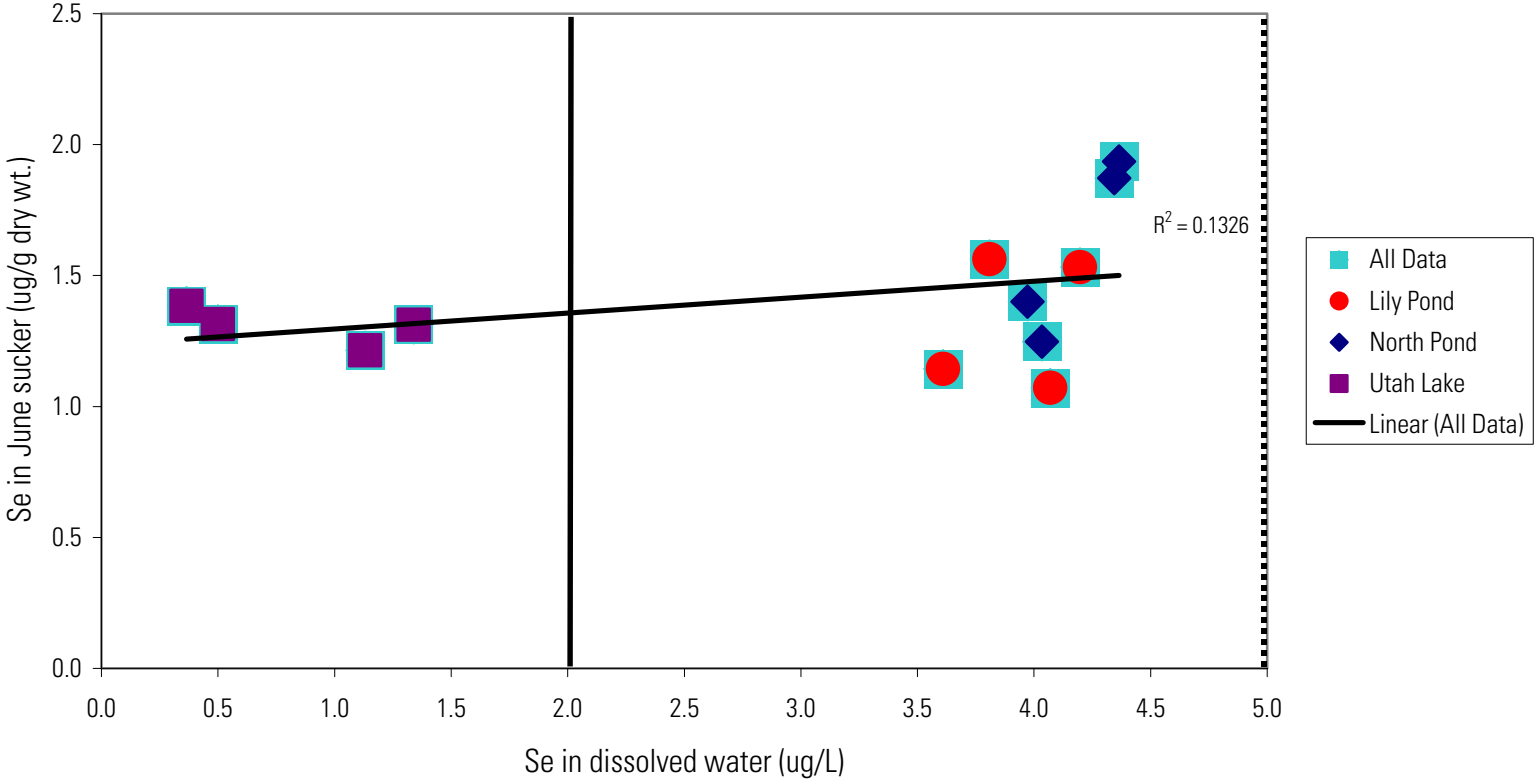

Figure 13. Relationship between selenium concentrations in June sucker (μg/g dry wt.) and filtered (i.e., dissolved) water (μg/L). Lemly (1993) proposed a selenium threshold for whole-body fish of 4 μg/g dry wt. DeForest et al. (1999) proposed a threshold for warmwater fish of 10 μg/g dry wt. The solid vertical line represents the lower limit of the selenium threshold for water proposed by Lemly (1993). The dashed vertical line represents the lower limit of the selenium threshold for water proposed by DeForest et al. (1999).

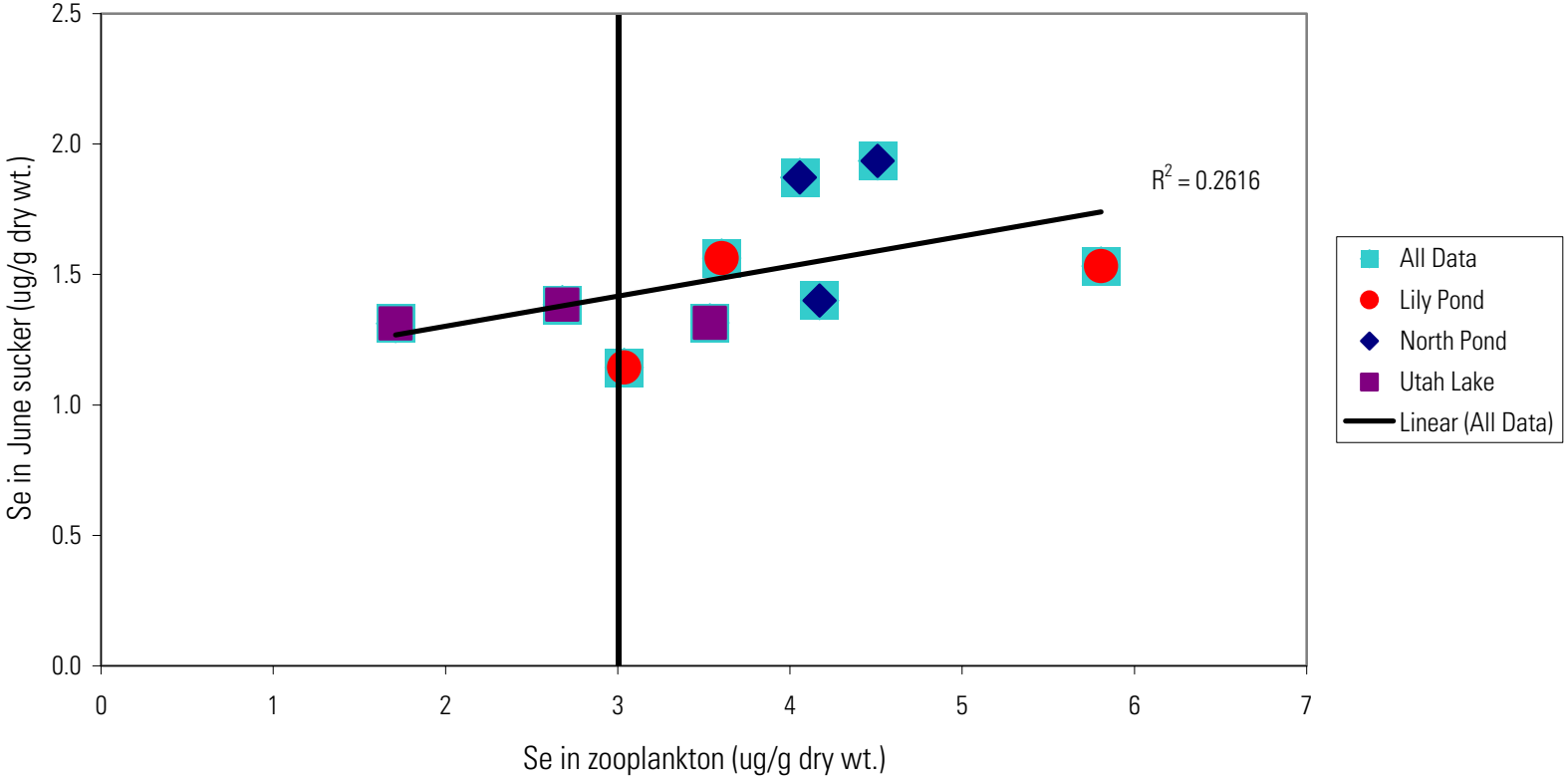

Figure 14. Relationship between selenium concentrations in June sucker (μg/g dry wt.) and zooplankton (μg/g dry wt.). Lemly (1993) proposed a selenium threshold for whole-body fish of 4 μg/g dry wt. DeForest et al. (1999) proposed a selenium threshold for warmwater fish of 9 μg/g dry wt. The vertical line represents selenium threshold for food-chain organisms proposed by Lemly (1993). DeForest et al. (1999) proposed a selenium threshold for food-chain organisms of 10 μg/g dry wt.