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A Site-Specific Assessment of the Risk of Ammonia to Endangered Colorado Pikeminnow and Razorback Sucker Populations in the Upper Colorado River Adjacent to the Atlas Mill Tailings Pile, Moab, Utah

Final Report to the

U.S. Fish and Wildlife Service
Division of Environmental Quality
Off-Refuge Contaminant Assessment Program
Salt Lake City, UT 84119

by

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TABLE OF CONTENTS **PAGE**

APPENDICES (Refer to compact disk located in envelope, back cover)

EXECUTIVE SUMMARY

The Atlas Mill Tailings Pile is located adjacent to the Upper Colorado River near Moab, Utah. Milling of ore ceased in 1984 and the Atlas Corporation subsequently declared bankruptcy. The U.S. Department of Energy (USDOE) is the current manager of the site and is evaluating cleanup options that include remediation of groundwater at the site. This reach of the Upper Colorado River was declared as critical habitat for two endangered fish species (Colorado pikeminnow, *Ptychocheilus lucius;* and razorback sucker, *Xyrauchen texanus*) by the U.S. Fish and Wildlife Service (USFWS) because it is one of the few existing areas that contains known spawning and rearing habitats for these fishes. Monitoring data indicates that the groundwater entering the Upper Colorado River is contaminated with ammonia, metals, and radiochemicals. The U.S. Fish and Wildlife Service is concerned that contaminated groundwater from the Atlas Mill Tailings Pile may be impacting endangered fish populations within this critical habitat.

The U.S. Geological Survey (USGS) conducted a site-specific risk assessment to determine if groundwater entering the Upper Colorado River from beneath the tailings pile could impact populations of the endangered Colorado pikeminnow and razorback sucker. Spatial mapping of contaminant plumes in the river was conducted over several hydrologic regimes and seasons from August 1998 to August 2000. Laboratory and field toxicity tests were conducted with Colorado pikeminnow, razorback sucker, and fathead minnow (*Pimephales promelas*) to determine their sensitivity to ammonia and site waters. The effects of contaminated groundwater on macroinvertebrate communities were also assessed.

Results indicated that the Atlas Mill Tailings Pile represents a localized source of groundwater input containing elevated levels of ammonia, metals, and radiochemicals that exceed Utah state water quality criteria during the low-water hydrologic period ranging from August to March. The particular area of concern is a 10-m wide backwater that extends downstream from Moab Wash for a distance of approximately 500 m. The area of contamination varies with hydrologic regime but in general is confined to an area less than 5,000 m^2. Highest observed concentrations of ammonia (1,200 mg/L total ammonia) occur at river discharge conditions of less than 5,000 cfs during the late summer, fall, and winter periods. River discharge above 5,000 cfs, which occurs from April to July, quickly dilutes ammonia concentrations to levels below those of toxicological concern.

Toxicity testing indicated that Colorado pikeminnow, razorback sucker, and fathead minnow were of similar sensitivity to ammonia with 28-d LOEC values for mortality ranging from 2.19 to 4.35 mg/L total ammonia at pH=8.25 and temperature =25 ^{o}C (0.14 to 0.31 mg/L unionized ammonia). Mortality and growth endpoints were similar in sensitivity to ammonia for the three species. An accelerated life-testing model indicated that Colorado pikeminnow could be sensitive to 90-d chronic exposures as low as 0.17 mg/L unionized ammonia. Colorado pikeminnow were more sensitive to ammonia at lower (8 ^{o}C) temperatures than at average conditions (18 ^{o}C). These lower temperatures coincide with the maximum observed ammonia exposures in the river since they occur in late winter. The late summer season is also of concern because primary productivity in backwaters can increase the pH of water to 9.0, which increases the exposures to the unionized form of ammonia. On-site toxicity tests demonstrated that site waters were directly toxic to both the endangered Colorado pikeminnow as well as the standard surrogate fathead minnow. Highest observed field concentrations caused instantaneous mortality in both controlled laboratory and *in-situ* field studies. Comparisons of laboratory and field results indicate that ammonia is the primary contaminant of concern due to high exposures and the rapid onset of toxicity. Metals and radiochemicals, although sometimes elevated above criteria, did not contribute to toxicity. There were no statistically significant differences in macroinvertebrate distributions that could be attributed to contaminated groundwater.

Collectively, the data indicate that the Atlas Mill Tailings Pile represents a localized input of contaminated groundwater that threatens endangered fish species in the area. The current Utah state water quality criteria for total ammonia is 0.71 mg/L (assuming average conditions of pH=8.25 and temperature =25 ^{o}C), or approximately 0.06 mg/L unionized ammonia. The Utah state water quality criteria for ammonia, if met, would be protective of Colorado pikeminnow and razorback sucker by a factor of at least 2 based on conservative toxicity endpoints. Therefore, remediation of groundwater entering the Colorado River to meet existing Utah state water quality criteria for ammonia would eliminate toxicological concerns for Colorado pikeminnow and razorback sucker.

INTRODUCTION

Various sections of the un-impounded portions of the Upper Colorado River above Lake Powell have been declared critical habitat (Fed. Reg. 59:13374-13400) for four endangered fish species: Colorado pikeminnow *(Ptychocheilus lucius)*, razorback sucker *(Xyrauchen texanus)*, humpback chub *(Gila cypha)*, and bonytail chub *(Gila elegans)*. The U.S. Fish and Wildlife Service (USFWS), under the auspices of Section 7 of the Endangered Species Act, must seek to protect these species and determine if any private, state, or federal activities could jeopardize remaining populations of these endangered species.

The abandoned Atlas Mill Tailings Pile, located on the western bank of the Upper Colorado River near Moab, Utah, is a perceived threat to endangered fish species of the Upper Colorado River (USFWS 1998). This tailings pile lies in the immediate vicinity of critical habitat for both the Colorado pikeminnow and the razorback sucker. The U.S. Department of Energy (USDOE), in cooperation with other federal and state agencies, is currently evaluating several options for long-term management and remediation of the tailings pile (e.g., capping in place, removal, etc.) based on environmental, economic, and legal factors. In 1998, the U.S. Fish and Wildlife Service requested that the Columbia Environmental Research Center (CERC), Biological Resources Division (BRD), U.S. Geological Survey (USGS), Columbia, MO conduct research to determine the potential adverse impacts of the tailings pile to endangered fish species of the Upper Colorado River. This final report presents the background information, research results, and conclusions derived from this research.

History of the Atlas Mill Facility

The Atlas Mill Tailings Pile is located on the west bank of the Upper Colorado River in the 100-yr flood plain. The property and facilities were originally owned by the Uranium Reduction Company and regulated by the U.S. Atomic Energy Commission, precursor to the U.S. Nuclear Regulatory Commission (USNRC). The Atlas Corporations acquired he mill and tailings pile in 1962. Milling of ore at the Atlas site resulted in a large tailings pile located approximately 230 m from the west bank of the Upper Colorado River and 3.7 km northwest of Moab, Utah. The pile occupies about 53 ha of land and is about 0.8 km in diameter and 30 m high. The pile rises to an elevation of 1,237 m above mean sea level with a height of about 27 m above the surface of the Colorado River terrace (USFWS, 1998).

Drainage from the pile was estimated by Oak Ridge National Laboratory in Grand Junction, Colorado (ORNL/GJ) to be between 25 and 75 L/min and could take up to 270 years to dissipate; similarly, it is estimated that concentrations of contaminants in the adjacent groundwater will not reach a steady state for approximately 240 years (ORNL, 1998a). The groundwater contamination plume extends beyond the Atlas property to the south and is over 1,700 m wide and 10 m deep and discharges directly into the Colorado River (ORNL, 1998b). The plume for some contaminants (e.g., ammonia, uranium, molybdenum and nitrates) is mature and these constituents have been discharging to the river since the early 1970's (ORNL, 1998c). The U.S. Fish and Wildlife Service believes that for other contaminants (e.g., selenium), the plume has not fully reached the bank of the Colorado River (USFWS, 1998).

Atlas Corporation ceased operation of the mill and ore milling in 1984. Atlas Corporation activities at the Atlas site were covered by USNRC Source Material License SUA-917 and regulated under the Title II Uranium Mill Tailings Radiation Control Act of 1978. The Atlas Corporation previously planned to close and reclaim the site. However, in 1998, the company declared bankruptcy and was not able to complete a Corrective Action Plan (CAP) for approval by the USNRC. Thus, the remedial action plan for the site remains incomplete. In 2001, the U.S. Congress transferred management and remediation of the former Atlas facility from the USNRC to the USDOE. The USDOE is currently working with stakeholders to evaluate options for short-term stabilization, groundwater remediation, and ultimate cleanup of the site.

Significance of Research to the USFWS and Other Management Agencies

Colorado pikeminnow populations now only occupy a portion of historical habitats in the Upper Colorado River Basin in Colorado, New Mexico, Utah and Wyoming (USFWS, 1996). The most important rearing area in the Colorado River for young-of-year Colorado pikeminnow is between Moab, Utah and the confluence with the Green River (USFWS, 1996). The Colorado River Fisheries Project implemented an Interagency Standardized Monitoring Program in 1986 to monitor population trends of the Colorado pikeminnow in the Colorado River Basin. Low numbers of Colorado pikeminnow (between 1 and 28 fish) were consistently collected between 1986 and 1996 near the Atlas mill tailings site between River Miles 68-49. Both adults and sub-adults were collected in Moab Wash and directly below the tailings pile. Sampling results indicated that anywhere from 0 to 53 young-of-year Colorado pikeminnow were captured at individual sites between River Miles 48-84 (Osmondson et al., 1997). In a mark-recapture study of adult Colorado pikeminnow, 21 of 51 (41%) fish in this sampling reach were caught in the Moab Valley area between River Miles 57 and 65 (Osmundson et al., 1997. The Atlas Mill Tailings Pile site is located at the top of the Moab Valley at River Mile 64 and thus lies within the reach of habitats documented to contain current populations of Colorado pikeminnow. No razorback suckers have been captured in the Upper Colorado River in many years.

A potential spawning site for Colorado pikeminnow exists upstream of the Atlas site above Westwater Canyon. Larval Colorado pikeminnow are consistently found from above Moab to the confluence of the Colorado River with the Green River. This includes the Upper Colorado River section in the vicinity of the Atlas Mill Tailings Pile. The geomorphological and hydrological characteristics of the Upper Colorado River significantly change in the Moab Valley and produce shallow, low velocity nursery habitat for larval and young-of-year Colorado pikeminnow. These observations have led to the declaration of these waters as critical habitats for the species (USFWS, 1998).

The USFWS Utah Field Office has been assessing the proposed reclamation of the Atlas Mill Tailings Pile since 1983. At that time, the Utah Field Office expressed its concern in a letter to the Assistant Regional Director concerning a review of the Emergency and Remedial Response Information System Inventory and identified concerns about possible effects on Colorado pikeminnow and razorback sucker. On June 26, 1997, the Service issued a draft jeopardy biological opinion (DBO) to the USNRC. Since issuance of the DBO, the USFWS has been working with the USDOE, the U.S. Environmental Protection Agency (USEPA), the State of Utah, and Trustees to resolve the issues and determine the best means of reclamation of the site. The USFWS issued a revised draft biological opinion (RDBO) on April 14, 1998 to the Region 6 Regional Office. The RDBO concluded jeopardy to the four endangered Colorado River fishes from the contaminated leachate leaking into the Colorado River from the tailings pile. The RDBO included three reasonable and prudent alternatives to avoid jeopardy: (1) expedite planning and implementation of a groundwater corrective action plan; (2) defer the decision on capping the pile until expeditiously arranged bioassay studies could be conducted to more effectively determine cleanup levels required to remove jeopardy to listed species; and (3) payment of a depletion fee to the Colorado River Recovery Program to offset the impacts of the 154.3 acre-ft water depletion identified for the proposed action (USFWS, 1998). Data collected by ORNL further supports the USFWS RDBO in concluding that the Atlas Mill Tailings Pile is a site-specific point source of ammonia and that remedial activity that does not address groundwater quality will jeopardize endangered fish species due to the continued leaching of contaminated groundwater into the Colorado River.

The RDBO was based on the best available data and opinion of USFWS resource professionals. The USFWS contends that continued input of contaminated groundwater to the river is sufficient to jeopardize populations of endangered Colorado River fishes because the Atlas Mill Tailings Pile is located within a critical fish nursery area (USFWS, 1998). All three constituent elements of designated critical habitat for Colorado pikeminnow and razorback sucker will be adversely modified: 1) water that is of good quality; 2) physical habitat potentially habitable by fish during all life stages; and 3) a biological environment capable of providing a food supply for the endangered fishes (USFWS, 1998). The Service contends that any proposed reclamation project that does not include groundwater remediation will therefore continue to jeopardize fish populations.

The development of a corrective action plan is dependent on a determination of the spatial and temporal patterns of contaminant exposure to endangered fish in the area. In addition, directed studies were needed to determine the relative sensitivity of endangered species to contaminants of concern. This report describes the results of two years of studies that determined the sensitivity of Colorado pikeminnow and razorback sucker to ammonia and contaminated surface and groundwater. In addition, we conducted seasonal spatial mapping of contaminants. These results are provided to the USFWS to assist the USDOE and other federal and state agencies in developing effective remedial action plans for the site in order to reduce any potential contaminant impacts on remaining populations of endangered fishes in the Upper Colorado River.

OBJECTIVES

This study had four objectives:

1) Conduct spatial mapping to determine the distribution of ammonia, metal, and radiochemical concentrations in the Upper Colorado River adjacent to and below the Atlas Mill Tailings Pile in order to estimate exposures to endangered fishes;

2) Conduct toxicity testing with early life stages of fathead minnow, Colorado pikeminnow, and razorback sucker to determine relative species sensitivity to ammonia;

3) Determine if contaminated groundwater was impacting macroinvertebrate distributions adjacent to the Atlas Mill Tailings Pile; and

4) Compare the toxicity of ammonia, metals and radiochemicals to measured environmental concentrations to conduct a site-specific risk assessment.

METHODS

Site Mapping of Contaminant Exposures

In 1998, we mapped the area in a regular grid framework using a PLGR Global Positioning System (GPS) Receiver (Rockwell International Corporation, Cedar Rapids, IA) to establish a study area extending from 1,000 m above to 1,000 m below the Moab Wash (Figure 1). Samples were taken routinely at the edge of the shoreline (referred to as nearshore) and at lateral positions located 1 m, 5 m, 10 m, 30 m, and mid-channel locations perpendicular to the shoreline. Site descriptions are provided in Table 1. This sampling grid was used for all subsequent sampling in years 1999-2000. The Moab Wash was selected as the delineation of the upper end of the exposure area based on previous sampling by other agencies (Utah Department of Environmental Quality, Utah DEQ, 1996; ORNL/GJ, 1998a, 1998b). Sites upstream and on the east side of the river from the tailings pile were established as reference sites. Field sampling was conducted during the following six periods: August 1998; February 1999; June 1999; September 1999; February 2000, and August 2000. Field sampling dates were chosen to span the range of hydrologic conditions in the river in order to characterize the relative contribution of ground and surface water inputs to localized conditions. Note that sample sites were always located at the same relative distance from Moab Wash based on standard GPS locations; however, the lateral distance from the toe of the bank differed seasonally due to the formation of point bars or decreased water levels (i.e., nearshore samples were taken at the edge of the river). Daily estimates of river discharge were obtained from the USGS stream-flow gaging station located at Cisco, UT (latitude N38°48'38" longitude W109°17'34" NAD27; drainage 24,100 sq. miles). Hydrologic data can be viewed at http://waterdata.usgs.gov/ut/nwis/inventory/?site_no=09180500.

Sample Collection and Analysis

Water samples were taken as hand grabs (surface samples) or using a 4-L Beta Bottle (bottom samples; Wildlife Supply Co., Saginaw, MI). Groundwater was sampled from porewater pits dug into the shoreline approximately 0.5 m from the edge of the river. Pits were dug (0.5 m wide; 0.25 m depth) at an elevation slightly above river level to ensure that the pit was filled with incoming groundwater and that the pit was not contaminated from river water or wave action. Once dug, the pit walls were stabilized using an acid-washed polyethylene cylinder covered with polyethylene sheeting to minimize surface contamination, sunlight, and thermal alteration. In most cases, groundwater entered rapidly due to the sandy nature of the alluvium. Groundwater entry was readily evident due to the rapid clearing of water in the pit entering from the up-bank slope. The presence of groundwater in the sample pits was demonstrated by the sharp contrast in water quality variables including increased ammonia and conductivity, and decreased pH and temperature conditions compared to adjacent river water. All groundwater samples were collected as grabs by submerging a polyethylene sample bottle below the surface of the water.

In August 1998, we collected both surface and bottom samples at each grid intersection to determine if there were depth-related differences in water quality. The results of this sampling indicated that river waters were well mixed. Thereafter, all water samples were collected as bottom samples since they were most likely to reflect groundwater input. All water samples, once collected, were placed on ice or under refrigerated conditions ($\leq 4\ ^{o}C$) until analyzed for ammonia, metals, and radiochemicals.

On all sample dates, both groundwater and surface water samples were analyzed *in situ* for temperature, pH, dissolved oxygen and conductivity using a Hydrolab DataSonde 3 Multiparameter Water Quality Instrument and Surveyor 4 Data Display (Hydrolab Instruments, Austin, TX). In August 1998, we deployed Hydrolab DataSonde 1 Multiparameter Water Quality Instruments in the river at four sites (MW-5m; D2-5m; D6-5m, and D6-10m) over a 48-h period to evaluate diurnal changes in water quality (i.e., temperature, pH, conductivity, and dissolved oxygen) that might affect the speciation and toxicity of ammonia. In August 2000, we deployed Hydrolab DataSonde 3 Multiparameter Water Quality Instruments in the river at two sites (D6-5m and D8-5m) over a 48 h period to evaluate diurnal changes in water quality.

Hardness and alkalinity were analyzed via titrimetric methods (APHA, 1995). Ammonia was analyzed as total ammonia using a Technicon Autoanalyzer II System using a salicylate/nitroprusside colorimetric reaction (detection limit 0.1 mg/L total ammonia-N). Most samples for ammonia were analyzed the same day of collection; in cases where this was not possible samples were immediately preserved using ultrapure H_2SO_4 (Aldrich Chemical Company, St. Louis, MO) to pH < 2 (APHA, 1995). All ammonia samples were analyzed within the 30-d regulatory criterion for sample integrity (APHA, 1995). Ammonia concentrations were calculated based on a five-point standard curve. Precision and accuracy were determined based on triplicate analysis of independent, certified ammonia standards on each day. Calculations of unionized ammonia were based on the algorithm of Thurston et al. (1977) using only field-measured *in situ* pH values as described above. All total and unionized ammonia concentrations were expressed as mg/L as N.

Quality assurance summaries of the ammonia analyses are presented in Table 2, Table 3, and Figure 2. A total of 312 quality assurance samples for ammonia were analyzed over the course of two years with an average recovery of $99.91 \pm 6.16\%$. In August 2000, we submitted eleven samples (August 2000 field samples) to Energy Labs, Inc. (Billings, MT) for verification of the method. Results revealed close correspondence between the two methods. Relative percent difference between the laboratories ranged from 0-13% (Table 2). A plot of the data indicted that the data were highly correlated (Figure 2). In addition, a total of 16 independent quality assurance samples were measured at CERC during this interval and are summarized in Table 3. These data again reveal that the method was accurate within 6% of the actual ammonia concentrations.

Water samples for analysis of dissolved metals (ICP-MS analysis of 30 metals) were filtered using a 0.45 μm polycarbonate membrane. Dissolved metal samples were acidified to pH < 2 using ultrapure nitric acid (Aldrich Chemical Co., St. Louis, MO). Water samples for gross alpha radiation, gross beta radiation, and gamma radiation were not filtered or acidified. Whole sediment samples for analysis of total metals and radiochemicals (1-L sample) were taken using a polyethylene scoop (shallow samples and shore soil samples) or a stainless steel Petite Ponar dredge (Wildlife Instruments Co., Saginaw MI). Samples were decanted to remove excess water, and tightly capped. All sediment and water samples for analysis of metals and radionuclides were placed on ice (temperature < 4 ^oC) and shipped for overnight delivery to the USEPA National Air and Radiation Environmental Laboratory (NAREL) in Montgomery, Alabama.

Mercury was analyzed using automated cold vapor atomic adsorption (AV) according to NAREL Method 7471; other metals were analyzed by inductively coupled plasma mass spectrometry (ICPMS) according to NAREL Method 3051/6020. Quality assurance data for metals (spike and blank recoveries) are presented in Appendices 1 and 2. Methods detection limits and control limits varied depending on individual metal constituents. Metals were only reported for those that met control limits of 75-125% recovery of spikes for each sample set (see individual appendices for data qualifiers). Data qualifiers are listed in the data appendices as a U (analyzed but undetected) or B (less than the minimum reporting level but greater than the instrument detection limit).

Gross alpha and gross beta radiation were measured using the NAREL ALPBET procedure. The NAREL procedure involves subtraction of an instrument background measurement from the gross sample measurement. Both values are positive; however, random measurements in the two variables can cause the gross value to be less than the background and result in a measured activity that is less than zero. Although negative values have no physical significance, they do have statistical significance in the comparison of samples from multiple sites. Thus, the NAREL results are provided whether positive, negative, or zero and are presented with a 2-sigma measurement of uncertainty and a sample-specific estimate of the minimum detectable concentrations (MDC). The activity, uncertainty, and MDC are given in the same units and are presented with each sample in the Appendices.

Gamma constituents were measured using gamma spectroscopy. The activity and 2-sigma uncertainly for a radionuclide are reported only if the radionuclide is detected by the procedure; thus, unlike the gross alpha/gross beta analysis, the results of the gamma data are never negative or zero. Radionuclides that are not detected are not presented in the Appendices with the exceptions of Ba-140, Co-60, Cs-137, I-131, K-40, Ra-226, and Ra-228. If one of these seven radionuclides is undetected, the data is reported as "Not Detected" (ND) along with a sample-specific estimate of the MDC (see appropriate Appendices).

Laboratory Toxicity Testing

Larval and juvenile fathead minnows were purchased from Aquatic BioSystems (Fort Collins, CO). Larval Colorado pikeminnow and razorback sucker were provided by Roger Hamman of the Dexter National Fish Hatchery (USFWS, Dexter, NM). Toxicity testing was conducted using standard procedures of the USEPA (USEPA, 1994) and the American Society for Testing and Materials (ASTM, 1997).

Laboratory toxicity testing with ammonia was conducted using ammonium chloride (J.T. Baker Chemical Company, Phillipsburg, NJ). Studies varied in length ranging from 72 h to 28 d. Larval fish were tested in 1000-ml beakers (800 ml test water). Juvenile fishes were tested in either 4 or 19-L glass test containers depending on the relative size of the fish to maintain biomass levels of < 0.5 g/L (USEPA, 1994; ASTM, 1997).

In all studies (with the exception of tests to discern effects of temperature) test containers were maintained at constant temperature (25 ^{o}C) under a 16h:8h light:dark photoperiod. Test concentrations were renewed daily by siphoning approximately 80% of the water from each beaker prior to replacement with fresh solution. Total ammonia was measured daily in both newly renewed and removed test waters to determine the accuracy and precision of the ammonia exposures. Temperature (YSI Model 54 Meter), dissolved oxygen (YSI Model 54 Meter), and pH (Orion Model 940 Meter) were measured daily in the control, low, medium, and high treatment concentrations prior to renewal (e.g., 24-h old exposure water) in addition to the newly mixed test waters. Unionized ammonia, the toxic form, was calculated based on temperature and pH according to Thurston et al. (1977). Alkalinity, hardness, and conductivity were measured in the control, low, medium, and high concentrations of test waters at the beginning and end of the tests to verify that water quality conditions were stable. All water quality measures were conducted using CERC Standard Operating Procedures, which are developed in accordance with methods recommended by the APHA (1995) and manufacturers' recommendations.

Fish were fed brine shrimp nauplii *ad libitum* two times per day at least 6 h apart. Fish growth was determined at weekly intervals in 28-d chronic tests by measuring lengths of individual fish from digitized photographs. At the end of the study, fish were euthanized with tricaine methanesulfonate (MS-222) and immediately dried (60 ^{o}C) and weighed for final weights. The following studies were conducted according to individual stated objectives:

Sensitivity of larval and juvenile endangered fish to ammonia

A series of tests were conducted to establish general species sensitivity; sensitivity of various lifestages; and the time-dependant effects of ammonia on toxicity. The tests consisted of standard 96-h and 28-d procedures as described by USEPA (1994) and ASTM (1997). Three replicate test containers of each test concentration were used. Endpoints included mortality and growth.

Effects of water quality and site-specific conditions on the toxicity of ammonia

Studies were conducted to determine if the source of water (i.e., site-specific conditions) had an effect on the toxicity of ammonia. These studies were conducted because hardness levels are known to alter the sensitivity of fish to ammonia (USEPA, 1999).

In July 1998, approximately 200 L of Colorado River water was collected at the Hwy 191 Bridge located 1 km above the Atlas Mill Tailings Pile (i.e., low in ammonia) and was shipped on ice ($\leq$ 4 ^{o}C) in polyethylene carboys to the CERC. Water was stored at $\leq$ 4 ^{o}C until use. Four days prior to the study 60-d old Colorado pikeminnow and larval fathead minnow were acclimated to respective test waters (either CERC well water or Colorado River water). Ammonium chloride was delivered in a 50% dilution series ranging from 0-64 mg/L (total ammonia as N) consisting of eight concentrations (e.g., 64, 32, 16, 8, 4, 2, 1, and 0 mg/L); each concentration was tested in triplicate. Larval fathead minnow (< 48-h old) and juvenile Colorado pikeminnow (approximately 60-d old) were tested in side-by-side experiments in well water (72-h exposure) using the same experimental design to test the effects of ammonia across species and water sources. Exposure waters were renewed daily. The 72-h test duration was selected due to logistical problems in obtaining sufficient amounts of water for longer testing. Juvenile Colorado pikeminnow were tested in this study because the larval life-stage was not available at test time. Test temperature was maintained at 25 ^{o}C. Water quality was monitored daily as described above.

In a second test the effects of site-specific water quality conditions were tested at the CERC in August 2000. Water was collected at Highway 191 (reference site) and approximately 280 m downstream of Moab Wash (near site D6) and transported to the CERC in 220-L barrels. Water was serially diluted (range 0–60 mg/L total ammonia) to test the effects of ammonia on juvenile Colorado pikeminnow and larval fathead minnows under standard laboratory conditions. Juvenile Colorado pikeminnow and larval fathead minnow were also exposed in CERC well water that was spiked with ammonium chloride (test solutions ranged 0-60 mg/L total ammonia). Test water was renewed daily. Mortality was measured daily over a 96-h period. Temperatures were maintained at 25 ^{o}C. Water quality was measured as described above.

Effects of pH and temperature on sensitivity of Colorado pikeminnow and fathead minnow

We studied the effects of pH and temperature on the sensitivity of juvenile Colorado pikeminnow to ammonia in October (140-d old fish) and November (160-d old fish) of 1999. Studies were conducted in 19-L glass jars each containing 15 L of test water. The pH of test waters was adjusted using diluted 1.0 N NaOH, 1.0 M KH_2PO_4, and 0.5 M H_3BO_3 (ASTM, 1997). Temperatures were adjusted using thermostatically-controlled tank heaters. Tests were conducted at three temperatures (8, 18, and 28 ^{o}C) and three pH levels (8.0, 8.5 and 9.0). Test conditions were selected based on the range of temperature and pH conditions commonly observed in the Upper Colorado River near Moab, Utah. Mortality was assessed at the following exposure times: 15 min, 30 min, 1 h, 3 h, 12 h, 24 h, 48 h, 72 h, and 96 h.

Field Toxicity Testing

Two types of field toxicity tests were conducted to determine the toxicity of site waters adjacent to the Atlas Mill Tailings Pile: on-site mobile laboratory testing and *in situ* cage studies. These studies were conducted and evaluated to determine whether or not there was additive, antagonistic, or synergistic toxicity of ammonia and other constituents (e.g., metals, radiochemicals, etc.) in site waters.

Mobile laboratory toxicity test of Upper Colorado River water, August 1998

In August 1998 a mobile testing trailer was placed at the Canyonlands National Park Headquarters located approximately four miles from the Atlas Mill Tailings Pile. A sample was obtained from a porewater pit dug near Moab Wash. Approximately 100 L of groundwater were placed in collapsible carboys and held on ice ($\leq$ 4 ^{o}C) until test initiation. Use of collapsible carboys insured a minimum airspace to minimize volatility of ammonia and other test constituents. A separate 100-L composite sample was taken near Hwy 191 for use in 50% serial dilutions of the groundwater. Test waters were maintained on ice at $\leq$ 4 ^{o}C until tested. Seven-day static renewal studies were initiated within 24 h of sample collection. Tests were conducted in temperature-controlled water baths (25 ^oC) located in the mobile testing trailer with lighting maintained at a 16h:8h light:dark schedule. Ten juvenile Colorado pikeminnow (90-d old) were tested in each of three replicate 4-L jars per site. Prior to renewal, test waters were poured into clean 4-L test jars and acclimated to test temperature. Mortality, ammonia, pH, dissolved oxygen, and temperature were determined daily. Alkalinity, conductivity, and hardness were determined every other day. Fish were fed brine shrimp *ad libitum* two times per day at least 6 h apart. At the end of the study the fish were euthanized with MS-222.

Mobile laboratory toxicity test of Upper Colorado River water, February 2000

A study was conducted to determine the effect of temperature on the toxicity of ammonia in site waters from below Moab Wash. Ten 10-L water samples were collected from a backwater area located 250 m below Moab Wash (site D5) using a peristaltic pump and placed in collapsible carboys. Reference water was collected above Moab Wash at the Hwy 191 site. Samples were placed on ice and transported to the mobile laboratory located at Canyonlands National Park Headquarters and stored at $\leq$ 4 ^oC until tested. A 4-d static renewal study was conducted using 60-d old fathead minnows using 50% dilutions of the site water. Prior to renewal, test waters were poured into clean 4-L test jars and acclimated to one of two test temperatures (8 ^oC and 25 ^oC) before renewal. Lighting was maintained at a 16h:8h light:dark schedule. Mortality, ammonia, pH, dissolved oxygen, and temperature were determined daily. Alkalinity, conductivity, and hardness were determined every other day. Fish were fed brine shrimp *ad libitum* two times per day at least 6 h apart. At the end of the study the fish were euthanized with MS-222 and immediately dried (60 ^oC) and weighed for final weights. Parallel *in situ* studies were conducted as described below.

In situ toxicity test conducted in Upper Colorado River, February 2000

In situ toxicity of Upper Colorado River water adjacent to the tailings pile was determined using in-river exposures of 60-d old fathead minnows in cages. Polyethylene cages were constructed using modified submersible bait containers (Bass Pro Shops, Springfield, MO). Cages were modified by attachment of a solid polyethylene sampling access door in place of the factory-installed port. In addition, PVC pipe was used to either float the cages at the surface (i.e., air-filled pipe) or sink the cages to the bottom (i.e., sand-filled pipe) to determine the relative toxicity of surface and bottom waters. Fathead minnows were acclimated over a 48-h period to 8 ^oC in the mobile testing laboratory. Fish were transferred to the study site in coolers maintained at 8 ^oC. Ten fish were added to each cage. Cages were deployed at each of 10 sites (three replicate cages per site) at either bottom or surface locations for the 96-h exposure duration. Locations were selected based on previous mapping of both ammonia and conductivity to establish a gradient of exposure conditions. Fish were observed twice daily for mortality or loss of equilibrium. Dissolved oxygen, temperature, pH, conductivity, salinity, and ammonia were measured daily at each cage location (either at the surface or bottom). Care was taken to minimize disturbance of sediments during cage deployment and sampling. The fish were not fed during this study.

Field Assessment of Benthic Macroinvertebrate Distributions

Benthic macroinvertebrates were collected from selected sites in February 1999. Reference sites were selected above Moab Wash and on the east side of the river (Courthouse Wash, UX, U4, E4 and E10) and within the area of potential impact (Moab Wash, D2, D4, D6, D8, and D10). Macroinvertebrate samples were taken with a 0.1-m^2 (35.7-cm diameter) modified Hess sampler fitted with a conical 400-μm mesh nylon net. At each location, one sample was taken at each of three distances from the shoreline: 1 m, 5 m and 10 m. To avoid sampling previously disturbed substrate, all sampling proceeded upstream and away from the bank. The sample area was isolated by pushing the Hess sampler into the substrate by hand. A small hand rake was used to disturb the substrate for approximately three minutes while the invertebrates were collected in the net attached to the downstream side of the sampler. Samples were concentrated in the net and placed into labeled 1-L wide-mouth jars containing 80% ethanol. Samples were transported to CERC for sorting and identification. Aquatic macroinvertebrate samples were sorted in their entirety under a dissecting microscope with 10X magnification. During sorting, organisms were removed from debris and placed into two separate vials: 1) taxa such as midge larvae (Diptera: Chironomidae) that require mounting on glass slides for identification, and 2) those that can be identified with a dissecting microscope and do not require slide mounting techniques. Permanent mounts of midge larvae and their head capsules were made with CMCP-10 mounting media (Masters Chemical Company, Des Plaines, IL), and were allowed to dry for 4-6 weeks before taxonomic identification. All invertebrates were identified to the genus level or the lowest taxonomic level possible. Insects were identified using Merritt and Cummins (1996); non-insect macroinvertebrates were identified using Thorp and Covich (1991) and Pennak (1989).

A sediment sample was taken for particle size analysis from each location where benthic invertebrates were taken. Samples were taken using a plastic scoop that was used to remove the top 4 cm of sediment. Sediment was sieved on site for large particles > 2 mm; the remaining sand, silt, and clay fraction was placed in 1-L HDPE bottles and transported to CERC for analysis. Particle composition (sand, silt, clay) was determined using a hydrometer and the Bouyoucos Method (ASTM, 1997). Depth and current velocity were measured at each site using a Swoffler current meter (Ben Meadows Co., Janesville, WI). Particle size classifications were based on modifications of methods described in Hamilton and Bergersen (1984) and Platts et al. (1983). Size distributions of sediment particles were evaluated as described in Table 4.

Data Analysis

Toxicity data was analyzed using either Probit or non-linear interpolation methods to calculate Lethal Concentration (LC) values depending on the best model fit to the data. Chronic incipient mortality (i.e., predicted 7, 14, 30, 60, and 90-d responses at 0.01, 0.05, 0.10, 0.50, 1.0, and 5.0 % mortality) was calculated using the accelerated life testing procedures of Sun et al. (1995). Lowest Observable Effect Levels (LOECs) and No Observable Effect Levels (NOECs) were calculated using one-way analysis of variance. Analysis of variance and regression analysis were used to analyze macroinvertebrate data. All statistical analyses were conducted 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. Data that were not normally distributed were normalized using either arcsin square root (percentage data), $\log_{10}$ transformations, or non-parametric ranking as recommended in Snedecor and Cochran (1969).

RESULTS

Spatial Mapping of Contaminants

Field assessments of the distribution of ammonia concentrations in the Upper Colorado River adjacent to the Atlas Mill Tailings Pile were conducted on six occasions: August 1998, February 1999, June 1999, September 1999, February 2000, and August 2000. Sample dates were selected to span a continuum of hydrologic and seasonal conditions that control the water quality of backwater areas used by larval and juvenile fishes. The range of hydrologic conditions that occurred during the study is presented in Figure 3. The Utah state chronic water quality criteria for total ammonia are presented in Table 5. Total ammonia criteria vary across pH and temperature due to the shifts in equilibria between the two constituents of total ammonia (unionized ammonia, $NH_{3;}$;and the ammonium ion, NH_4^+). Unionized ammonia (NH_3) is the fraction of ammonia that is toxic to fish. However, laboratory results for ammonia (and therefore criteria) are usually presented as total ammonia since the unionized ammonia/ammonium ion equilibrium is pH-dependant and pH can shift in transport and storage. Comparisons across studies, however, frequently refer to unionized ammonia that is calculated from equilibria equations. For breadth of interpretation, we present data as both total ammonia and unionized ammonia (calculated from the pH equilibrium equations). To further assist the reader, we provide Table 6 that presents the percentages of unionized ammonia at various pH and temperature regimes (from Thurston et al., 1977).

August 1998

Field sampling was conducted over a 7-d period from August 14-20, 1998. River discharge during this period was approximately 3,000 cfs that is typical of the late summer condition when larval and juvenile Colorado pikeminnow are most likely to use shallow backwater areas such as the area adjacent to Moab Wash. For sampling locations refer to Figure 1. Under these hydrologic conditions there is an extensive backwater (approximately 2 ha in size) below Moab Wash. During this time sites D2, D4, and D6 were primarily fed via groundwater whereas sites D8 and D10 received inflow of river water that resulted in dilution of incoming groundwater from beneath the pile.

Concentrations of total ammonia measured at nearshore areas (i.e., in the river at the bank-water interface) were measured at concentrations up to 224 mg/L at site D2 located 100 m downstream of Moab Wash (Table 7; Figure 4); this site was strongly influenced by groundwater entering the river directly from soil fissures located at the tamarisk root line. Concentrations of total ammonia at the bank interface decreased at downstream locations (e.g., site D4, 35 mg/L; site D6, 19 mg/L, and site D8, 5 mg/L). Concentrations of total ammonia were also elevated at the 1-m locations (i.e., perpendicular distance from bank) at these same sites (Figure 4). For example, concentrations of 33, 14, 14, and 4 mg/L total ammonia were measured at sites D2, D4, D6, and D8, respectively. Total ammonia concentrations at the 10-m lateral strata exceeded 1.0 mg/L at only one location (site D2). Thus, it was evident that in nearshore areas ammonia concentrations greatly exceeded Utah state water quality standards (Table 5; 30-d chronic level of 1.21 mg/L total ammonia assuming pH = 8.0 and temperature of 25 ^{o}C) during the sampling period but were confined to a zone of less than 10 m from the shore.

Groundwater samples, taken as seeps from pits excavated at the edge of the river (referred to as pore water) indicated that groundwater emerging from beneath the Atlas Mill Tailings Pile adjacent to the river exceeded Utah state water quality standards for total ammonia by a factor of up to 500 under worst-case conditions. Soil pore waters measured at the immediate confluence of Moab Wash with the Upper Colorado River contained 477 mg/L total ammonia and increased to 685 mg/L total ammonia at site D2 (Table 8). Concentrations in groundwater reached a maximum of 771 mg/L (site D4) (Figure 4). Note that these waters are undiluted groundwater immediately adjacent to the stream.

The influence of groundwater was also evident in an evaluation of standard water quality variables including pH and conductivity (Table 7). Contaminated groundwater reaching the river from beneath the pile characteristically exhibits a decrease in pH and an increase in conductivity. For example, in August 1998 conductivity ranged from 1.01-7.10 mmhos (ambient river approximately 1.10 mmhos) in samples from sites D2 to D20 (Table 7). Conductivities that were measured higher than 0.20 mmhos above ambient river concentrations were highly correlated with total ammonia levels ($r^2 = 0.98$, $p \leq 0.01$). Thereafter, conductivity was used as a surrogate indicator of high ammonia exposures in subsequent field mapping studies because it is rapidly done *in situ* using real-time instrumentation. Temperature and dissolved oxygen remained within levels suitable for survival of Colorado pikeminnow and other fishes. The full complement of routine water quality data for August 1998 is presented in Appendix 3.

Subsequent monitoring of pH indicated that levels of up to 8.69 were measured in two areas near Moab Wash, and were measured at up to pH = 9 in some backwaters during late evening. An increase of pH from 8.5 to 9 (at 25 ^{o}C) would result in a doubling of the percentage of unionized ammonia (the toxic form) under these conditions (Table 6; Thurston et al., 1977). Thus, we evaluated the spatial and temporal changes in pH, conductivity, dissolved oxygen, and temperature to further determine the effect of site-specific conditions on the potential toxicity of ammonia to fish.

Eight continuously recording Hydrolab Water Quality Monitors were deployed at four sites over a 2-d interval: sites MW-5m, D2-5m, D6-5m, and D6-10m. These locations reflected a continuum of groundwater versus river-influenced conditions. Results indicated that both hydrologic sources (groundwater and river) influenced static and diurnal water quality conditions. The influence of water source can be observed by a diurnal graph of conductivity (Figure 5). Highest conductivity occurred in backwater sites with a large groundwater influence (e.g., sites D6-5m and D2-5m) compared to sites with a larger river influence (e.g., sites MW-5m and D6-10m); however, diurnal changes in conductivity did not occur. Dissolved oxygen exhibited strong diurnal changes with a peak at mid-day due to the evolution of oxygen during primary production; overnight, dissolved oxygen decreased due to community respiration (Figure 6). However, there were relatively few differences in dissolved oxygen due to location. In contrast, there were strong spatial and diurnal differences in pH (Figure 7). Highest pH occurred at site MW-5m since it was largely influenced by inflows from the main river; furthermore, diurnal changes were observed at this site with the lowest occurring at night (during CO_2 production via over-night respiration) and highest in the afternoon (due to CO_2 assimilation by algae). Similar diurnal changes occurred at other sites; however, the comparable diurnal pH levels across locations differed due to increasing groundwater influence. For example, the lowest pH occurred at site D2-5m due to the lower pH of incoming groundwater. Temperatures exhibited similar diurnal patterns across sites since depths were standardized and all stations received similar inputs of sunlight (Figure 8).

Concentrations of metals in nearshore surface waters were below Utah state water quality criteria (Table 8). Porewater samples, reflective of incoming groundwater, exceeded Utah state water quality criteria for copper at site D2 (286 µg/L) and Moab Wash (77 µg/L) (Table 8; Figure 9). Manganese concentrations varied spatially with no obvious relationship to the location of the tailings pile (Table 8). A complete listing of metals data is provided in the Appendices.

Radiochemicals were elevated in surface and groundwaters near Moab Wash in August 1998 (Table 8). Utah state water quality criteria for gross alpha radiation (15 pCi/L) was exceeded in nearshore surface water at Moab Wash (54 pCi/L) and site D2 (54 pCi/L) (Figure 10). Gross beta radiation was elevated in nearshore surface water in Courthouse Wash (40 pCi/L) but was below Utah state water quality criteria of 50 pCi/L (Figure 11). Neither Ra226 nor Ra228 were detected in surface water at any site in August 1998.

Radiochemicals in porewater samples (i.e., groundwater) at Moab Wash and site D2 were elevated in both gross alpha and gross beta radiation up to 1700 pCi/L at Moab Wash and site D2 whereas reference measurements (Courthouse Wash, the Island, site E4, and site E10) were low (Table 8; Figures 10 and 11). Ra226 was elevated in pore water sampled in Moab Wash (379 pCi/L), which exceeds the USEPA groundwater criteria of 5 pCi/L (USEPA, 1998) (Figure 12). We detected no Ra228 in soil pore waters (Figure 13). In addition, we observed no spatial relationships between metals or radiochemicals associated with soil or sediment samples due to the alluvial nature of these matrices. Sampling in August 1998 was the first initial sampling done at the site and does not reveal the complete spatial pattern of radiochemical contamination at the site. This was explored further during the February 1999 field sampling discussed below. A complete listing of raw data for metals and radiochemicals samples collected in August 1998 is presented in the Appendices.

February 1999

Field sampling was also conducted from February 22-27, 1999, when discharge of the Colorado River was approximately 3,250 cfs (Figure 3). This hydrologic condition represents the approximate long-term average condition experienced in the area due to the absence of snowmelt and reservoir discharge to the river. This level of discharge results in a significant backwater below Moab Wash.

Concentrations of total ammonia in February 1999 were elevated in nearshore surface waters and pore waters downstream of Moab Wash (Table 9). Total ammonia in nearshore surface waters ranged from 2.17 mg/L (site UX) to 71.5 mg/L (site D6). Nearshore surface waters exhibited moderately elevated levels of conductivity, alkalinity, and hardness at locations where total ammonia was elevated. Total ammonia in pore waters ranged from 2.25 mg/L (site UX) to 665 mg/L (site D6) total ammonia (Table 9). Conductivity, alkalinity, and hardness in pore waters were all elevated in a positive relationship with total ammonia in locations downstream from Moab Wash which further indicates the influence of groundwater emerging from beneath the Atlas Mill Tailings Pile. Dissolved oxygen ranged from 7.27 to 10.50 mg/L in surface waters and from 4.03 to 8.44 mg/L in pore waters. Temperatures ranged from 4.7 to 14.0 °C across surface and porewater sites; however, inter-site comparisons of temperature should not be done because all sites could not be sampled simultaneously and temperatures varied due to time of day.

Total ammonia and unionized ammonia in relation to metals and radiochemicals in nearshore and pore waters are presented in Table 10. Highest levels of total and unionized ammonia in surface waters were observed at sites D6 and D8. Manganese and zinc were increased in nearshore samples at sites D6, D8, and D10; however, copper showed no significant trends in these samples. Only one nearshore surface water sample (Courthouse Wash, 21 μg/L) exceeded the Utah state water quality criteria for copper (12 μg/L); this site was also high in manganese on this date. This was the only sampling date that copper was elevated at Courthouse Wash which indicates that either this was a sampling artifact or a single event discharge. Highest levels of gross alpha and gross beta radiation were observed at site UX (330 and 125 pCi/L, respectively) and exceeded Utah state water quality standards. Gross alpha and gross beta radiation in nearshore surface waters were elevated at all stations below Moab Wash and ranged from 15-78 pCi/L. However, these levels were well below those measured at site UX. Radium 226 in nearshore samples exceeded the Utah criteria of 5 pCi/L at two sites: UX (260 pCi/L) and site D8 (150 pCi/L).

Porewater samples were consistently elevated above Utah state water quality criteria for total ammonia, unionized ammonia, metals, and radiochemicals at sites located below Courthouse Wash (Table 10). However, the spatial relationships of various constituents varied. For example, highest concentrations of total ammonia were measured at sites from Moab Wash to site D10 (range 492 to 665 mg/L total ammonia). Highest concentrations of manganese in pore waters occurred at site D4 (9,450 μg/L) followed by sites Courthouse Wash, D2, D6, and Moab Wash (range 4,470 to 6,790 μg/L manganese). Copper concentrations in pore waters were highest at sites D4, Moab Wash, and Courthouse Wash (range 127-370 μg/L). The high level of copper in pore waters at Courthouse Wash confirmed the elevated concentration observed in surface water. Highest levels of zinc in pore waters were at sites D2-D6 (73-137 μg/L). High gross alpha and gross beta radiation (> 500 pCi/L) were observed at sites UX, Moab Wash, D2, D4, D6, and D10. Spatial variations among these water quality variables reveal the complexity of the groundwater contamination of the site. Ammonia and metals are generally highest downstream of Moab Wash in relation to the location of the Atlas Mill Tailings Pile with occasional exceptions due to unknown sources in the Courthouse Wash tributary. Radiochemicals, however, were elevated above and below Moab Wash. High levels of gross alpha and gross beta radiation were observed at site UX, located approximately 600 m above Moab Wash, reflecting continued contaminated groundwater entry from the original milling site. However, high levels of gross alpha and gross beta radiation were also observed downstream of Moab Wash as far as site D10. These observations are consistent with spatial groundwater measurements made previously in the area surrounding the milling site and tailings pile areas (ORNL, 1998b).

Lateral and longitudinal comparisons of water quality constituents are further presented in Figures 14-20; these data demonstrate the degree to which the main river diluted constituents at the 1, 5, and 10 m lateral distances from shore. For example, ammonia was elevated downstream at site D2 at the nearshore and

1-m strata (Figure 14). Ammonia was also elevated at the 1 and 5-m strata at sites D4, D6, D8, and D10. However, nearly complete dilution was observed at the 10 m distance at all locations (Figure 14). Dissolved copper was elevated to 178 µg/L (15-fold over the Utah criteria) in one surface sample located at site D6 at an apparent upwelling of groundwater at the 5-m location (Figure 15). Similar elevations of gross alpha radiation (1170 pCi/L; Figure 16) and gross beta radiation (1720 pCi/L; Figure 17) were observed at the site D6-5m upwelling. The highest observed gamma constituent in water was Ra226 that was elevated in soil pore waters at sites MW, D2, D4, and D6 (Figure 18) (range 490-920 pCi/L) and greatly exceeds the USEPA groundwater criteria of 5 pCi/L (USEPA, 1998). Concentrations were elevated above 5 pCi/L at sites D4-10m (110 pCi/L), D8-NS (150 pCi/L), and D8-10m (160 pCi/L). A high level of Ra226 was also observed at site E4-1m (210 pCi/L). Levels of other constituents, including Ra228 (Figure 19), were below reporting limits.

There were no spatial trends in total metals in sediment observed in relation to the tailings pile in February 1999. Metals data for soil and sediment are presented in Appendix 11. Highest levels of gross alpha radiation in sediments (47 pCi/g) were determined in shore soil at site U4 that is adjacent to the original uranium milling site (Figure 20). Similar, elevated levels were observed downstream at Moab Wash (45 pCi/g) followed by site D4 (27 pCi/g), UX (24 pCi/g), and D8 (19 pCi/g). Gross alpha radiation decreased with distance from the shore, although remained elevated at lateral distances of up to10 m from shore at site UX and downstream to site D6. Highest levels of gross alpha radiation in "reference sediments" occurred at site E4 locations at 5 and 10 m (17 and 19 pCi/g). Gross beta radiation in sediments was elevated above background (30 pCi/g) at only two shore soil sites (U4 and Moab Wash; 36 and 41 pCi/g, respectively) adjacent and downstream of the milling site (Figure 21). Gamma constituents, including Ra226 and Ra228 were less than 5 pCi/g sediments at all sites in February 1999 (Figures 22 and 23) with the exception of Ra226 at site UX-5m (7pCi/L, Figure 22).

June 1999

Additional field sampling was conducted on June 8, 1999. Discharge of the Colorado River was approximately 15,000 cfs (Figure 3). This discharge is typical of the late spring/early summer hydrologic condition. Discharge during this period typically increases to a peak in late June and early July due to increasing snowmelt. During this season the river recharges localized groundwater and minimizes any influence of contaminated groundwater on localized surface waters. Water levels at this time reached the base of the tamarisk line indicating bank-full conditions. There were no backwaters in the area and current velocities averaged or exceeded 1 m/sec. This made it impossible to conduct porewater sampling of groundwater; only surface waters were sampled. During this season we restricted our sampling to measurements of temperature, pH, dissolved oxygen, conductivity, and ammonia. Neither metals nor radiochemicals were sampled due to the large influence of the river on water and sediment conditions.

Total ammonia in surface waters were near ambient, background river concentrations and ranged from 0.12 mg/L (D8-NS) to 0.40 mg/L (D6-1, 5,10m) during the June 1999 period (Table 11, Figure 24). However, there was relatively little spatial variation in this parameter. Likewise, there was little variation in temperature, pH, conductivity, or dissolved oxygen. These observations are consistent with the high discharge, early snowmelt conditions which result in well-mixed conditions. Conductivity values (approximately 0.5 mmhos) in June were decreased over two-fold from February and is further evidence of the influence of snowmelt as opposed to groundwater. Collectively, the data indicate that groundwater beneath the pile is thoroughly diluted by the river during high-flow periods. Therefore, direct exposure of endangered species is not a concern during this late spring/early summer period.

September 1999

Field sampling was conducted from September 15-19, 1999. Discharge of the Colorado River was approximately 6,500 cfs during this period, which is higher than the average condition for the river during this late summer period (Figure 3). We originally intended to sample the river in August to corroborate August 1998 observations. However, extended high water delayed the sampling. We elected to sample under this condition because it provided additional information concerning the influence of hydrology on localized river condition but preceded the winter, coldwater period. Metals and radiochemicals were not sampled during this time due to cost constraints. However, these constituents were not suspected of being elevated during this time as described below.

Surface water total ammonia concentrations exceeded Utah state water quality criteria levels (approximately 1 mg/L under observed pH conditions) at only three nearshore sites: Moab Wash (15 mg/L), D2 (4 mg/L), and D4 (8 mg/L) (Table 12; Figure 25). Conductivity, alkalinity, and hardness were only slightly elevated above background concentrations at all sites with the exception of Courthouse Wash that is a separate tributary to the river. Temperature and dissolved oxygen showed little spatial variation.

Porewater samples, in contrast, revealed a continued influence of groundwater adjacent to the river below Moab Wash (Table 12, Figure 25). Total ammonia levels ranged from 587 to 1082 mg/L from Moab Wash to site D6; peak total ammonia was observed in pore waters from site D2 (100 m downstream of Moab Wash). Total ammonia in porewater at sites D8 and D10 were significantly lower (13.4 and 0.11 mg/L, respectively). Conductivity, alkalinity, and hardness were also elevated below Moab Wash as predicted from previously demonstrated relationships with total ammonia. One departure observed, however, was the elevation in conductivity in pore water at site

UX that occurred in spite of low ammonia. This continues to reflect, in part, the historical difference in groundwater near the actual milling site compared to groundwater below the tailings pile. Collectively, the data indicates that under discharge conditions of 6,500 cfs there is contaminated groundwater entering the Colorado River adjacent to the pile; however the river dilutes the groundwater to levels below those of toxicological concern.

February 2000

Field sampling was conducted from February 21-27, 2000. Discharge of the Colorado River was approximately 3,300 cfs, and was quite similar to February 1999 conditions (Figure 3). Sampling was conducted to corroborate observations made the previous year and to facilitate supporting, parallel on-site and *in situ* toxicity tests (results provided under Toxicity Test Section). The full complement of water quality, ammonia, metals, and radiochemical sampling was conducted.

Water quality sampling in February 2000 confirmed previous observations made in August 1998 and February 1999: the Atlas Mill Tailings Pile represents a localized input of contaminated groundwater to the backwater area below Moab Wash under low-flow conditions.

Total ammonia in nearshore surface waters ranged from 2 (site D2) to 41 mg/L in the backwater area below Moab Wash (Table 13). Ammonia was highest at site D6 and was observed to exceed Utah state water quality criteria for total ammonia as far downstream as site D10. Total ammonia at site D2 (2.02 mg/L) was comparatively low due to the increase in the size of the point bar below Moab Wash; thus, this sampling location was not in a backwater but rather at the edge of the river and approximately 50 m east of the previous sampling locations. This point bar was unusually large in February 2000 due to the input of red sand from the Moab Wash; this sand originated from construction activity during the installation of an underground gas line in the upper end of the wash adjacent to Hwy 191. Conductivity, alkalinity, and hardness were highest at site D6 in parallel with total ammonia. Total ammonia concentrations at reference locations (Hwy 191, Courthouse Wash, and UX) were less than 1 mg/L (Table 13).

Concentrations of manganese were elevated in surface waters at Moab Wash, D2, D4, D6, and D8 (range 129-410 µg/L) (Table 14). Copper concentrations exceeded the Utah state water quality criteria (12 µg/L) at two sites (D2, 14 µg/L; and D4, 15 µg/L). Zinc concentrations exhibited slight elevations in surface water (range 30-55 µg/L) with the exception of a markedly decreased level (6 µg/L) in Courthouse Wash. Gross alpha and gross beta concentrations were elevated from site UX downstream to site D10 (range 23-143 pCi/L) compared to the reference locations of Hwy 191 and Courthouse Wash.

Total ammonia concentrations in pore waters were elevated from Moab Wash downstream to site D10 and exceeded high concentrations as observed in August 1998 and February 1999 (Table 13). In contrast, total ammonia in pore water at site UX was much higher (64 mg/L) than previously observed. Total ammonia was at background levels at Hwy 191 (< 1 mg/L) and slightly elevated at Courthouse Wash (3 mg/L). Conductivity, alkalinity, and hardness were also elevated in pore waters at Moab Wash and sites D2, D4, D6 and D10.

Metal concentrations in pore waters (manganese, copper, and zinc) increased below Moab Wash compared to reference sites (Courthouse Wash and Hwy 191) but varied spatially (Table 14). Highest variation was observed in manganese, ranging from a low of 633 μg/L (site D8) to a high of 13,000 (site D10). Copper and zinc concentrations in pore water consistently exceeded Utah state water quality criteria at Moab Wash and sites downstream with the exception of site D8 (below criteria). Gross alpha and gross beta radiation exhibited highly elevated levels (32-2,190 pCi/L) at sites UX and downstream. Ra226 was elevated above criteria levels in pore water at 4 sites: UX, MW, D2, and D6. Site D8 pore water was much lower in total ammonia, metals, and radiochemicals compared to sites D6 and D10 which is similar to observations made in February 1999.

Spatial distributions (i.e., lateral range from nearshore to 10 m from shore) of total ammonia, dissolved copper, gross alpha radiation, gross beta radiation, and gamma radiation in February 2000 are presented in Figures 26-35. Total ammonia concentrations in surface waters were much lower than those in pore waters but were consistently elevated above water quality criteria in nearshore locations at sites downstream of Moab Wash (Figure 26). The total ammonia concentration was highly elevated at site D6-5m (1530 mg/L). This site is an apparent upwelling of groundwater during this season and was also previously observed in February 1999. Similar evidence of upwelling at site D6-5m was observed in dissolved copper (Figure 27), gross alpha radiation (Figure 28), and gross beta radiation (Figure 29). Similar evidence of upwelling was documented in August 2000 sampling (described below). Copper exceeded the Utah state water quality criterion of 12 μg/L at seven pore water sites and seven surface water sites (Figure 27). The highest concentration was observed at site U4-1m (64 μg/L). Elevated copper concentrations were also observed at site E10-5m (34 μg/L), which cannot be unequivocally explained. Spatial distributions of gross alpha indicated that groundwater greatly exceeded Utah criteria (15 pCi/L) from sites UG to D10 and reached up to 2,410 pCi/L at site U4 (adjacent to the milling site) (Figure 28). Gross alpha concentrations in the river proper were far lower, but still exceeded Utah criteria as far as 10 m from shore at sites D2, D6, D8, and D10. Highest in-stream levels occurred at the site D6-5m upwelling (1200 pCi/L). Similar observations were made for gross beta radiation in water, including the upwelling phenomena observed at site D6-5m (2,450 pCi/L gross beta) (Figure 29). However, there were far fewer Utah criteria exceedances for gross beta compared to gross alpha. Highest gamma concentrations were observed as Ra226 where groundwater contained up to 697 pCi/L at site MW (Figure 30). Ra226 was elevated in surface waters above the USEPA criteria (5 pCi/L) at D6-nearshore (103 pCi/L), D10-1m (60 pCi/L), UX-1m (164 pCi/L), and U4-1m (170 pCi/L). However, Ra228 was not elevated in any sample (Figure 31).

Total metal concentrations in sediment for February 2000 are presented in Appendix 24. There were no spatial differences in total sediment metals observed in relation to the Moab Wash site since sediments are of alluvial origin and not from the tailings. Sediment concentrations of gross alpha radiation (Figure 32) and gross beta radiation (Figure 33) indicated that samples were only occasionally elevated above background The lone exception to these findings was elevated gross alpha (101 pCi/L) and gross beta (129 pCi/L) observed in Courthouse Wash (Figures 32 and 33). Overall these concentrations are not alarming, and could reflect in part the natural background levels due to geologic formations in the area. No gamma constituents were observed above reporting limits in sediments in February 2000 as indicated in Figure 34 (Ra226) and Figure 35 (Ra228).

August 2000

Field sampling was conducted from August 8-13, 2000. Discharge of the Colorado River was approximately 3,050 cfs (Figure 3), which is similar to the long-term average for this season.

Total ammonia was elevated in nearshore surface waters downstream of Moab Wash and ranged from 1.4 (Moab Wash) to 43.3 mg/L (site D6) total ammonia (Table 15); concentrations at MW and sites D2 were low during this season since they were sampled at the outside of the Moab Wash point bar and were largely influenced by the main river. Total ammonia concentrations generally decreased at distances 1, 5, and 10 m from shore due to mixing (Figure 36). However, concentrations were still elevated at the 1, 5, and 10 m locations at sites D5 and D6 because they were located in a backwater with little river influence. Elevations in total ammonia were associated with increases in conductivity, alkalinity, and hardness. Highest levels of conductivity, alkalinity, and hardness were measured at Courthouse Wash, which also contained elevated ammonia (Table 15). These were the highest measures of these constituents made at Courthouse Wash over the 2-yr study duration. Given the location of this “reference” location these observations must be attributed to sources other than the Atlas Mill Tailings Pile. Measures of temperature and pH reached 27.4 °C and 8.59, respectively, at Moab Wash, and resulted in increased proportions of unionized ammonia in accordance with equilibrium shifts in this backwater due to increased primary production. The pH of surface water at site D6 was the lowest observed due to the influence of groundwater in the nearshore area. However, temperatures, pH, and dissolved oxygen were within narrow and acceptable ranges for fishes.

Total ammonia in porewater samples were highest between Moab Wash and sites D2-D6 (range 519-617 mg/L); total ammonia in pore water was much lower at site D8 (7 mg/L) but increased to 162 mg/L at site D10 as noted in previous low-water conditions (Table 15). Total ammonia was positively correlated with conductivity, alkalinity, and hardness; it was negatively correlated with pH. These relationships are due to groundwater constituents as previously observed during low-water conditions of August 1998, February 1999, and February 2000.

In August 2000, we again measured the diurnal and spatial changes in basic water quality parameters of conductivity, dissolved oxygen, pH, and temperature (Figures 37-40). Two sites were contrasted: site D6-5m and site D8-5m. Results indicated that site D6-5m had a much stronger groundwater influence than D8-5m. Site D6-5m had 15-fold higher conductivity (approximately 17-20 mmhos) compared to site D8-5m (approximately1.0 mmhos) (Figure 37). Slight diurnal variation was observed at site D6-5m with highest levels peaking at near midnight. Diurnal variations in dissolved oxygen were significant at both sites with the daily range higher at site D6-5m (6 mg/L) compared to site D8-5m (2 mg/L) (Figure 38). Peak concentrations at both sites occurred in the late afternoon due to associated increases in oxygen from primary productivity. The relative difference in ranges across sites, however, could be due to several factors related to either differences in net production, oxidation of ammonia to nitrate, or simply dilution by the river. However, studies were not conducted to partition these differences. Site D6-5m exhibited lower levels of pH (range 7-7.1) compared to site D8-5m (range 8.3-8.6) due to the influence of groundwater (Figure 39). This decreased pH is expected in part to decrease the risk of ammonia due to the lower proportion of unionized ammonia that exists with lowered pH (USEPA, 1999). Originally, we were concerned that diurnal primary productivity would increase pH in accordance with basic carbonate chemistry. However, that prediction was in part mitigated by the overall lower pH conditions of the groundwater in areas of greatest total ammonia concentration. These trends can be seen in evaluating the general trends in pH, total ammonia, and unionized ammonia as presented in Table 15. Table 15 clearly indicates, however, that the decrease in pH only partially ameliorates the risk since total ammonia is so high in the Atlas Mill leachate. Temperature trends primarily reflected the combination of daily air temperature fluctuations and the relative degree of exchange of the backwater with the main river (Figure 40). For example, site D6-5m had less exchange with the main river and was shallow water habitat (approximately 30 cm). Thus, incident sunlight led to significantly higher temperature at site D6-5m compared to site D8-5m.

Neither metals nor radiochemicals were measured in August 2000 in either surface or pore water since the data derived from August 1998 and August 1999 were considered sufficient to explain localized conditions during late summer.

Laboratory Toxicity Testing

Nine separate toxicity studies were conducted under laboratory and field conditions (Table 16). There were four objectives of these studies: 1) determine the relative sensitivity of Colorado pikeminnow, razorback sucker, and fathead minnow to ammonia; 2) determine the relative sensitivity of Colorado pikeminnow and razorback sucker to the extant database for ammonia; 3) determine the effects of water quality and site-specific conditions on the toxicity of ammonia to Colorado pikeminnow; and 4) determine the effect of temperature and pH on toxicity of ammonia to Colorado pikeminnow.

Sensitivity of larval endangered fishes to ammonia

The relative sensitivity of larval Colorado pikeminnow, razorback sucker, and fathead minnow were studied using 28-d static renewal exposures ranging from 0 to15 mg/L total ammonia. Endpoints examined included mortality, length, and weight. Note that the Colorado pikeminnow and razorback sucker studies could not be conducted concurrently due to differences in spawning season for each species. Thus, larval fathead minnow tests were conducted each time as the reference species.

<u>Sensitivity based on mortality:</u> The sensitivity of larval fishes to ammonia was measured using 28-d static renewal studies. Colorado pikeminnow, razorback sucker, and fathead minnow exhibited 28-d LC50s of 10.3, 10.1, and 9.75 (range 9.5-10.0) mg/L total ammonia, respectively (Table 17). The 28-d LC50s of unionized ammonia for larval Colorado pikeminnow, razorback sucker, and fathead minnow (corrected for pH and temperature) were 0.72, 0.63, and 0.67 (range 0.64-0.70 mg/L), respectively (Table 17). Time-dependant comparisons of toxicity data based on the LC50 indicated that mortality concentrations causing 50% mortality at days 4, 7, 14, 21, and 28 differed by less than 25% (Figure 17). For example, the LC50 for fathead minnow changed from a range of 12.5-12.6 mg/L total ammonia (range at day 4 to a range of 9.5-10.0 mg/L total ammonia at day 28. Thus, a comparison of the data using the LC50 as an endpoint indicated that the larval Colorado pikeminnow, razorback sucker, and fathead minnow were similar in sensitivity to ammonia (Table 17).

The 28-d Lowest Observable Effect Level (28-d LOEC) of total ammonia that caused significant mortality in larval fathead minnow in the April 1999 test was 4.35 mg/L (0.31 mg/L unionized ammonia) and resulted in 17% mortality (Table 18). The 28-d No Observable Effect Level (28-d NOEC) was 2.23 mg/L total ammonia (0.16 mg/L unionized ammonia). Significant mortality occurred as early as day 14 at 4.35 mg/L total ammonia and day 7 at 8.53mg/L total ammonia (0.60 mg/L unionized ammonia). The highest exposure concentration (13.36 mg/L total ammonia; 0.93 mg/L unionized ammonia) resulted in 67%

mortality by day 7. However, mortality did not significantly increase in any treatment between days 14 and 28 (Table 18).

The second 28-d larval fathead minnow study was conducted in July 1999; larval fathead minnow were more sensitive in this study compared to the April 1999 study. In July 1999 the 28-d LOEC for total ammonia was 2.19 mg/L (0.15 mg/L unionized ammonia) and resulted in 17% mortality (Table 19). The 28-d NOEC could not be determined since 2.19 mg/L total ammonia was the lowest concentration tested. Significant mortality occurred as early as day 7 at 2.19 mg/L total ammonia. The highest exposure concentration (13.70 mg/L total ammonia; 0.97 mg/L unionized ammonia) resulted in 60% mortality by day 7. Mortality only slightly increased between day 7 and 28 (Table 19).

Colorado pikeminnow were less sensitive to a 28-d chronic exposure to ammonia compared to the fathead minnow. The 28-d LOEC for mortality was 13.24 mg/L total ammonia (0.85 mg/L unionized ammonia) at day 28; the 28-d NOEC was 7.74 mg/L total ammonia (0.54 mg/L unionized ammonia) (Table 20). The first statistically significant mortality was observed on day 14 at the highest concentration tested.

The razorback sucker was less sensitive than the fathead minnow but more sensitive than the Colorado pikeminnow based on the 28-d LOEC for ammonia. Razorback sucker exhibited a 28-d LOEC value of 2.33 mg/L total ammonia (0.14 mg/L unionized ammonia); the 28-day NOEC value could not be measured since mortality was measured at the lowest concentration tested (Table 21). Mortality rates were actually similar, but low, at concentrations ranging from 2.33-8.60 mg/L total ammonia (range 10-13.33% mortality). However, the highest concentration tested (13.48 mg/L total ammonia; 0.85 mg/L unionized ammonia) was significantly toxic by day 14 (23% mortality) and reached 100% mortality by day 28.

<u>Sensitivity based on growth</u>: Sublethal toxicity was measured based on fish lengths (days 7, 14, 21, and 28) and final weights (day 28) in 28-d static renewal studies.

The Lowest Observable Effect Concentration (LOEC) for length of larval fathead minnows determined in April 1999 was 13.36 mg/L total ammonia (0.93 mg/L unionized ammonia) whereas the No Observable Effect Concentration (NOEC) was 8.53 mg/L (0.60 mg/L unionized ammonia) (Table 22). Significant effects on growth were not observed until day 14 of the study at the highest concentration (Table 22). The LOEC for weight of larval fathead minnows was 13.36 mg/L total ammonia (0.93 mg/L unionized ammonia) (Table 23), which was equivalent to the concentration causing significant effects on length. Note that statistical differences in weight were measured at lower concentrations; however, these concentrations actually increased weights compared to the control (Table 23).

In July 1999 we again conducted a 28-d static renewal test with fathead minnows and ammonia. The 28-d LOEC for length in July 1999 was 13.70 mg/L total ammonia (0.97 mg/L unionized ammonia) (Table 24); there were no significant effects on weight at the highest concentrations tested (13.70 mg/L total ammonia) (Table 25). Thus, chronic effects on growth in July 1999 were slightly less than those observed in April 1999.

Chronic studies of the effects of ammonia on growth of razorback sucker, conducted at the same time as the April 1999 fathead minnow study, revealed that razorback suckers were slightly more sensitive to ammonia than the fathead minnows. The 28-d LOEC for length was 8.60 mg/L total ammonia (0.53 mg/L unionized ammonia) (Table 26). The 28-d NOEC for length of razorback suckers was 4.53 mg/L total ammonia (0.27 mg/L unionized ammonia). Significant effects on length of razorback sucker were first observed on day 14 in the highest concentration (13.48 mg/L total ammonia; 0.85 mg/L unionized ammonia) (Table 26); however, total mortality eventually occurred at this concentration. There were no significant effects on weight of razorback sucker at the highest concentration in which survival occurred (8.60 mg/L total ammonia; 0.53 mg/L unionized ammonia) (Table 27).

Chronic studies of the effects of ammonia on growth of Colorado pikeminnow (July 1999) revealed that this species was slightly more sensitive than either the fathead minnow or razorback sucker. The most sensitive LOEC for length was observed on day 7 at 7.74 mg/L total ammonia (0.54 mg/L unionized ammonia; the NOEC on this date was 4.17 mg/L total ammonia (0.29 mg/L unionized ammonia) (Table 28). Similar statistical results were obtained on days 14 and 21. However, by day 28 the LOEC for length had increased to 13.24 mg/L total ammonia (0.92 mg/L unionized ammonia). The 28-d LOEC for weight of larval Colorado pikeminnow was 7.74 mg/L total ammonia (0.54 mg/L unionized ammonia) (Table 29). Thus, weight was a 2-fold more sensitive indicator of ammonia exposure than length for the Colorado pikeminnow when comparing 28-d data.

<u>Relative sensitivity of Colorado pikeminnow compared to other species</u>: The results of this study clearly indicate that the Colorado pikeminnow, razorback sucker, and fathead minnow are similar in sensitivity to ammonia. The USEPA (1999; also see references therein) has compared the relative acute and chronic sensitivity of various fish and invertebrate species in the recent development of the revised water quality criterion document for ammonia. These acute and chronic species sensitivity profiles are presented in Figures 41 and 42, respectively.

The acute toxicity of ammonia (adjusted to pH = 8.0 and temperature 25 ^{o}C) ranged from 10-50 mg/L total ammonia for fish and from 15–390 mg/L for aquatic invertebrates (Figure 41). The LC50 value for Colorado pikeminnow (15.24 mg/L; July 1998 toxicity test adjusted to pH = 8.0 and temperature 25 ^{o}C) is at approximately the 20th percentile in fish sensitivity to ammonia, which indicates that it is more sensitive than 80% of the theoretical fish community. The Colorado pikeminnow is much more acutely sensitive to ammonia than most species of aquatic invertebrates, which indicates that a direct, toxic effect to Colorado pikeminnow is more likely than an indirect food chain effect. A precise comparison of the acute toxicity to razorback sucker cannot be made since there was no mortality at the highest concentration tested (15 mg/L total ammonia) at 96h (Table 17). However, the lack of mortality at this concentration indicates that it is not the most sensitive species compared to the cumulative species profile for acute toxicity of ammonia.

The chronic community response profiles for total ammonia (EC20 endpoint; adjusted to pH = 8.0 and temperature 25 ^{o}C) range from 2-9 mg/L total ammonia for fish, and from 2-20 mg/L total ammonia for invertebrates (Figure 42). The EC20 for the Colorado pikeminnow (mortality data from July 1999; adjusted to pH = 8.0 and temperature 25 ^{o}C) is 4.27 mg/L ammonia. The EC20 for the razorback sucker (mortality data from April 1999; adjusted to pH = 8.0 and temperature = 25 ^{o}C) is 2.52 mg/L total ammonia. This indicates that the razorback sucker and Colorado pikeminnow are within the 20-50% range of species sensitivity to ammonia. The data further indicate that the existing Utah water quality criterion of 1.21 mg/L total ammonia (Table 5; 30-d chronic criterion at pH = 8.0, temperature 25 ^{O}C) is protective of the Colorado pikeminnow and razorback sucker by at least a factor of 2 based on the EC20 from 28-d chronic tests (Figure 42).

It should be noted, however, that the EC20 and the LC50 are not appropriate risk assessment endpoints for endangered species. The Endangered Species Act specifies that individual fish must be protected as opposed to 20% or 50% of a population. The chronic LC20 and the acute LC50 are, however, standard toxicological endpoints commonly used in the literature for interspecies comparisons of sensitivity. Calculation of a lethal level necessary to protect individuals of an endangered species requires modeling to predict a level of total ammonia protective of individual fish. This approach is discussed below.

Modeling of chronic effects: To predict the concentration of ammonia protective of individual fish of a Colorado pikeminnow population we applied the time-dependant method of Sun et al. (1995) (Table 30). This method calculates the predicted toxicity at various mortality rates and future dates by modeling the time and concentration relationships derived from acute toxicity testing results. For this modeling effort we used the data from a 4-d static renewal study with 90-d old Colorado pikeminnow (mean pH = 8.1; mean temperature = 25 ^{o}C) to predict concentrations that cause various low rates of mortality over time (e.g., days 7, 14, 30, 50, and 90). For example, the concentration of total ammonia expected to cause 5% mortality (e.g., EC5) at 7, 14, 30, 60, and 90 days would be 15.40, 13.60, 11.90, 10.60, and 9.8 mg/L total ammonia, respectively, comparable unionized ammonia toxicities, calculated from pH and temperature-corrected data, are 1.02, 0.91, 0.79, 0.70, and 0.65 mg/L unionized ammonia, respectively over the same time interval (Table 30). Estimates of ammonia concentrations causing mortality of 0.01% of fish in a population (i.e., 1 in 10,000 fish) at 7, 14, 30, 60, and 90 days would be 4.16, 3.67, 3.22, 2.87, and 2.66 mg/L total ammonia, respectively; comparable unionized ammonia toxicities would be 0.27, 0.24, 0.21, 0.19, and 0.17 mg/L unionized ammonia, respectively over the same time interval (Table 30). These are modeled data, but based on a published, peer-reviewed model. Although the data indicate an increasing rate of progression of mortality over time, it is important to note that the existing Utah state water quality criteria for ammonia (0.8 mg/L total ammonia at pH = 8.1, T = 25 ^{o}C; from Table 5) would be protective of Colorado pikeminnow at mortality rates as low as 0.01% at 90-d exposure (i.e., 3X safety factor). Thus, the existing water quality criterion for ammonia, although calculated from a fish community perspective to protect 95% of species, would be protective of individuals of a Colorado pikeminnow population based on the mortality endpoint.

Attempts to use the model of Sun et al. (1995) to model chronic effects on razorback sucker were not successful due to insufficient mortalities early in the study.

Effects of water quality and site-specific conditions on the toxicity of ammonia

In 1999, the USEPA (USEPA, 1999) published a revision of the water quality guidelines for ammonia. This revision recommended that site-specific testing of ammonia toxicity should be conducted where endangered species may be exposed to determine the effects of localized water quality conditions on ammonia toxicity. Thus, we conducted two studies comparing the sensitivity of Colorado pikeminnow to ammonia in well water in addition to actual site waters from the Upper Colorado River.

In July 1998 we conducted a study to compare the relative sensitivity of 60-d old Colorado pikeminnow exposed to ammonia (added as ammonium chloride) in Colorado River water compared to CERC well water. The Colorado pikeminnow was less sensitive to total ammonia when exposed in Colorado River water (72-h LC50 = 31.0 mg/L) compared to exposures conducted in CERC well water (72-h LC50 = 12.3 mg/L total ammonia (Table 31). However, when the data were compared based on unionized ammonia there was little difference (Colorado River water 72-h LC50 = 1.39 mg/L unionized ammonia; CERC well water 72-h LC50 = 1.62 mg/L unionized ammonia) (Table 31). The apparent difference in sensitivity to total ammonia among varying water qualities is explained by the influence of pH, which controls the equilibrium of ammonia between the ionized and unionized form. Unionized ammonia (NH_3) is far more toxic than the ionized (NH_4^+) form and the fraction of unionized ammonia increases with pH (Thurston et al., 1977). Criteria for ammonia are usually expressed as total ammonia due to changes in equilibrium that can occur during sampling, transport, and analysis. In addition, most methodologies that analyze ammonia use the unionized form as the analyte. However, criteria are adjusted using a pH nomograph that accounts for the effect of *in situ* pH and temperature. Thus, in our studies we found apparent variation in sensitivity to total ammonia varied with water quality, but that the variation was explained by the pH-driven ammonia equilibrium reactions. This topic was further explored in separate studies that evaluated the effect of pH and temperature described below.

In August of 2000 we conducted another study to assess the effect of water quality on the sensitivity of both Colorado pikeminnow and fathead minnow to ammonia. Both species were exposed to each of two water types: 1) Colorado River water obtained from near site D6 (65 mg/L total ammonia) that was serially diluted with reference water (Colorado River at Hwy 191; no ammonia); or 2) CERC well water spiked with ammonium chloride (high nominal concentration as 65 mg/L total nitrogen as N). The source of dilution water had a significant effect on the toxicity of ammonia to fishes. For example, total ammonia was more toxic to both species when diluted with CERC well water compared to Colorado River water (Table 32). Furthermore, the onset of toxicity of ammonia to fathead minnow was more rapid in CERC well water. Significant mortality was noted with fathead minnows in the highest concentration tested in as little as 3 hours in CERC well water. Mortality was delayed to 6 h in fathead minnows exposed to the Colorado River water. Significant mortality in Colorado pikeminnow was not noted until 24 h in CERC well water and 48 h in Colorado River water. Again, the apparent difference in ammonia sensitivity across water sources or water qualities was due to differences in pH conditions. When unionized ammonia concentrations were calculated based on differential pH levels it was determined that responses were similar across water qualities. For example, the 24-h EC50 for Colorado pikeminnow ranged from 2.50-2.85 mg/L unionized ammonia whereas the fathead minnow was somewhat more sensitive (24-h LC50 1.71-1.88 mg/L unionized ammonia). However, after 96 h exposure both species exhibited similar LC50 values (i.e., the confidence intervals overlapped) ranging from 1.46-2.19 mg/L unionized ammonia. These sensitivities were similar to those measured at other similar 72-96 h exposure intervals (Table 17; Table 31) in the laboratory regardless of species or source of ammonia (i.e., actual Colorado River water from below Moab Wash or ammonia-spiked CERC well water). These findings indicate that ammonia is the primary toxicant associated with groundwater entering the Colorado River from beneath the tailings pile for two reasons: 1) species sensitivities were similar across the various studies described above, and 2) measured levels of metals and radiochemicals were below levels known to be toxic to fishes.

Influence of pH and temperature on toxicity of ammonia to Colorado pikeminnow

Both temperature and pH are known to influence the toxicity of ammonia to fishes. Ammonia toxicity increases with increasing pH due to the increase in the unionized form. The effect of temperature, however, is species dependant. Therefore, the USEPA recommends that directed, species-specific studies should be conducted in cases where endangered fishes may be exposed. Therefore, we conducted studies to determine the relative sensitivity of Colorado pikemninnow to ammonia under three temperatures (8, 18, and 28 ^{o}C) and three pH regimes (8.0, 8.5, and 9.0). These conditions were chosen to bracket conditions found in the Upper Colorado River.

The toxicity of total ammonia to 6-month old Colorado pikeminnow increased with increasing pH in accordance with classical ammonia equilibria theory (Table 33). However, when adjusted for the actual proportion of unionized ammonia there were no differences across pH within a given temperature (Table 33). For example, the 96h LC50 ranged from 0.38-0.47 mg/L unionized ammonia at 8 ^{o}C; 0.86-1.64 mg/L unionized ammonia at 18 ^{o}C; and 0.96-1.69 mg/L unionized ammonia at 28 ^{o}C. Given the overlap of confidence intervals around these endpoints there were no differences across pH within a given temperature.

Temperature influences, however, were significant. Colorado pikeminnow were 2-3 fold more sensitive to unionized ammonia at 8 ^{o}C than at either 18 ^{o}C or 28 ^{o}C (Table 33). This is significant given that the highest potential exposures of Colorado pikeminnow occur during winter when river discharge is lowest. However, the existing Utah water quality criterion for ammonia at 10 ^{o}C and pH = 8.0 (3.56 mg/L total ammonia; Table 5) would still be protective of the species (over five-fold protection based on the LC50 of 22 mg/L total ammonia; Table 33) even though there is an increased sensitivity of the species at lower temperatures.

Field Toxicity Testing

Sensitivity of Colorado pikeminnow to groundwater in on-site mobile laboratory tests

In August 1998 we conducted an on-site toxicity test in a mobile laboratory in Moab, Utah to determine the toxicity of groundwater from beneath the Atlas Mill tailings pile. Approximately 60 L of pore water was obtained from a pit dug near Moab Wash, which contained approximately 500 mg/L total ammonia. Water was serially diluted using reference Colorado River water from beneath the Hwy 191 Bridge. Seven-day toxicity tests were conducted using 90-d old juvenile Colorado pikeminnow.

Groundwater from near the Moab Wash site was instantly toxic to pikeminnow at the highest concentrations tested; the 0.25 h LC50 was 107 mg/L total ammonia (pH = 8.05; temperature = 25 ^{o}C) or 6.72 mg/L expressed as unionized ammonia (Table 34). Toxicity was total in the 50, 25, and 12.5% dilutions over the course of the study which corresponds to total ammonia concentrations > 40 mg/L total ammonia. Greater than 50% mortality occurred at the lowest dilution tested (6.25% groundwater dilution, or 30.94 mg/L total ammonia; 1.68 mg/L unionized ammonia). Thus, a definitive LC50 could not be obtained in this study. However, this testing demonstrated that undiluted groundwater from beneath the Atlas Mill Tailings pile was a significant threat to juvenile Colorado pikeminnow in potential backwaters of the river. Subsequent studies, described below, clearly demonstrated this toxicity.

Sensitivity of fathead minnow to ammonia in on-site laboratory tests

In February 2000 we conducted an on-site toxicity test in a mobile laboratory in Moab, Utah to verify that Colorado River water adjacent to the Atlas Mill tailings pile was toxic to fishes. Juvenile fathead minnows were tested using 96-h acute toxicity tests in parallel with the *in situ* test described below. Approximately 200 L Colorado River water was collected 250 m downstream of Moab Wash (a backwater near site D5) to obtain a water sample containing approximately 50 mg/L total ammonia. The site was selected based on an *in-situ* measurement of conductivity (approximately 7,000 μmhos), which is associated with groundwater contamination below the pile. A similar amount of water was collected near the Hwy 191 reference site for use in dilution. Two temperature regimes were tested: 6 ^{o}C, to approximate conditions described below in the *in situ* test; and 25 ^{o}C to approximate the standard laboratory test condition.

Fathead minnows were more sensitive to serially diluted Colorado River water at 8 ^{o}C compared to 25^{o}C (Table 35). Similar observations were made with juvenile Colorado pikeminnow in the previously described laboratory study (Table 33). Mortality was first observed following 16 h exposure at both temperatures. Mortality increased over time in both temperature treatments with a final 96-h LC50 of 21.8 and 35.4 mg/L total ammonia at 8 and 25 ^{o}C, respectively. The 96-h LC50 for unionized ammonia (corrected for pH and temperature) was 0.55 and 1.87 mg/L unionized ammonia for 8 and 25^{o}C, respectively. These results indicate, similar to results with Colorado pikeminnow, that winter exposure of juvenile fishes may be the period of greatest concern due to both higher exposure and higher sensitivity. These sensitivities of juvenile fishes, however, bracket those measured in 28-d larval fish studies (Table 17). Although it is plausible that larval fishes could be more sensitive at 8 ^{o}C this is not ecologically relevant since they are spawned in July and would be approximately 180-d of age. Therefore, it appears that even though the fish are more sensitive at colder temperatures, the Utah water quality criteria (adjusted as 0.06 mg/L unionized ammonia) are still protective.

Sensitivity of fathead minnow to ammonia exposed using in situ toxicity tests

In February 2000, we conducted *in situ* toxicity tests to demonstrate the toxicity of site waters to fishes. Juvenile fathead minnows were acclimated to actual site conditions using Hwy 191 water and then placed in various locations near Moab Wash (Figure 43). Daily observations of mortality were made in addition to real-time measures of pH, temperature, and ammonia.

Deployment of juvenile fathead minnows at sites D2, D4, and D6 on February 25, 2000 resulted in instantaneous mortality in some cases due to extremely high concentrations of ammonia (over 1000 mg/L total ammonia; over 2.0 mg/L unionized ammonia (Table 36). Total mortality was observed within 24 h at sites D2, D4, and D6. Note that these ammonia concentrations greatly exceed those from the standard transect samples (Table 15, Figure 36). However, these cage locations were deliberately selected based on measurement of localized conductivities in order to test worst-case conditions. Actual cage deployment itself may have disturbed bottom sediments and released high levels of ammonia from pore waters as indicated by the higher levels of ammonia measured in cages at the bottom compared to the top of the water column. However, this is not a totally unrealistic phenomenon due to the presence of beavers, wading birds, waterfowl, wind activity, and wave action that occurs in these shallow backwaters.

Due to the high toxicity, we re-deployed cages from sites D2 and D4 to additional sites near D6 the following day in order to determine a threshold for impacts and to develop a range of reference, partial mortality, and total mortality conditions (Table 37; Figure 43). Survival at 96 h ranged from 0-4 % (reference sites E4 and UX) to 100% at site D6-bottom located at the sediment-water interface. Total ammonia concentrations ranged from 0-1,180 mg/L across sites. However, low pH conditions associated with incoming groundwater resulted in unionized ammonia concentrations ranging from 0-10.21 mg/L unionized ammonia. Although total mortality occurred at site D6c-bottom (average 10.21 mg/L unionized ammonia) there was only partial mortality (52–72%) at sites D6b, D6c surface, D6d, and D6e where average unionized ammonia exposures ranged from 0.67-1.90 mg/L. Thus, observed *in situ* toxicity occurred at similar unionized ammonia concentrations as those repeatedly demonstrated in laboratory and on-site mobile lab studies. These results, paired with the analytical results for metals and radiochemicals, indicate that ammonia is the primary toxicant of concern adjacent to the tailings pile.

Benthic Invertebrate Distributions in the Upper Colorado River, February 1999

In February 1999 we sampled benthic invertebrate distributions above and below Moab Wash to determine if ammonia or other groundwater constituents were influencing invertebrate food resources. Over 40 macroinvertebrate taxa were found during the sampling effort (Table 38). Total numbers of invertebrate taxa ranged from 1-23 taxa per sample (Figure 44). Total numbers of invertebrates ranged from 2 - 1,672 per sample (Figure 45). Simpson’s Dominance Index ranged from 0.12 (meaning many species that were evenly distributed) to 1.0 (indicating that a single species was present in the sample) (Figure 46).

The benthic invertebrate community was not affected by ammonia concentrations. Two types of analyses were conducted: 1) analysis of variance (used to test for significant differences above and below Moab Wash), and 2) bivariate and multiple regression. A two-way analysis of variance of invertebrate data was conducted to determine the effects of location (e.g., upstream, downstream, or east side of river) and strata (e.g., 1, 5, or 10 m). The upstream treatment consisted of sites UX, U4, and MW; downstream treatment consisted of sites D6, D8, D10; and the eastside treatment consisted of sites E4 and E10. There were no significant differences by location or strata for either total number of taxa or total numbers of invertebrates (Table 39). However, there was a significant effect of location on Simpson's Dominance. A Duncan's Multiple Range Test indicated that Simpson's Dominance was higher for the east side of the river compared to either upstream or downstream locations (Table 40). Thus, there was no apparent effect of the tailings pile on the invertebrate community.

Correlation analysis was used to determine which parameters had the greatest influence on the benthic invertebrate community. The bivariate correlations of taxa richness ($r^2 = 0.055$), total numbers of individuals ($r^2 = -0.226$), and Simpson's Dominance ($r^2 = -0.151$) with unionized ammonia were not significant ($p < 0.249$); thus, unionized ammonia explained little of the variation in benthic invertebrate community distributions. The full correlation matrix, which examined all bivariate correlations for the February 1999 data, is presented in Table 41. Highest correlations occurred within the benthic invertebrate parameters, which is not surprising since they are auto-correlated to some degree. For example, as number of taxa increases the Simpson's Dominance value decreases. Number of taxa was positively correlated with total sum of individuals ($r^2 = 0.665$), %fine gravel ($r^2 = 0.529$), %cobble ($r^2 = 0.492$), %coarse gravel ($r^2 = 0.445$) and current velocity ($r^2 = 0.389$); total number taxa was negatively correlated with percent sand-silt-clay ($r^2 = -0.587$) and %sand ($r^2 = -0.540$). Total number of invertebrates was not significantly correlated with any single parameter. Simpson's Dominance was negatively correlated with depth ($r^2 = -0.362$). Numerous other bivariate correlations were observed but must be viewed with caution. For example, hardness is significantly correlated with calcium and magnesium because it is the titrated sum of divalent cations. Another positive correlation was observed with ammonia and gravels. However, these variables co-exist at sites D6, D8, and D10 due to the combined effects of groundwater leachates and the incoming river current below Moab Wash. Thus, correlations are sometimes artifacts of the data and are not causal relationships.

Multiple, stepwise regression gave the best interpretation of the benthic invertebrate data. A maximum of six variables were used in the multiple regressions. Variables selected, along with partial and total model r-square values, are presented in Table 42. Total number of invertebrates was best explained (67% of the variance) by a four-variable model of Cu, pH, velocity, and depth. Number of taxa was best explained (76% of the variance) by the six-variable combination of %sand, velocity, copper, pH, unionized ammonia, and dissolved oxygen. Simpson's Dominance was best explained (67% of the variance) by the six-variable combination of depth, alkalinity, pH, conductivity, %sand, and %clay. Note that the correlation coefficients do not exactly match the data in Table 41 because the multiple stepwise regression deletes any observation with a missing cell; therefore, the model outputs and number of observations differ between the bivariate and multivariate models. Copper was a significant variable selected for the multivariate model for total number invertebrates and number of taxa. However, the total percentage of variation explained was less than 25%. Copper concentrations were never high enough to be toxic to invertebrates. However, copper was highest at sites MW and D6 where physical habitat was good (i.e., gravels with slow current) and invertebrate numbers were highest. Thus, there may be a significant association of invertebrate community parameters with copper but this does not imply causality. Collectively, both bivariate correlation and multiple regression indicate that variation in benthic invertebrate community distributions were more strongly determined by variables other than ammonia or metals and that physical habitat characteristics were a dominant factor.

DISCUSSION

Field monitoring has indicated that ammonia is the major contaminant of concern for endangered fishes in the vicinity of the Atlas site. Under low-flow conditions (< 5,000 cfs) of late summer, fall, and winter the concentrations of ammonia in backwater areas can exceed the existing water quality criteria of ammonia by several orders of magnitude because groundwater from beneath the Atlas Mill Tailings Pile is a major hydrologic influence. At higher flows this backwater does not exist and ammonia concentrations are diluted by river flow.

Ammonia primarily exists in two forms: unionized (NH_3) and the ionized ammonium ion (NH_4^+). The relative distribution of the two forms is controlled by pH and temperature. It is the unionized form of ammonia that is most toxic (USEPA, 1999). Unionized ammonia can induce numerous effects in fish including loss of equilibrium, gill hyperplasia, and mortality due to the lack of an outward diffusion gradient across the gill surface during high aqueous exposures.

The data presented in this report is the first published information concerning the sensitivity of Colorado pikeminnow and razorback suckers to ammonia. Results of our studies indicated that the 28-d LC50s for larval Colorado pikeminnow, razorback sucker, and fathead minnow were 10.3, 10.1, and 9.8 mg/L total ammonia, respectively (at pH = 8.25, temperature = 25 °C), or 0.72, 0.63, and 0.67 mg/L unionized ammonia (corrected for pH and temperature), respectively. The 28-d LOEC for mortality of larval Colorado pikeminnow, razorback sucker, and fathead minnow were 13.2, 2.33, and 4.35 mg/L total ammonia, respectively (at pH = 8.25, temperature = 25 °C), or 0.92, 0.14, and 0.31 mg/L unionized ammonia (corrected for pH and temperature), respectively. Although the LOEC for razorback sucker was significantly lower than the other two species, the mortality observed at this level was only 10% and was statistically similar at higher concentrations up to 8.60 mg/L total ammonia (0.53 mg/L unionized ammonia). In addition, the LOEC for razorback sucker is still a factor of three greater than the Utah chronic water quality criterion of 0.71 (at pH = 8.25, temperature 25 °C; Table 5). The chronic data also indicated that mortality was of similar or greater sensitivity as the sub-lethal growth endpoints. Similar results were observed in Lost River suckers exposed to ammonia where mortality was a more sensitive response than the sub-lethal endpoints including growth, whole-body ion content, and swimming performance (Meyer and Hansen, 2002).

Our fathead minnow data compares favorably with other published data. Mayes et al. (1986) conducted acute toxicity tests with juvenile fathead minnows and determined a 96-h LC50 value of 1.50 mg/L unionized ammonia; 28-d larval tests gave a 28-d EC50 of approximately 0.4 mg/L unionized ammonia compared to the control value. In addition, Mayes et al. (1986) determined that survival was four-fold more sensitive than growth in 28-d embryo-larval tests. Thurston et al. (1983) examined the results of thirty-five acute toxicity tests with fathead minnows exposed to ammonia and found a range of 0.75 to 3.4 mg/L unionized ammonia (96-h LC50's). Thurston et al. (1986) conducted 30-d embryo larval tests with fathead minnows and found similar LOEC and LC50 values between 0.42 and 0.91 mg/L unionized ammonia; although histopathological effects (brain lesions) were found at concentrations as low as 0.22 mg/L unionized ammonia these had no apparent effects on growth or survival of the species. Other researchers have documented histopathological changes at sublethal concentrations of ammonia in carp (Flis, 1963), rainbow trout (Smith and Piper, 1974), juvenile turbot (Le-Ruyet Person et al., 1997), and channel catfish (Soderberg et al., 1984). However, in no cases were these changes associated with decreases in growth or increases in mortality. Mitchell and Cech (1983) examined supposed gill hyperplasia attributed to ammonia in channel catfish and found that these changes were not due to ammonia but rather residual monochloramine (NH_2Cl) from municipal water treatment. This may cause doubt concerning any histopathological data from tests using ammonium chloride due to the potential to form monochloramine. To our knowledge, however, this has not been further studied.

Interspecies comparison of this data to the USEPA (1999) database for ammonia toxicity further indicates that the endangered fishes are not more sensitive than standard surrogate species when using equivalent endpoints. Similarly, Sappington et al. (2001) compared Colorado pikeminnow, razorback sucker, and seven other endangered species exposed to five separate model chemicals and demonstrated narrow interspecies variation to individual chemicals with an overall sensitivity range of less than two. Beyers et al. (1994) found that both endangered Colorado pikeminnow and bonytail chub were of relatively low sensitivity compared to other fishes when exposed to carbaryl and malathion.

The majority of comparisons of the sensitivity of endangered species and standard surrogate species have been based on traditional response levels such as the EC20, LC50, or a LOEC. These effect levels are traditionally used for development of water quality criteria where the goal is to protect 95% of species. This approach, however, may not be adequate for protection of endangered species where the goal is to protect individual members of a species. In these cases, site-specific risk assessments, based on a safety factor or a low-level calculated effect concentration (i.e., EC1) should be used. The latter approach is preferred since it is statistically based and derived from actual data. We calculated low-level responses ranging from the EC5 and downward to the EC0.01 using the method of Sun et al. (1995). Using these conservative approaches we estimated that effects could be assumed on individuals (LC0.01) at concentrations as low as 0.17 mg/L unionized ammonia. However, even under these conservative estimates the existing Utah water quality criteria adjusted to unionized ammonia (0.065 mg/L unionized ammonia; calculated from Tables 4 and 5) would be protective of individuals of a theoretical Colorado pikeminnow population by a factor of over two.

Several other dissolved inorganic constituents, including copper and zinc, were measured at levels exceeding state water quality standards near the Moab Wash. However, concentrations of these constituents were transient and do not approach levels demonstrated in the laboratory as acutely toxic to razorback suckers or Colorado pikeminnow. For example, acute toxicities of dissolved metals, including uranium, boron, arsenic, and zinc generally exceed 10 mg/L for these endangered species (Hamilton, 1997; Hamilton and Buhl, 1997). However, data on chronic toxicity of these elements to Colorado pikeminnow and razorback suckers are not available. Although others have suggested that synergistic effects among mixtures of contaminants may be possible (Hamilton and Buhl, 1997; Irwin et al., 1997) we found no apparent additive or synergistic activities with ammonia and other constituents observed in on-site studies because toxicities measured fell within the range of standard laboratory test sensitivities to ammonia.

Selenium concentrations in water adjacent to the Atlas Mill Tailings Pile range from 1-4 μg/L as total selenium, which approaches the water quality criterion of 5 μg/L (USEPA, 1987). Selenium is of particular concern in the western United States due to its propensity to undergo microbial transformations that lead to biomagnification in aquatic food webs (Hamilton, 1998). Concentrations of selenium above 5 μg/L have been shown to result in reproductive failure and developmental abnormalities in fish and birds (Hermanutz et al., 1992; Lemly et al., 1993). However, our data provides no indication that selenium from the Atlas Mill Tailings Pile is elevated to levels of localized concern.

Radiochemicals do not appear to be a concern to aquatic biota in the Colorado River adjacent to the Atlas Mill Tailings Pile. During low-water periods (August and February) Ra-226/228, gross alpha, and gross beta radiation were frequently observed in pore waters and nearshore surface waters at levels above the Utah criteria for drinking water (5, 15 and 50 pCi/L, respectively). These criteria were developed for the protection of humans drinking 2-L of water, daily, over 70 years of exposure, and with at least an order of magnitude safety factor incorporated within mammalian toxicity testing. For example, at the concentrations utilized for Ra-226/228 (5 pCi/L) the total dose contributed to a person at this level would be 5 mrem/year additional dose or a incidence risk of 5×10^{-4}. If uranium were detected at 30 pCi/L it would correspond to an additional dose of 27 mrem/year or an incidence risk of 30×10^{-4}.

It is important to recognize that these levels of toxicological concern are not based upon aquatic organism toxicity testing. This is important to note since the concentrations of these radionuclides during low flow periods ranged from below detection levels to “elevated” concentrations above drinking water criteria both above and below the Atlas site. The radioisotopes of concern are naturally occurring and include both alpha emitters (e.g., the uranium isotopes and radium-226) and beta emitters (e.g., radium-228 and potassium-40). They are found in certain rock types and other aquifer-forming materials that contain trace amounts of the radioactive isotopes of uranium, thorium, and radium. These naturally-occurring radionuclides contribute to the overall background radiation dose which organisms living in the Colorado River Basin are continuously exposed.

There are currently no radio-ecological cleanup criteria or protective media concentrations for aquatic organisms. For radiation ecological protection the United States and international opinion has been summarized in National Council on Radiation Protection and Measurements Report 109 (NCRP, 1991) and in the International Atomic Energy Agency Report No. 332 (IAEA, 1992). Both scientific bodies have concluded that a specific radiation dose limit of 1 rad/day (10 mGy/d) poses no risk to aquatic organisms based upon both literature and mesocosm studies. Using this radiation dose level one can back calculate the concentration of Ra-226/228 and total uranium which would pose a concern to aquatic organisms of the Colorado River. Theoretically, an aquatic organism would receive a dose of 1 rad/d if exposed to a dissolved water concentration equaling 14.5 µCi/L of Ra-226, 28.6 µCi/L of Ra-228, and 43.5 µCi/L of total U-238 (assuming for each radionuclide an external dose coefficient in rad/d per pCi/L of 6.9×10^{-8}, 3.5×10^{-8}, and 2.3×10^{-8}, respectively).

However, since radioisotopes also emit beta and gamma particles from both the parent and decay products coming from the U-238 decay chain, both water and contaminated sediments must be assumed to contribute to the total dose. Using the maximum non-negative radionuclide concentrations found in river surface waters and sediments during this study for Ra-226, Ra-228, and total U-238, the dose to a hypothetical aquatic organism would be as follows (refer to Appendix for data):

Ra-226 Highest concentrations of Ra-226 occurred in the following samples from 1999 and 2000: site UX, nearshore, February 1999, ID#00386262L, water, 260 ± 270 pCi/L; site U4, 1m offshore, February 2000, ID#00406464H, water, 170 ± 79 pCi/L; site UX, 5m offshore, February 1999, ID#00386366U, sediment, 6.6 ± 2 pCi/g; and site D6, nearshore, February 2000, ID#00407014N, sediment, 5.6 ± 0.4 pCi/g. Thus, the maximum dose for the year 1999 approximates 25.5 rad/day from Ra-226 in water and only 0.5 µrad/day from sediment and approximately 16.7 rad/day from Ra-226 in water in the year 2000 and only 0.4 µrad/day from sediments. The dose calculation assumes that the aquatic organism is exposed to the maximum Ra-226 water and sediment concentrations for 24 hours per day using a dose conversion factor for external gamma of 2.5×10^{-5} rad/d per pCi/L (water) and 2.5×10^{-5} rad/d per pCi/g (sediments).

Ra-228 No dissolved water values for Ra-228 were detected in either 1999 or 2000. Maximum values for sediment included the following: site CHW, mid-channel, February 1999, ID#00386465W, sediment, 1.1 ± 0.1 pCi/g; and site D6, nearshore, February 2000, ID#00407014N, sediment, 1.2 ± 0.1 pCi/g. Thus, the maximum dose for the year 1999 and 2000 approximates 0.4×10^{-9} rad/day from Ra-228 in sediment. The dose calculation assumes that the aquatic organism is exposed to the maximum Ra-228 sediment concentrations for 24 hours per day using a dose conversion factor for external gamma of 1.3×10^{-5} rad/d per pCi/L (water) and 1.3×10^{-5} rad/d per pCi/g (sediments).

U238 Calculations using Th234 decay product as surrogate for U238 parent (assuming radioactive decay equilibrium since U238 does not emit a gamma photon) resulted in dissolved water concentrations less than minimum detectable concentrations reportable by the laboratory for both 1999 and 2000 collection periods. Highest concentrations for sediments included the following: site U4, 1m offshore, February 1999, ID#00386469A, sediments, 6.0 ± 0.5 pCi/g; site UX, 1m offshore, February 2000, ID#00407192G, sediments, 65 ± 4 pCi/g. Thus, the maximum dose for the year 1999 and 2000 approximates 14 and 2 μrad/day, respectively, from U-238 in sediment. The dose calculation assumes that the aquatic organism is exposed to the maximum U-238 sediment concentrations for 24 hours per day using a dose conversion factor for external gamma of 8.5×10^{-6} rad/d per pCi/L (water) and 8.5×10^{-6} rad/d per pCi/g (sediments).

Dose calculations shown above illustrate that the dominating radiological concern is from the external gamma emissions arising from Ra-226 concentrations found in water and sediments. However, the maximum water concentrations of Ra-226 were found at sites UX and U4, which are up-river of both the tailings pile and Moab Wash. Therefore the maximum dose to aquatic receptors does not appear to be attributable to leaching from the tailings pile. Instead the highest Ra-226 in water results from the area near the original ore-processing site. Sediment concentrations were low in most cases, which is not surprising considering the scouring processes known to occur during spring runoff along the river. The only site that may contain elevated Ra-226 in sediments potentially attributable to the tailings pile is the sampling location D6. However, the dose resulting from this concentration is not considered to be of radiological importance when compared to national and international radioecological guidelines.

Effects of radiation on sensitive aquatic receptors are difficult to separate and tease apart from the natural ecological temporal and spatial variations that occur along the Colorado River. These difficulties include radionuclide dose-rate dependencies, biological repair at the molecular and cellular level of any potential radiological damage, seasonal differences in organism response, environmental water characteristics (temperature, salinity, pH, chemicals), and timing and expression of any radio-biological damage since reproductive responses (embryonic and gametogensis development) are more sensitive indicators of radiation effects than mortality.

Therefore, based upon the radiation doses predicted for the fish population, it appears that there would be no significant biological impacts to fish populations caused by radionuclide concentrations sampled in the Colorado River waters and sediments. This conclusion is based upon two factors: 1) radio-ecological guidelines of 1 rad/day (NCRP, 1991); and 2) maximum radionuclide concentrations sampled in the river and sediments near the Atlas mill tailings site.

Although effects of uranium and other radionuclides have been induced in fishes in controlled studies there have never been any documented population effects on fishes exposed to environmental effects of ionizing environmental radiation in actual field studies (Eisler, 1994). Radiochemical concentrations are elevated in groundwater below the Atlas Mill Tailings Pile; however, these waters do not result in high radiation exposures to fish.

Collectively, these results indicate that ammonia is the primary toxicological concern for endangered fishes in the vicinity of the Atlas Mill Tailings Pile. Exposures are highest during the low-water period of August through February in backwater areas below Moab Wash. These exposures have been shown to be lethal to the Colorado pikeminnow, razorback sucker, and fathead minnow in laboratory, on-site, and *in situ* field studies. These backwater habitats have been designated as critical habitats for endangered fish species due to historical and recent monitoring studies. Thus, continued input of contaminated groundwater presents an unreasonable risk to these species. The existing Utah water quality criteria for ammonia, if met, would protect these endangered fishes from the deleterious effects of ammonia based on both traditional and conservative (e.g., EC0.01) endpoints.

SUMMARY

Ammonia concentrations in the vicinity of the Atlas Mill Tailings Pile exceed Utah water quality criteria by three orders of magnitude in backwater areas under the low-water discharge period of August to March. These ammonia concentrations are far in excess of toxicity thresholds for endangered fish. Levels of metals (e.g., copper, manganese, and zinc) and radiochemicals are elevated in some areas but do not approach levels of concern. The current Utah water quality criteria for ammonia, if met, would be would be protective of Colorado pikeminnow and razorback sucker populations by a factor of at least two.

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Table 1. Sampling locations in the upper Colorado River near Moab, Utah used during the 2-yr study.

Site	Category	GPS location		Field notes
		Latitude	**Longitude**	
Hwy 191	Reference	38° 36' 17.2"	109° 34' 42.9"	1 km above Moab Wash
CHW	Reference	38° 21' 56.3"	96° 06' 54.9"	Courthouse Wash (CHW)
UG	Reference	38° 36' 9.7"	109° 35' 14.9"	900 m above Moab Wash
UX	Reference	38° 36' 8.1"	109° 35' 16.3"	500 m above Moab Wash
U4	Reference	38° 36' 1.8"	109° 35' 20.6"	200 m above Moab Wash
U2	Reference	38° 35' 57.7"	109° 35' 23.5"	100 m above Moab Wash
Island	Reference	38° 35' 39.8"	109° 35' 23.6"	North end of Center Island
E4	Reference	38° 35' 47.4"	109° 35' 18.5"	East side Mathesson Reserve
E10	Reference	38° 35' 36.7"	109° 35' 19.2"	East side Mathesson Reserve
MW	Exposure	38° 35' 54.5"	109° 35' 24.3"	Moab Wash (MW)
D2	Exposure	38° 35' 51.4"	109° 35' 26.2"	100 m downstream of MW
D4	Exposure	38° 35' 48.6"	109° 35' 27.2"	200 m downstream of MW
D6	Exposure	38° 35' 45.3"	109° 35' 28.6"	300 m downstream of MW
D8	Exposure	38° 35' 42.3"	109° 35' 29.6"	400 m downstream of MW
D10	Recovery	38° 35' 39.1"	109° 35' 30.2"	500 m downstream of MW
D15	Recovery	38° 35' 30.9"	109° 35' 31.6"	750 m downstream of MW
D16	Recovery	38° 35' 29.3"	109° 35' 31.5"	800 m downstream of MW
D18	Recovery	38° 35' 26.1"	109° 35' 30.8"	900 m downstream of MW
D20	Recovery	38° 35' 23.1"	109° 35' 29.1"	1,000 m downstream of MW

Table 2. Inter-laboratory comparison of ammonia samples analyzed at CERC and Energy Labs, Inc. (EL). Samples were taken August 10, 2000.

Site	**Location**	**CERC, Ammonia (mg/L as N)**	**EL, Ammonia (mg/L as N)**	**Mean Value**	**Absolute Difference**	**Relative % Difference (RPD)[1]**
D2	Nearshore	3.1	3.5	3.29	0.42	13
D4	Nearshore	39.0	37.0	38.0	2.00	5
D6	Soil pore	549	539	544	10.0	2
D6	Nearshore	43.3	39.6	41.5	3.70	9
D8	Nearshore	9.1	8.8	8.93	0.27	3
D10	Soil pore	162	173	168	11.0	7
D10	Nearshore	2.9	2.7	2.79	0.14	5
E4	Nearshore	0.1	<0.1	<0.05	0	0
Blank		<0.01	<0.1	0.01	0	0
Hach Standard[2]		1.0	1.1	1.03	0.04	4

[1] Relative percent difference calculated as RPD = (x1-x2)/(mean)*100.

[2] Results of n=16 analyses.

Table 3. Quality assurance samples analyzed at CERC on August 10, 2000.

Hach Standard ID	Nominal Concentration Ammonia (mg/L as N)	Measured Concentration Ammonia (mg/L as N)	Difference between values	Accuracy (%)[1]	RPD[2]
1	1.00	1.01	0.01	101	1.00
2	1.00	1.01	0.01	101	1.00
3	1.00	1.01	0.01	101	1.00
4	1.00	1.01	0.01	101	1.00
5	1.00	1.00	0	100	0
6	1.00	1.00	0	100	0
7	1.00	1.02	0.02	102	1.98
8	1.00	1.03	0.03	103	2.96
9	1.00	1.00	0	100	0
10	1.00	1.01	0.01	101	1.00
11	1.00	0.97	0.03	97	3.05
12	1.00	0.98	0.02	98	2.02
13	1.00	1.05	0.05	105	4.88
14	1.00	1.01	0.01	101	1.00
15	1.00	1.06	0.06	106	5.83
16	1.00	1.03	0.03	103	2.96

Average percent accuracy (mean + S.D.) = 1.013 $\pm$ 0.016 (n=16).

[1] Accuracy = (x1/x2)*100.

[2] RPD = (x1-x2)/mean*100.

Table 4. Classification of substrate materials by particle size (modified from Hamilton and Bergersen 1984, citing Platts et al. 1983).

Class Name	**Size Range (mm)**
Very coarse gravel/cobble/boulder	> 38.1
Coarse gravel	19 - 38.1
Fine gravel	9.5 - 19
Very fine gravel	2 - 9.5
Sand/silt/clay	< 2
Sand	0.062 - 2
Silt	0.004 - 0.062
Clay	0.00024 - 0.004

Table 5. Utah 30-d chronic criteria for total ammonia in mg/L as N[1].

pH	Temperature (^{o}C)						
	0	5	10	15	20	25	30
7.00	10.6	10.6	10.6	7.24	5.00	3.49	2.47
7.25	9.17	9.72	9.16	6.24	4.31	3.02	2.14
7.50	7.91	7.91	7.91	5.40	3.73	2.62	1.86
7.75	6.29	6.29	6.28	4.30	2.98	2.10	1.50
8.00	3.56	3.56	3.56	2.44	1.71	1.21	0.87
8.25	2.03	2.03	2.03	1.40	0.99	0.71	0.52
8.50	1.17	1.17	1.17	0.82	0.58	0.43	0.32
8.75	0.56	0.56	0.56	0.40	0.30	0.23	0.18
9.00	0.41	0.41	0.41	0.30	0.23	0.18	0.15

[1] http://www.rules.utah.gov/publicat/code/r317/r317-002.htm#T15.

Table 6. Percent unionized fraction of ammonia at various pH and temperature regimes.[1]

pH	Temperature (^{o}C)						
	0	5	10	15	20	25	30
7.00	0.08	0.13	0.19	0.27	0.40	0.57	0.80
7.10	0.10	0.16	0.23	0.34	0.50	0.71	1.00
7.20	0.13	0.20	0.29	0.43	0.63	0.89	1.26
7.30	0.17	0.25	0.37	0.54	0.79	1.12	1.58
7.40	0.21	0.31	0.47	0.68	0.99	1.41	1.98
7.50	0.26	0.39	0.59	0.86	1.24	1.77	2.48
7.60	0.33	0.50	0.74	1.08	1.56	2.22	3.11
7.70	0.41	0.62	0.93	1.35	1.95	2.77	3.88
7.80	0.52	0.78	1.16	1.70	2.44	3.47	4.84
7.90	0.65	0.98	1.46	2.13	3.06	4.33	6.01
8.00	0.82	1.23	1.83	2.67	3.82	5.38	7.46
8.10	1.03	1.55	2.29	3.33	4.76	6.69	9.21
8.20	1.29	1.94	2.87	4.16	5.92	8.27	11.30
8.30	1.62	2.43	3.58	5.18	7.34	10.20	13.80
8.40	2.03	3.04	4.47	6.44	9.07	12.50	16.80
8.50	2.55	3.80	5.56	7.97	11.20	15.30	20.30
8.60	3.19	4.74	6.91	9.83	13.70	18.50	24.30
8.70	3.98	5.90	8.54	12.10	16.60	22.20	28.80
8.80	4.96	7.31	10.50	14.70	20.00	26.40	33.70
8.90	6.16	9.03	12.90	17.90	24.00	31.10	39.00
9.00	7.64	11.10	15.70	21.50	28.40	36.30	44.60

[1] Table modified from Thurston et al., 1977.

Table 7. Water quality of nearshore samples in the Upper Colorado River, August 1998.[1]

Site	**Location**	**Total Ammonia (mg/L as N)**	**Unionized Ammonia (mg/L as N)[2]**	**Field pH**	**Field Temp (°C)**	**Field Cond (mmhos)**	**Field DO (mg/L)**
Island	Nearshore	0	0	8.54	24.3	1.06	6.7
E4	Nearshore	0	0	8.47	26.6	1.10	8.2
E10	Nearshore	0	0	8.38	23.8	1.07	7.4
U1	Nearshore	0	0	8.58	25.0	1.19	8.7
U2	Nearshore	0	0	8.69	25.5	1.20	8.3
MW	Nearshore	21	2.87	8.38	27.5	1.90	11.2
D2	Nearshore	224	19	8.03	31.0	7.10	4.8
D4	Nearshore	35	4.90	8.36	28.0	2.15	9.8
D6	Nearshore	19	1.76	8.22	26.0	1.70	8.5
D8	Nearshore	5	0.54	8.38	24.5	1.29	8.3
D10	Nearshore	1	0.10	8.51	24.3	1.23	7.0
D14	Nearshore	0	0.06	8.47	23.9	1.10	7.8
D16	Nearshore	0	0	8.48	23.6	1.10	7.2
D18	Nearshore	0	0	8.45	24.4	1.10	7.4
D20	Nearshore	0	0	8.49	24.4	1.01	7.6

[1] Discharge 3,000 cfs.

[2] Calculated based on field pH and field temperature (Thurston et al., 1977).

Table 8. Nearshore surface and porewater chemistry measured in the Upper Colorado River, August 1998.[1]

Site	Total Ammonia (mg/L as N)	Unionized Ammonia (mg/L as N)	Manganese (μg/L)	Copper (μg/L)	Zinc (μg/L)	Gross Alpha (pCi/L)	Gross Beta (pCi/L)	Ra226 (pCi/L)
				Nearshore Surface Waters				
CERC Well	0.30	<0.10	15	2	8	0	0	0
Hwy 191	0.20	<0.10	22	4	8	7	0	0
CHW	0.40	<0.10	28	5	5	7	40	0
Island	<0.10	<0.10	1	3	40	8	0	0
E4	<0.10	<0.10	6	7	4	0	19	0
E10	<0.10	<0.10	7	5	3	6	0	0
MW	21.0	2.90	53	5	8	54	12	NS[4]
D2	224	42.00	24	5	25	54	12	0
				Soil Pore Waters				
CHW	0.50	0.10	145	8	48	0	0	0
Island	<0.01	<0.10	38	4	18	0	29	0
E4	<0.01	<0.10	6	4	18	0	8	0
E10	<0.01	<0.10	8	4	8	0	29	0
MW	477	19.4	28	77	12	905	601	379
D2	685	58.2	42	286	71	1700	1160	0
Criteria	**1.21**[2]	**0.06**	**NE**[3]	**12**[2]	**110**[2]	**15**[2]	**50**[2]	**5**[2]

[1] Discharge 3,000 cfs.

[2] Criteria from Utah Department of Environmental Quality (1999) for Class 3B rivers. 30-d chronic ammonia criteria based on pH = 8.0 and temperature = 25 °C. Copper criteria based on a water hardness of 100 mg/L. Note that radionuclide criteria are for human drinking water; wildlife criteria do not exist.

[3] Criteria not established (NE).

[4] Not sampled (NS).

Table 9. Water quality in the Upper Colorado River, February 1999.[1]

Site	Total Ammonia (mg/L as N)	Unionized Ammonia (mg/L as N)	Field pH	Field Temp (°C)	Field Cond (mmhos)	Alkalinity (mg/L as $CaCO_3$)	Hardness (mg/L as $CaCO_3$)	Field DO (mg/L)
				Nearshore Surface Waters				
CHW	NS[2]	NS	8.54	5.1	1.38	NS	NS	10.21
UX	2.17	0.12	8.50	9.3	1.30	172	282	7.27
MW	6.57	0.30	8.48	7.9	0.94	164	394	NS
D2	4.44	0.24	8.51	9.3	1.42	158	360	10.50
D4	9.26	0.41	8.37	10.8	0.42	156	370	7.40
D6	71.5	2.04	8.08	13.5	2.47	194	572	7.40
D8	35.7	1.17	8.32	8.2	1.95	170	446	8.55
D10	9.29	0.32	8.40	6.5	1.71	182	530	8.42
				Soil Pore Waters				
CHW	4.78	0.04	7.81	4.7	7.30	540	1786	8.44
UX	2.25	0.03	7.77	13.0	3.80	572	1312	6.69
MW	492	2.36	7.46	8.6	8.42	560	2592	4.20
D2	593	2.42	7.34	10.1	1.78	818	4452	6.00
D4	499	1.59	7.10	14.0	1.60	664	4596	5.00
D6	665	7.79	7.79	10.4	13.9	814	4992	4.44
D8	43.7	0.78	8.07	7.6	1.78	164	448	8.11
D10	428	1.62	7.27	11.2	2.25	1054	5704	4.03

[1]Discharge 3,250 cfs.

[2] Not sampled (NS).

Table 10. Nearshore surface and porewater chemistry measured in the Upper Colorado River, February 1999.[1]

Site	Total Ammonia (mg/Las N)	Unionized Ammonia (mg/L as N)	Manganese (μg/L)	Copper (μg/L)	Zinc (μg/L)	Gross alpha (pCi/L)	Gross beta (pCi/L)	Ra226 (pCi/L)
			Nearshore Surface Waters					
CHW	NS[4]	NS	158	21	1	1	10	NS
UX	2.17	0.12	40	5	6	330	125	260
MW	6.57	0.30	91	5	4	32	23	0
D2	4.44	0.24	69	2	3	15	21	0
D4	9.26	0.41	58	6	12	26	15	0
D6	71.5	2.04	281	8	18	72	49	0
D8	35.7	1.17	169	4	16	65	19	150
D10	9.29	0.32	197	6	58	78	38	0
			Soil Pore Waters					
CHW	4.78	0.04	6790	127	13	0	0	0
UX	2.25	0.03	294	78	6	7100	6270	0
MW	492	2.36	4470	263	41	720	845	510
D2	593	2.42	6450	73	73	1060	1220	650
D4	499	1.59	9450	370	137	560	790	490
D6	665	7.79	5850	77	92	1700	1820	920
D8	43.7	0.78	623	31	17	21	42	0
D10	428	1.62	623	31	17	1710	2210	0
Criteria	**1.21**[2]	**0.06**	**NE**[3]	**12**[2]	**110**[2]	**15**[2]	**50**[2]	**5**[2]

[1] Discharge 3,250 cfs.

[2] Criteria from Utah Department of Environmental Quality (1999) for Class 3B rivers. 30-d chronic ammonia criteria based on pH = 8.0 and temperature = 25 °C. Copper criteria based on a water hardness of 100 mg/L. Note that radionuclide criteria are for human drinking water; wildlife criteria do not exist.

[3] Criteria not established (NE).

[4] Not sampled (NS).

Table 11. Water quality in the Upper Colorado River, June 1999.[1]

Site	Location	Total Ammonia (mg/L as N)	Unionzed Ammonia (mg/L as N)	Field pH	Field Temp (°C)	Field Cond (mmhos)	Field DO (mg/L)
			Nearshore Surface Waters				
Hwy 191	Midchannel	0.15	0.01	8.17	15.6	0.49	5.57
CHW	Nearshore	0.16	0.01	8.13	16.7	0.51	5.61
UX	Nearshore	0.16	0.01	8.18	15.7	0.49	5.22
MW	Nearshore	0.16	0.01	8.19	15.8	0.49	6.37
D2	Nearshore	0.17	0.01	8.16	15.3	0.49	7.44
D4	Nearshore	0.13	0.01	8.17	15.3	0.49	6.35
D6	Nearshore	0.13	0.01	8.17	15.2	0.49	6.73
D8	Nearshore	0.12	0.01	8.19	15.3	0.49	6.02
D10	Nearshore	0.15	0.01	8.21	15.1	0.49	6.35
			Soil Pore Waters				
Hwy 191	Pore	NS[2]	NS	NS	NS	NS	NS
CHW	Pore	NS	NS	NS	NS	NS	NS
UX	Pore	NS	NS	NS	NS	NS	NS
MW	Pore	NS	NS	NS	NS	NS	NS
D2	Pore	NS	NS	NS	NS	NS	NS
D4	Pore	NS	NS	NS	NS	NS	NS
D6	Pore	NS	NS	NS	NS	NS	NS
D8	Pore	NS	NS	NS	NS	NS	NS
D10	Pore	NS	NS	NS	NS	NS	NS

[1] Discharge was approximately 15,000 cfs under bankfull conditions.

[2] Not sampled (NS) due to high water.

Table 12. Water quality in the Upper Colorado River, September 1999.[1]

Site	Location	Total Ammonia (mg/L as N)	Unionzed Ammonia (mg/L as N)	Field pH	Field Temp (°C)	Field Cond (mmhos)	Alkalinity (mg/L $CaCO_3$)	Hardness (mg/L $CaCO_3$)	Field DO (mg/L)
				Nearshore Surface Waters					
Hwy 191	Nearshore	0.15	0.02	8.35	19.4	0.88	164	340	6.29
CHW	Midchannel	0.10	0.01	8.3	18.9	0.81	66	140	6.54
UX	Nearshore	0.21	0.01	8.28	18.5	0.87	142	320	9.16
MW	Nearshore	15.0	1.62	8.37	23.7	1.66	194	NS[2]	9.71
D2	Nearshore	3.81	0.35	8.43	19.1	0.90	144	336	7.57
D4	Nearshore	8.12	0.48	8.23	19.0	1.20	172	392	8.62
D6	Nearshore	0.43	0.03	8.39	18.0	0.91	154	338	7.62
D8	Nearshore	0.36	0.02	8.28	17.9	0.86	158	344	7.17
D10	Nearshore	0.11	0.01	8.34	19.6	0.87	160	356	6.55
				Soil Pore Waters					
CHW	Pore	2.60	0.01	7.16	18.6	1.15	500	460	2.50
UX	Pore	0.14	0	7.57	18.6	3.58	472	1270	4.75
MW	Pore	653	3.35	7.20	17.3	1.47	796	3750	4.09
D2	Pore	1082	7.11	7.26	18.8	2.41	1168	5200	4.27
D4	Pore	884	4.98	7.23	17.6	2.12	996	4560	4.11
D6	Pore	587	2.56	7.14	17.0	2.10	1064	5440	2.84
D8	Pore	13.4	0.16	7.57	17.5	3.45	276	1450	3.91
D10	Pore	0.11	0	7.79	17.8	1.04	156	390	5.70

[1] Discharge 15,000 cfs.

Table 13. Water quality in the Upper Colorado River, February 2000.[1]

Site	Location	Total Ammonia (mg/L as N)	Unionized Ammonia (mg/L as N)	Field pH	Field Temp (°C)	Field Cond (mmhos)	Alkalinity (mg/L)	Hardness (mg/L)	Field DO (mg/L)
				Nearshore Surface Waters					
Hwy 191	Nearshore	0.80	0.01	7.98	6.2	1.19	116	500	8.14
CHW	Nearshore	0	0	7.91	8.1	1.15	228	390	8.63
UX	Nearshore	0.90	0.01	7.91	6.7	1.21	162	322	9.03
MW	Nearshore	19.1	0.21	7.90	6.4	1.36	166	359	7.79
D2	Nearshore	2.02	0.02	7.80	6.6	1.28	162	340	7.85
D4	Nearshore	18.5	0.32	7.99	9.5	1.92	168	408	8.77
D6	Nearshore	41.4	0.56	7.83	11.0	2.61	374	512	8.10
D8	Nearshore	27.9	0.48	8.00	9.3	2.10	182	454	8.03
D10	Nearshore	5.20	0.06	7.87	7.4	1.45	178	350	8.82
				Soil Pore Waters					
Hwy 191	Pore	0.90	0	7.45	7.5	4.05	1100	600	2.30
CHW	Pore	3.10	0.01	7.27	6.9	4.01	580	1600	4.24
UX	Pore	63.6	0.36	7.50	9.7	2.62	456	1040	4.07
MW	Pore	332	1.65	7.51	7.6	15.4	840	4360	5.96
D2	Pore	602	0.99	7.00	8.4	17.6	780	3900	2.79
D4	Pore	710	0.99	6.83	11.3	19.0	880	5350	1.03
D6	Pore	705	1.42	7.02	10.5	20.2	1300	4780	2.66
D8	Pore	16.7	0.26	8.00	7.9	1.82	172	432	5.15
D10	Pore	303	0.87	7.19	9.9	20.4	1000	5780	2.93

[1] Discharge 3,300 cfs.

Table 14. Nearshore surface and porewater chemistry measured in the Upper Colorado River, February 2000.[1]

Site	Location	Total Ammonia (mg/L as N)	Unionized Ammonia (mg/L as N)	Manganese (μg/L)	Copper (μg/L)	Zinc (μg/L)	Gross alpha (pCi/L)	Gross beta (pCi/L)	Ra226 (pCi/L)
				Nearshore Surface Waters					
Hwy 191	Nearshore	0.80	0.01	101	11	37	16	14	0
CHW	Midchannel	0	0	114	3	6	3	9	0
UX	Nearshore	0.90	0.01	110	8	34	47	41	0
MW	Nearshore	19.10	0.21	129	11	36	29	28	0
D2	Nearshore	2.02	0.02	242	14	55	23	33	0
D4	Nearshore	18.5	0.32	292	15	43	38	44	0
D6	Nearshore	41.4	0.56	410	9	32	75	143	103
D8	Nearshore	27.9	0.48	224	10	30	60	68	0
D10	Nearshore	5.20	0.06	142	9	36	25	36	0
				Soil Pore Waters					
Hwy 191	Pore	0.90	0	2280	8	28	-8	0	0
CHW	Pore	3.10	0.01	5750	4	12	11	24	0
UX	Pore	63.6	0.36	5920	8	26	1140	1230	662
MW	Pore	332	1.65	12000	29	105	649	1190	697
D2	Pore	602	0.99	7840	16	124	899	998	547
D4	Pore	710	0.99	8000	23	127	652	1340	0
D6	Pore	705	1.42	5970	19	135	1020	1720	184
D8	Pore	16.7	0.26	633	6	13	32	56	0
D10	Pore	303	0.87	13000	21	105	1250	2190	0
Criteria		**1.21** [2]	**0.06**	**NE**[3]	**12**[2]	**110**[2]	**15**[2]	**50**[2]	**5**[2]

[1] Discharge 3,300 cfs.

[2] Criteria from Utah Department of Environmental Quality (1999) for Class 3B rivers. 30-d chronic ammonia criteria based on pH = 8.0 and temperature = 25 °C. Copper criteria based on a water hardness of 100 mg/L. Note that radionuclide criteria are for human drinking water; wildlife criteria do not exist.

[3] Criteria not established (NE).

Table 15. Water quality in the Upper Colorado River, August 2000.

Site	Location	Total Ammonia (mg/L as N)	Unionized Ammonia (mg/Las N)	Field Temp (°C)	Field pH	Conductivity (mmhos)	Alkalinity (mg/L)	Hardness (mg/L)	Field DO (mg/L)
				Nearshore Surface Waters					
Hwy 191	Nearshore	0.10	0.01	25.7	8.36	1.04	140	428	6.39
CHW	Midchannel	7.70	1.31	27.7	8.47	4.49	408	700	8.45
UX	Nearshore	0.10	0.01	24.9	8.38	1.08	140	357	5.32
MW	Nearshore	1.40	0.29	27.4	8.59	1.90	148	347	6.51
D2	Nearshore	3.10	0.29	25.6	8.24	1.21	100	381	7.32
D4	Nearshore	39.1	3.67	25.9	8.23	3.22	104	470	6.52
D6	Nearshore	43.3	2.65	24.7	8.07	1.67	230	455	7.41
D8	Nearshore	9.10	1.22	30.9	8.26	1.41	136	348	7.49
D10	Nearshore	2.90	0.33	28.4	8.26	1.32	80	381	6.81
				Soil Pore Waters					
Hwy 191	Pore	0.30	0.02	23.6	8.08	4.99	220	224	1.57
CHW	Pore	7.50	0.22	23.6	7.77	1.64	510	480	2.66
UX	Pore	0.40	0.01	22.9	7.76	3.17	406	727	1.94
MW	Pore	519	18.56	25.1	7.81	14.90	800	3187	3.01
D2	Pore	577	5.67	25.1	7.24	19.3	720	3830	1.67
D4	Pore	617	5.00	23.3	7.21	19.5	780	403	2.64
D6	Pore	549	6.94	23.1	7.41	20.9	920	3463	3.64
D8	Pore	6.70	0.55	27.1	8.14	1.64	300	546	3.45
D10	Pore	162	2.76	22.2	7.57	18.6	940	7190	3.08

[1] Discharge 3,050 cfs.

[2] Not sampled (NS).

Table 16. Summary of toxicity tests conducted[1].

Date	Location	Test	Temp (°C)	pH	Life stage / species	Water Type	Endpoint	Water Quality[2]
July 1998	CERC lab	acute (3-d)	25	ambient	60-d juvenile CPM 2-d larval FHM	Colorado River CERC well	mortality	Appendix 25
August 1998	mobile lab	acute (7-d)	25	ambient	90-d juvenile CPM	Colorado River pore water	mortality	Appendix 26
April 1999	CERC lab	chronic (28-d)	25	ambient	7-d larval RBS 2-d larval FHM	CERC well	mortality growth	Appendix 27
July 1999	CERC lab	chronic (28-d)	25	ambient	4-d larval CPM 2-d larval FHM	CERC well	mortality growth	Appendix 28
October 1999	CERC lab	acute (4-d)	8, 18, 28	ambient	140-d juvenile CPM	CERC well	mortality	Appendix 29
November 1999	CERC lab	acute (4-d)	8, 18, 28	8.0, 8.5, 9.0	160-d juvenile CPM	CERC well	mortality	Appendix 30
February 2000	mobile lab	acute (4-d)	8, 25	ambient	60-d juvenile FHM	Colorado River	mortality	Appendix 31
February 2000	*in situ*	acute (4-d)	8	ambient	60-d juvenile FHM	Colorado River	mortality	Appendix 32
August 2000	CERC lab	acute (4-d)	25	ambient	60-d juvenile CPM 2-d larval FHM	Colorado River CERC well	mortality	Appendix 33

[1] CPM = Colorado pikeminnow. FHM = fathead minnow. RBS = razorback sucker. CERC = Columbia Environmental Research Center.

[2] Appendix of water quality data associated with each toxicity test.

Table 17. Acute toxicity of ammonia to larval Colorado pikeminnow, razorback sucker, and fathead minnow in 28-d exposures.

			Total Ammonia LC50, mg/L as N (95% C.I.)						
Species	**Test Date**	**Water Type**	**Day 1**	**Day 2**	**Day 4**	**Day 7**	**Day 14**	**Day 21**	**Day 28**
Colorado pikeminnow	July 1999	CERC Well	>13.2	>13.2	>13.2	>13.2	11.4 (8-13)	10.3 (8-13)	10.3 (8-13)
Fathead minnow	July 1999	CERC Well	>13.7	>13.7	12.5 (8-14)	12.1 (8-14)	11 (8-14)	10.5 (8-14)	10.0 (8-14)
Razorback sucker	April 1999	CERC Well	> 13.5	>13.5	>13.5	>13.5	>13.5	10.9 (9-13)*	10.1 (9-14)*
Fathead minnow	April 1999	CERC Well	>13.4	>13.4	12.6 (11-17)	10.3 (9-12)	9.5 (8-13)*	9.5 (8-13)*	9.5 (8-13)*

* Expressed as range (not 95% C.I).

			Unionized Ammonia LC50, mg/L as N (95% C.I.)						
Species	**Test Date**	**Water Type**	**Day 1**	**Day 2**	**Day 4**	**Day 7**	**Day 14**	**Day 21**	**Day 28**
Colorado pikeminnow	July 1999	CERC Well	>0.92	>0.92	>0.92	>0.92	0.79 (0.54-0.92)	0.72 (0.54-0.92)	0.72 (0.54-0.92)
Fathead minnow	July 1999	CERC Well	>0.97	>0.97	0.88 (0.56-0.97)	0.85 (0.56-0.97)	0.77 (0.56-0.97)	0.74 (0.56-0.97)	0.70 (0.56-0.97)
Razorback sucker	April 1999	CERC Well	>0.85	>0.85	>0.85	>0.85	>0.85	0.68 (0.53-0.85)*	0.63 (0.53-0.85)*
Fathead minnow	April 1999	CERC Well	>0.93	>0.93	>0.93	>0.93	0.64 (0.52-0.87)	0.64 (0.52-0.87)	0.64 (0.52-0.87)

* Expressed as range (not 95% C.I.).

Table 18. Percent mortality of fathead minnows exposed 28 d to ammonia, April 1999[1,2].

Nominal Total Ammonia (mg/L)	Total Ammonia (mg/L as N)	Unionized Ammonia (mg/L as N)	Exposure (d)			
			7	14	21	28
0	0.27 (± 0.30)	0.02 (± 0.02)	0.00 (± 0.00)	3.33 (± 5.77)	6.67 (± 5.77)	6.67 (± 5.77)
2.5	2.23 (± 0.20)	0.16 (± 0.04)	0.00 (± 0.00)	3.33 (± 5.77)	3.33 (± 5.77)	3.33 (± 5.77)
5	4.35 (± 0.30)	0.31 (± 0.07)	0.00 (± 0.00)	16.67* (± 11.55)	16.67* (± 11.55)	16.67* (± 11.55)
10	8.53 (± 0.40)	0.60 (± 0.16)	43.33* (± 5.77)	43.33* (± 5.77)	43.33* (± 5.77)	43.33* (± 5.77)
15	13.36 (± 1.00)	0.93 (± 0.20)	66.67* (± 5.77)	70.00* (± 0.00)	70.00* (± 0.00)	70.00* (± 0.00)

[1] Mean (standard deviation) of fish from (n=3) beakers.

[2] Significant difference from control ($p < 0.05$ level) indicated by "*".

Table 19. Percent mortality of fathead minnows exposed 28 d to ammonia, July 1999[1,2].

Nominal Total Ammonia (mg/L)	Total Ammonia (mg/L as N)	Unionized Ammonia (mg/L as N)	Exposure (d)			
			7	14	21	28
0	0.16 (± 0.21)	0.01 (± 0.02)	30.00 (± 17.32)	33.33 (± 23.09)	33.33 (± 23.09)	33.33 (± 23.09)
2.5	2.19 (± 0.57)	0.15 (± 0.06)	13.33* (± 11.55)	16.67* (± 15.28)	16.67* (± 15.28)	16.67* (± 15.28)
5	4.45 (± 0.42)	0.34 (± 0.11)	10.00* (± 10.00)	10.00* (± 10.00)	10.00* (± 10.00)	10.00* (± 10.00)
10	8.10 (± 0.32)	0.56 (± 0.11)	20.00* (± 10.00)	23.33* (± 11.55)	26.67* (± 15.28)	26.67* (± 15.28)
15	13.70 (± 1.14)	0.97 (± 0.23)	60.00* (± 20.00)	70.00* (± 26.46)	73.33* (± 28.87)	76.67* (± 23.09)

[1] Mean (standard deviation) of fish from (n=3) beakers.

[2] Significant difference from control ($p < 0.05$) level indicated by "*".

Table 20. Percent mortality of Colorado pikeminnows exposed 28 d to ammonia, July 1999[1,2].

Nominal Total Ammonia (mg/L)	Total Ammonia (mg/L as N)	Unionized Ammonia (mg/L as N)	Exposure (d)			
			7	14	21	28
0	0.20 (± 0.20)	0.02 (± 0.02)	3.33 (± 5.77)	6.67 (± 11.55)	6.67 (± 11.55)	6.67 (± 11.55)
2.5	1.98 (± 0.50)	0.14 (± 0.04)	3.33 (± 5.77)	6.67 (± 5.77)	6.67 (± 5.77)	6.67 (± 5.77)
5	4.17 (0±.65)	0.29 (± 0.08)	3.33 (± 5.77)	3.33 (± 5.77)	3.33 (± 5.77)	10.00 ± (10.00)
10	7.74 (± 0.63)	0.54 (± 0.12)	0.00 (± 0.00)	6.67 (± 11.55)	16.67 (± 15.28)	16.67 (± 15.28)
15	13.24 (± 1.70)	0.92 (± 0.25)	6.67 (± 11.55)	70.00* (± 17.32)	80.00* (± 10.00)	80.00* (± 10.00)

[1] Mean (standard deviation) of fish from (n=3) beakers.

[2] Significant difference from control ($p < 0.05$) level indicated by "*".

Table 21. Percent mortality of razorback suckers exposed 28 d to ammonia, April 1999[1,2].

Nominal Total Ammonia (mg/L)	Total Ammonia (mg/L as N)	Unionized Ammonia (mg/L as N)	Exposure (d)			
			7	14	21	28
0	0.19 (± 0.20)	0.01 (± 0.01)	0.00 (± 0.00)	0.00 (± 0.00)	3.33 (± 5.77)	3.33 (± 5.77)
2.5	2.33 (± 0.30)	0.14 (± 0.06)	0.00 (± 0.00)	0.00 (± 0.00)	10.00* (± 0.00)	10.00* (± 0.00)
5	4.53 (± 0.30)	0.27 (± 0.09)	0.00 (± 0.00)	3.33 (± 5.77)	10.00* (± 0.00)	13.33* (± 5.77)
10	8.60 (± 0.50)	0.53 (± 0.16)	0.00 (± 0.00)	0.00 (± 0.00)	13.33 (± 15.28)	13.33 (± 15.28)
15	13.48 (± 1.20)	0.85 (± 0.23)	3.33 (± 5.77)	23.33* (± 32.15)	43.33* (± 40.41)	100.00* (± 0.00)

[1] Mean (standard deviation) of fish from (n=3) beakers.

[2] Significant difference from control ($p < 0.05$) level indicated by "*".

Table 22. Average lengths (mm) of fathead minnows exposed 28 d to ammonia, April 1999[1,2].

Nominal Total Ammonia (mg/L)	pH	Total Ammonia (mg/L as N)	Unionized Ammonia (mg/L as N)	Length (mm) by day			
				7	14	21	28
0	8.22	0.27 (± 0.30)	0.02 (± 0.02)	6.49 (± 0.18)	8.83 (± 0.26)	11.24 (± 0.36)	12.63 (± 0.41)
2.5	8.24	2.23 (± 0.20)	0.16 (± 0.04)	6.63 (± 0.23)	9.00 (± 0.50)	11.28 (± 0.70)	12.32 (± 0.45)
5	8.24	4.35 (± 0.30)	0.31 (± 0.07)	6.91 (± 0.10)	8.54 (± 0.49)	10.34 (± 1.20)	12.57 (± 0.94)
10	8.25	8.53 (± 0.40)	0.60 (± 0.16)	6.60 (± 0.24)	8.95 (± 0.48)	11.03 (± 0.34)	13.57 (± 0.92)
15	8.25	13.36 (± 1.00)	0.93 (± 0.20)	6.72 (± 0.13)	7.42* (± 0.43)	10.39 (± 0.83)	10.59* (± 1.00)

[1] Mean (standard deviation) of fish from (n=3) beakers.

Table 23. Average weights (mg) of fathead minnows exposed 28 d to ammonia, April 1999[1,2].

Nominal Total Ammonia (mg/L)	pH	Total Ammonia (mg/L as N)	Unionized Ammonia (mg/L as N)	Mean weight (mg)	Duncan Grouping
0	8.22	0.27 (± 0.30)	0.02 (± 0.02)	4.75 (± 0.28)	A[2]
2.5	8.24	2.23 (± 0.20)	0.16 (± 0.04)	4.68 (± 0.61)	A
5	8.24	4.35 (± 0.30)	0.31 (± 0.07)	4.87 (± 0.94)	AB
10	8.25	8.53 (± 0.40)	0.60 (± 0.16)	5.92 (± 0.49)	B
15	8.25	13.36 (± 1.00)	0.93 (± 0.20)	2.36 (± 0)	C

[1] Mean (standard deviation) of fish from (n=3) beakers.

[2] Means with same letter are not significantly different.

Table 24. Average lengths (mm) of fathead minnows exposed 28 d to ammonia, July 1999[1,2].

Nominal Total Ammonia (mg/L)	pH	Total Ammonia (mg/L as N)	Unionized Ammonia (mg/L as N)	Length (mm) by day			
				7	14	21	28
0	8.26	0.16 (± 0.21)	0.01 (± 0.02)	7.01 (± 0.67)	8.98 (± 0.52)	11.80 (± 0.14)	14.23 (± 0.50)
2.5	8.23	2.19 (± 0.57)	0.15 (± 0.06)	7.80 (± 0.37	9.72 (± 0.45)	11.38 (± 0.55)	13.66 (± 0.12)
5	8.24	4.45 (± 0.42)	0.34 (± 0.11)	7.21 (± 0.23)	9.24 (± 0.51	11.07 (± 0.36)	13.34 (± 0.24)
10	8.23	8.10 (± 0.32)	0.56 (± 0.11)	6.55 (± 0.57)	8.92 (± 0.14)	10.84 (± 0.36)	13.15 (± 0.89)
15	8.23	13.70 (± 1.14)	0.97 (± 0.23)	5.94* (± 0.33)	8.43 (± 0.89)	11.68 (± 0.83)	12.65* (± 1.21)

[1] Mean (standard deviation) of fish from (n=3) beakers.

[2] Significant difference from control ($p < 0.05$) level indicated by "*".

Table 25. Average weights (mg) of fathead minnows exposed 28 d to ammonia, July 1999[1].

Nominal Total Ammonia (mg/L)	pH	Total Ammonia (mg/L as N)	Unionized Ammonia (mg/L as N)	Mean weight (mg)	Duncan Grouping
0	8.26	0.16 (± 0.21)	0.01 (± 0.02)	5.45 (± 0.52)	A[2]
2.5	8.23	2.19 (± 0.57)	0.15 (± 0.06)	5.22 (± 0.40)	A
5	8.24	4.45 (± 0.42)	0.34 (± 0.11)	4.57 (± 0.12)	A
10	8.23	8.10 (± 0.32)	0.56 (± 0.11)	4.75 (± 0.82)	A
15	8.23	13.70 (± 1.14)	0.97 (± 0.23)	4.31 (± 1.82)	A

[1] Mean (standard deviation) of fish from (n=3) beakers.

[2] Means with same letter are not significantly different.

Table 26. Average lengths (mm) of razorback suckers exposed 28 d to ammonia, April 1999[1,2].

Nominal Total Ammonia (mg/L)	pH	Total Ammonia (mg/L as N)	Unionized Ammonia (mg/L as N)	Length (mm) by day			
				7	14	21	28
0	8.16	0.19 (± 0.20)	0.01 (± 0.01)	8.62 (± 0.11)	10.29 (± 0.50)	12.63 (± 0.07)	14.13 (± 0.33)
2.5	8.20	2.33 (± 0.30)	0.14 (± 0.06)	7.98 (± 0.81)	9.80 (± 0.95)	13.26 (± 0.07)	13.28 (± 0.21)
5	8.20	4.53 (± 0.30)	0.27 (± 0.09)	8.05 (± 0.52)	10.13 (± 0.71)	12.85 (± 0.19)	14.78 (± 0.85)
10	8.25	8.60 (± 0.50)	0.53 (± 0.16)	8.54 (± 0.28)	10.14 (± 0.85)	11.99 (± 0.37)	13.00** (± 0.18)
15	8.24	13.48 (± 1.02)	0.85 (± 0.23)	8.76 (± 0.04)	7.61* (± 0.53)	8.22* (± 0.94)	No survival

[1] Mean (standard deviation) of fish from (n=3) beakers.

[2] Significant difference from control ($p < 0.05$) level indicated by "*".

Table 27. Average weights (mg) of razorback suckers exposed 28 d to ammonia, April, 1999[1].

Nominal Total Ammonia (mg/L)	PH	Total Ammonia (mg/L as N)	Unionized Ammonia (mg/L as N)	Mean weight (mg)	Duncan Grouping
0	8.16	0.19 (± 0.20)	0.01 (± 0.01)	3.79 (± 0.77)	A[2]
2.5	8.20	2.33 (± 0.30)	0.14 (± 0.06)	5.00 (± 0.10)	B
5	8.20	4.53 (± 0.30)	0.27 (± 0.09)	5.40 (± 0.25)	B
10	8.25	8.60 (± 0.50)	0.53 (± 0.16)	3.49 (± 0.55)	A
15	8.24	13.48 (± 1.20)	0.85 (± 0.23)	No survival	

[1] Mean (standard deviation) of fish from (n=3) beakers.

[2] Means with same letter are not significantly different.

Table 28. Average lengths (mm) of Colorado pikeminnows exposed 28 d to ammonia, July 1999[1,2].

Nominal Total Ammonia (mg/L)	pH	Total Ammonia (mg/L as N)	Unionized Ammonia (mg/L as N)	Length (mm) by day			
				7	14	21	28
0	8.28	0.20 (± 0.20)	0.02 (± 0.02)	9.35 (± 0.37)	10.51 (± 0.22)	11.60 (± 0.29)	14.48 (± 0.18)
2.5	8.24	1.98 (± 0.50)	0.14 (± 0.04)	8.52 (± 0.11)	10.47 (± 0.56)	12.10 (± 0.35)	14.31 (± 0.25)
5	8.23	4.17 (± 0.65)	0.29 (± 0.08)	8.44 (± 0.12)	10.40 (± 0.31)	12.35 (± 0.70)	14.50 (± 0.16)
10	8.23	7.74 (± 0.63)	0.54 (± 0.12)	7.46* (± 0.10)	9.38* (± 0.14)	10.90* (± 0.47)	13.61 (± 0.04)
15	8.24	13.24 (± 1.70)	0.92 (± 0.25)	7.28* (± 0.19)	8.36* (± 0.95)	9.79** (± 0.17)	11.65* (± 1.32)

[1] Mean (standard deviation) of fish from (n=3) beakers.

[2] Significant difference from control ($p < 0.05$) level indicated by "*".

Table 29. Average weights (mg) of Colorado pikeminnows exposed 28 d to ammonia, July 1999[1].

Nominal Total Ammonia (mg/L)	pH	Total Ammonia (mg/L as N)	Unionized Ammonia (mg/L as N)	Mean weight (mg)	Duncan Grouping
0	8.28	0.20 (± 0.20)	0.02 (± 0.02)	4.70 (± 0.32)	A[2]
2.5	8.24	1.98 (± 0.50)	0.14 (± 0.04)	4.76 (± 0.50)	A
5	8.23	4.17 (0±.65)	0.29 (± 0.08)	4.58 (± 0.25)	A
10	8.23	7.74 (± 0.63)	0.54 (± 0.12)	3.71 (± 0.25)	B
15	8.24	13.24 (± 1.70)	0.92 (± 0.25)	1.97 (± 0.28)	C

[1] Mean (standard deviation) of fish from (n=3) beakers.

[2] Means with same letter are not significantly different.

Table 30. Projected levels of chronic toxicity of ammonia to Colorado pikeminnow. Data was calculated using the method of Sun et al. (1995) based on the results of 96-h static renewal studies using 90-d old fish (pH = 8.1; temperature = 25 ^{o}C).

	Total Ammonia			**Unionized Ammonia**		
Time and % Mortality	**EC_x (mg/L)**	**Confidence Lower Limit (mg/L)**	**Confidence Upper Limit (mg/L)**	**EC_x (mg/L)**	**Confidence Lower Limit (mg/L)**	**Confidence Upper Limit (mg/L)**
7-d						
5.00%	15.40	11.0	19.80	1.02	0.73	1.32
1.00%	10.90	6.90	15.00	0.73	0.46	1.00
0.50%	9.40	5.60	13.20	0.63	0.37	0.88
0.10%	6.70	3.40	10.10	0.45	0.22	0.67
0.05%	5.80	2.80	8.90	0.38	0.13	0.59
0.01%	4.20	1.70	6.80	0.27	0.10	0.44
14-d						
5.00%	13.60	9.30	17.90	0.91	0.62	1.19
1.00%	9.70	5.80	13.50	0.64	0.38	0.90
0.50%	8.40	0.80	12.00	0.55	0.31	0.80
0.10%	6.00	2.90	9.10	0.39	0.19	0.60
0.05%	5.20	2.30	8.00	0.34	0.15	0.53
0.01%	3.70	1.30	6.10	0.24	0.08	0.40
30-d						
5.00%	11.90	7.80	16.10	0.79	0.52	1.07
1.00%	8.50	4.90	12.20	0.56	0.32	0.81
0.50%	7.30	3.90	10.70	0.48	0.26	0.71
0.10%	5.20	2.30	8.20	0.34	0.15	0.54
0.05%	4.50	1.90	7.20	0.30	0.12	0.47
0.01%	3.20	1.00	5.40	0.21	0.07	0.36
60-d						
5.00%	10.60	6.60	14.50	0.70	0.44	0.97
1.00%	7.50	4.10	11.00	0.50	0.27	0.73
0.50%	6.50	3.30	9.70	0.43	0.21	0.64
0.10%	4.60	1.90	7.40	0.30	0.12	0.48
0.05%	4.00	1.50	6.40	0.26	0.10	0.43
0.01%	2.90	0.90	4.90	0.19	0.05	0.32
90-d						
5.00%	9.80	5.90	13.60	0.65	0.39	0.91
1.00%	7.00	3.70	10.40	0.46	0.24	0.68
0.50%	6.00	2.90	9.10	0.40	0.19	0.61
0.10%	4.30	1.70	6.90	0.28	0.11	0.46
0.05%	3.70	1.30	6.00	0.24	0.09	0.40
0.01%	2.70	0.80	4.60	0.17	0.04	0.30

Table 31. Toxicity of ammonia to Colorado pikeminnow exposed in the laboratory in contrasting water qualities, July 1998.

	LC50 (range 95% C.I.) in mg/L					
	Total Ammonia			Unionized Ammonia		
Water Type	**24-h**	**48-h**	**72-h**	**24-h**	**48-h**	**72-h**
CERC Well	13.70 (8.20-30.70)*	12.30 (8.20-30.70)*	12.30 (8.20-30.70)*	1.72 (1.30-2.66)*	1.62 (1.30-2.66)*	1.62 (1.30-2.66)*
Colorado River	33.20 (29.90-70.50)*	31.0 (29.90-70.50)*	31.0 (29.90-70.50)*	1.49 (1.34-3.07)*	1.39 (1.34-3.07)*	1.39 (1.34-3.07)*

* Signifies range not 95% C.I but rather nonlinear interpolation is between those values.

Table 32. Toxicity data for laboratory toxicity test conducted at CERC, August 2000. Test water was collected 280 m downstream of Moab Wash and then serially diluted using water from site at Hwy 191. NL = non-linear regression.

		Total Ammonia LC50 (mg/L), 95% C.I.						
Species	**Water Type**	**1-h**	**3-h**	**6-h**	**24-h**	**48-h**	**72-h**	**96-h**
Colorado pikeminnow	Colorado River	>65.3	>65.3	>65.3	59.2 (29.5-65.3) NL[1]	43.9 (29.5-65.3) NL	43.9 (29.5-65.3) NL	43.9 (29.5-65.3) NL
Colorado pikeminnow	CERC Well	>62.8	>62.8	>62.8	38.3 (29.3-62.8) NL	33.6 (29.3-62.8) NL	28.1 (15.2-29.3) NL	21.1 (15.2-29.3) NL
Fathead minnow	Colorado River	>65.7	>65.7	49.4 (31.7-65.7) NL	30.2 (15.1-31.7) NL	28.9 (15.1-31.7) NL	26.0 (15.1-31.7) NL	26.0 (15.1-31.7) NL
Fathead Minnow	CERC Well	>59.5	>30.2	48.2 (30.2-59.5) NL	23.8 (20.1-27.8) Probit	23.0 (19.5-26.8) Probit	20.3 (17.5-23.8) Probit	20.3 (17.5-23.8) Probit

		Unionized Ammonia LC50 (mg/L), 95% C.I.						
Species	**Water Type**	**1-h**	**3-h**	**6-h**	**24-h**	**48-h**	**72-h**	**96-h**
Colorado pikeminnow	Colorado River	>3.11	>3.11	>3.11	2.85 (1.54-3.11) NL	2.19 (1.54-3.11) NL	2.19 (1.54-3.11) NL	2.19 (1.54-3.11) NL
Colorado pikeminnow	CERC Well	>3.85	>3.85	>3.85	2.50 (1.98-3.85) NL	2.23 (1.98-3.85) NL	1.90 (1.04-1.98) NL	1.44 (1.04-1.98) NL
Fathead Minnow	Colorado River	>3.99	>3.99	3.02 (1.95-3.99) NL	1.88 (1.09-1.95) NL	1.82 (1.09-1.95) NL	1.67 (1.09-1.95) NL	1.67 (1.09-1.95) NL
Fathead minnow	CERC Well	>3.92	>2.18	3.27 (2.18-3.92) NL	1.71 (1.44-1.99) Probit	1.65 (1.40-1.92) Probit	1.46 (1.25-1.72) Probit	1.46 (1.25-1.72) Probit

[1] Calculated using non-linear regression (NL)

Table 33. Influence of temperature and pH on toxicity of ammonia to Colorado pikeminnow (CPM). Colorado pikeminnows were approximately 6 months old. Testing conducted in CERC well water.

		Total Ammonia LC50 (mg/L), 95%C.I.			
Temp (°C)	**pH**	**24-h**	**48-h**	**72-h**	**96-h**
8.0	8.00	26.3 (16.2-30.9)	24.1 (16.2-30.9)	22.4 (16.2-30.9)	22.4 (16.2-30.9)*
8.0	8.50	7.6 (5.7-8.0) *	7.6 (5.7-8.0)*	7.6 (5.7-8.0)*	7.6 (5.7-8.0)*
8.0	9.00	3.3 (2.3-4.3)	3.3 (2.3-4.3)	3.3 (2.3-4.3)	3.3 (2.3-4.3)
18.0	8.00	>29.9	>29.9	>29.9	28.3 (15.9-29.9)*
18.0	8.50	11.0 (8.0-15.1)*	11.0 (8.0-15.1)*	11.0 (8.0-15.1)*	11.0 (8.0-15.1)*
18.0	9.00	6.3 (5.3-7.6)*	6.3 (5.3-7.6)*	6.3 (5.3-7.6)*	6.3 (5.3-7.6)*
28.0	8.00	36.3 (28.9-96.8)	22.1 (19.7-24.6)	20.6 (15.6-28.9) *	20.6 (15.6-28.9)*
28.0	8.50	9.6 (8.6-10.9)	9.5 (7.4-14.6)*	9.5 (7.4-14.6)*	9.5 (7.4-14.6)*
28.0	9.00	5.3 (4.4-6.3)*	5.3 (4.4-6.3)*	5.3 (4.4-6.3)*	5.3 (4.4-6.3)*

		Unionized Ammonia LC50 (mg/L), 95%C.I.			
Temp (°C)	**pH**	**24-h**	**48-h**	**72-h**	**96-h**
8.0	8.00	0.44 (0.28-.051)*	0.41 (0.28-0.51) *	0.38 (0.28-0.51)*	0.38 (0.28-0.51)*
8.0	8.50	0.44 (0.33-0.47)*	0.44 (0.33-0.47)*	0.44 (0.33-0.47)*	0.44 (0.33-0.47)*
8.0	9.00	0.47 (0.34-0.66)	0.47 (0.34-0.66)	0.47 (0.34-0.66)	0.47 (0.34-0.66)
18.0	8.00	>0.90	>0.90	>0.90	0.86 (0.53-0.90)*
18.0	8.50	0.90 (0.64-1.25)*	0.90 (0.64-1.25)*	0.90 (0.64-1.25)*	0.90 (0.64-1.25)*
18.0	9.00	1.64 (1.15-2.43)*	1.64 (1.15-2.43)*	1.64 (1.15-2.43)*	1.64 (1.15-2.43) *
28.0	8.00	2.67 (2.19-6.35)	1.74 (1.58-1.91)	1.64 (1.29-2.19)*	1.64 (1.29-2.19)*
28.0	8.50	0.99 (0 .90-1.11)	0.96 (.77-1.44)*	0.96 (.77-1.44)*	0.96 (.77-1.44)*
28.0	9.00	1.69 (1.26-2.33)*	1.69 (1.26-2.33)*	1.69 (1.26-2.33)*	1.69 (1.26-2.33)*

* Indicates could not calculate 95% C.I. using Probit Method due to no partial mortality. Data represents range of upper and lower concentrations.

Table 34. Toxicity of ammonia in groundwater to juvenile Colorado pikeminnow. Test was conducted in August 1998 using on-site testing in the mobile laboratory at Moab, Utah. Groundwater was obtained near Moab Wash and serially diluted using Hwy 191 reference water.

	Total Ammonia LC50 (mg/L), 95% C.I.										
Temp (^{o} C)	.25 h	1 h	2 h	15 h	37 h	43 h	58 h	68 h	120 h	140 h	160 h
25	107.09 (91.82–123.50)	44.13 (39.29-81.88)	35.13 (30.94-39.29)	34.50 (30.29-39.29)	32.34 (30.30-34.01)	32.34 (30.30-34.01)	32.03 (29.76-33.74)	<30.94	<30.94	<30.94	<30.94

	Unionized Ammonia LC50 (mg/L), 95% C.I.										
Temp (^{o} C)	.25 h	1 h	2 h	15 h	37 h	43 h	58 h	68 h	120 h	140 h	160 h
25	6.72 (5.82-7.75)	3.11 (2.85-5.00)	2.25 (2.04-2.45)	2.19 (2.00-2.40)	1.85 (1.61-2.07)	1.85 (1.61-2.07)	1.81 (1.55-2.03)	<1.68	<1.68	<1.68	<1.68

Table 35. Toxicity of ammonia to juvenile fathead minnows at two temperatures. Tests were conducted in February 2000 using on-site testing in the mobile laboratory at Moab, Utah. Test water was obtained from site D2 and serially diluted using Hwy 191 reference water.

	Total Ammonia LC50 (mg/L), 95% C.I.								
Temp. (° C)	2 h	16 h	24 h	41 h	48 h	65 h	72 h	87 h	96 h
8	>50.0	47.6 (39.3-74.7)	44.3 (37.5-61.6)	32.7 (20.1-99.0)	32.3 (21.0-70.7)	27.3 (18.5-41.1)	24.6 (12.9-34.9)	22.4 (20.8-24.0)	21.8 (20.2-23.2)
25	>49.4	70.8 (52.1-86.7)	54.2 (45.7-87.1)	48.3 (42.4-63.2)	50.8 (42.7-75.9)	47.7 (40.9-64.6)	42.2 (37.7-49.9)	37.4 (28.9-69.7)	35.4 (28.9-48.7)

	Unionized Ammonia LC50 (mg/L), 95% C.I.								
Temp. (° C)	2 h	16 h	24 h	41 h	48 h	65 h	72 h	87 h	96 h
8	>1.30	1.23 (1.01-1.91)	1.15 (0.97-1.59)	0.84 (0.56-1.66)	0.83 (0.57-1.44)	0.70 (0.49-0.98)	0.62 (0.38-0.85)	0.56 (0.52-0.61)	0.55 (0.50-0.59)
25	>2.84	4.31 (2.98-7.32)	3.14 (2.56-5.51)	2.71 (2.32-3.67)	2.81 (2.33-4.25)	2.62 (2.23-3.62)	2.29 (2.03-2.73)	1.98 (1.65-2.76)	1.87 (1.71-2.09)

Table 36. Results of *in situ* cage study using juvenile fathead minnow conducted February 2000. Cage was located at surface unless noted. Ranges are in parenthesis.

	Percent Mortality				Average Total Ammonia	Average Unionized Ammonia
Site	**24-h**	**48-h**	**72-h**	**96-h**	**(mg/L as N)**	**(mg/L as N)**
D2	100	100	100	100	704 n=1	5.92 n=1
D4a	100	100	100	100	1052 n=1	12.2 n=1
D4b	100	100	100	100	1186 n=1	4.74 n=1
D4c (surface)	100	100	100	100	594 n=2 (37.4-1150)	2.15 n=2 (0.72-3.57)
D4c (bottom)	100	100	100	100	1346 n=2 (1150-1541)	3.57 n=2 (3.56-3.57)
D6 (surface)	100	100	100	100	630 n=2 (29.3-1231)	1.29 n=2 (0.44-2.14)
D6 (bottom)	100	100	100	100	1231 n=1	2.5 n=1
D6b	10	13	50	70	132[1] n=6 (17.4-688)	1.12[1] n=6 (0.20-4.88)
E4	4	4	4	4	0.17 n=5 (0.12-0.2)	0.004 n=5 (0.002-0.007)
UX	0	0	0	0	0.20 n=5 (0.08-0.23)	0.004 n=5 (0.002-0.006)

[1] If highest value deleted, total ammonia = 21.2 mg/L; unionized ammonia = 0.37 mg/L.

Table 37. Results of *in situ* cage study using juvenile fathead minnow conducted February 2000. Results are from re-deployments of cages from locations listed in Table 34 to sites with lower ammonia concentrations. Cage was located at surface unless noted. Ranges are in parentheses.

Site	Percent Mortality 24-h	48-h	72-h	96-h	Average Total Ammonia (mg/L as N)	Average Unionized Ammonia (mg/L as N)
D6c (surface)	57	67	67	70	27.2 n=6 (19.4-34.4)	0.67 n=6 (0.29-1.18)
D6c (bottom)	87	93	100	100	1180 n=5 (809-1349)	10.2 n=5 (2.69-27.6)
D6d	48	52	52	52	78.4 n=6 (10.4-211)	1.57 n=6 (0.26-6.33)
D6e	6	26	39	68	34.9 n=6 (7.5-98.2)	0.72 n=6 (0.12-2.15)
D6b	10	13	50	70	244 n=6 (18.5-688)	1.90 n=6 (0.29-4.88)
E4	4	4	4	4	0.17 n=5 (0.1-0.2)	0.004 n=5 (0.002-0.007)
UX	0	0	0	0	0.2 n=5 (0.08-0.23)	0.004 n=5 (0.002-0.006)

Table 38. Species list from macroinvertebrate sampling, February 1999.

Phylum	Order	Family	Sub-family	Genus
Annelida	Oligochaeta	Naididae		
		Tubificidae		
Mollusca	Gastropoda	Ancylidae		
	Pelecypoda	Corbiculidae		*Corbicula*
Crustacea	Decapoda	Cumbaridae		C*ambarus*
Insecta	Hemiptera	Corixidae		
	Plecoptera	Perlidae		*Acroneuria*
	Odonata	Coenagrionidae		*Argia*
	Lepidoptera	Pyralidae		*Petrophila*
	Trichoptera	Brachycentridae		*Brachecentrus*
		Hydropsychidae		*Ceratopsyche*
	Coleoptera	Elmidae		*Dubiraphia*
				Microcylloepus
				Stenelmis
	Diptera	Ceratopogonidae		*Probezzia*
		Chironomidae	Chironiminae	*midge pupae*
				Chironomus
				Stenochironomus
				Paracladopelma
				Tribelos/ Phaenopsectra sp.
				Dicrotendipes
				Polypedium
				Tanytarsas
				Rheotanytarsus
				Paratanytarsus
			Orthocladiinae	*Orthocladius*
				Parakiefferiella
				Eukiefferiella
			Tanypodinae	
				Procladius
				Ablabesmyla
			Diamesinae	*genus A.*
		Empididae		*Chelifera*
				Hemerodromia
		Simuldae		*Cnephia*
				Simulium
				Ttwinnia
Bryozoa				*Urnatella*
Arachnida	Acarina			*unknown sp.*

Table 39. Results of nonparametric two-way ANOVA of macroinvertebrate indices. Main effects were location and strata.

Variable	Source	Degrees of Freedom	p> F[3]
Number Taxa	Model	8	0.5762
	Location[1]	2	0.5710
	Strata[2]	2	0.2582
	Location*Strata	4	0.7327
Total # Numbers	Model	8	0.8055
	Location	2	0.5183
	Strata	2	0.5871
	Location*Strata	4	0.6113
Simpson's Dominance	Model	8	0.0783
	Location	2	0.0038[3]
	Strata	2	0.7871
	Location*Strata	4	0.6641

[1] Location as main effect includes upstream (pooled sites UX, U4, and MW), downstream (pooled sites D6, D8, and D10), and east side (pooled sites E4 and E10).

[2] Strata as main effect refers to 1 m, 5 m, or 10 m from bank.

[3] Significant values represented by $p \leq 0.05$.

Table 40. Results of Duncan's Multiple Range tests of main effects of location on macroinvertebrate data from Table 38. Note: Data presented are actual data and not ranked values used in ANOVA.

Variable	Location[1]	N	Mean	Duncan Grouping[2]
Number Taxa	East side	5	11.2	A
	Upstream	9	14.3	A
	Downstream	9	14.4	A
Total Numbers	East Side	5	827	A
	Upstream	9	604	A
	Downstream	9	362	A
Simpson's Dominance	East Side	5	0.45	A
	Upstream	5	0.23	B
	Downstream	9	0.28	B

[1] Location as main effect includes upstream (pooled sites UX, U4, and MW), downstream (pooled sites D6, D8, and D10), and east side (pooled sites E4 and E10).

[2] Means with the same letter are not significantly different.

Table 41. Bivariate correlation matrix of macroinvertebrate endpoints and other physical and chemical variables, February 1999. Right quadrant is bivariate correlation coefficients (n=31 samples). Left quadrant is associated probability value.

P value \ Correlation Coefficient (r)	ntaxa	totsum	simpdom	temp	pH	turb	depth	velocity	cond	do	alk	hard	t_amm	un_amm	Ca	Mg	Cu	% cobble	% coarse gravel	% fine gravel	% very fine gravel	% sand, silt, clay	% sand	% silt	% clay
ntaxa		0.665	-0.512	-0.113	0.164	0.024	0.074	0.389	-0.095	-0.187	-0.106	-0.084	0.025	0.055	-0.130	-0.072	-0.118	0.492	0.445	0.529	0.420	-0.587	-0.540	-0.086	-0.051
totsum	<.0001		0.014	-0.190	0.295	-0.151	-0.062	0.284	-0.187	-0.138	-0.202	-0.200	-0.220	-0.226	-0.205	-0.175	-0.198	0.283	-0.033	0.217	0.050	-0.147	-0.311	0.103	0.266
simpdom	0.004	0.941		-0.015	0.009	-0.067	-0.362	-0.218	-0.034	0.187	-0.047	-0.051	-0.155	-0.151	-0.027	-0.052	-0.033	-0.233	-0.289	-0.257	-0.187	0.311	0.263	-0.110	0.148
temp	0.552	0.314	0.937		-0.543	0.512	-0.271	-0.434	0.579	-0.306	0.496	0.583	0.330	0.523	0.360	0.614	0.406	0.137	0.248	-0.034	-0.156	-0.108	-0.226	0.112	0.157
pH	0.388	0.113	0.963	0.002		-0.757	0.244	0.319	-0.914	0.126	-0.941	-0.907	-0.527	-0.392	-0.925	-0.873	-0.935	-0.281	-0.325	-0.069	0.042	0.240	0.163	0.131	0.078
turb	0.900	0.434	0.729	0.004	<.0001		-0.165	-0.275	0.728	-0.327	0.726	0.720	0.246	0.279	0.720	0.721	0.717	0.328	0.356	0.183	0.146	-0.340	-0.437	-0.105	0.234
depth	0.699	0.743	0.049	0.140	0.186	0.384		0.275	-0.067	0.086	-0.101	-0.056	-0.165	-0.177	-0.143	-0.029	-0.139	-0.232	-0.185	-0.026	0.007	0.160	0.237	-0.053	-0.114
velocity	0.034	0.129	0.246	0.015	0.080	0.141	0.134		-0.243	0.265	-0.252	-0.230	-0.157	-0.177	-0.245	-0.222	-0.252	0.004	-0.087	0.184	0.215	-0.061	-0.003	0.202	-0.220
cond	0.619	0.323	0.860	0.001	<.0001	<.0001	0.718	0.188		-0.073	0.979	0.997	0.189	0.046	0.899	0.989	0.930	0.177	0.154	0.008	-0.093	-0.105	-0.126	-0.035	0.056
do	0.324	0.466	0.322	0.094	0.501	0.078	0.645	0.150	0.696		-0.095	-0.024	-0.292	-0.302	-0.111	-0.034	-0.095	-0.131	-0.317	-0.243	-0.117	0.270	0.637	-0.349	-0.494
alk	0.579	0.283	0.805	0.005	<.0001	<.0001	0.590	0.171	<.0001	0.611		0.968	0.143	-0.015	0.967	0.942	0.984	0.178	0.153	0.003	-0.097	-0.103	-0.128	-0.027	0.061
hard	0.659	0.289	0.788	0.001	<.0001	<.0001	0.765	0.213	<.0001	0.896	<.0001		0.255	0.102	0.876	0.995	0.911	0.182	0.163	0.008	-0.086	-0.112	-0.105	-0.062	0.019
t_amm	0.900	0.261	0.432	0.080	0.003	0.207	0.392	0.417	0.326	0.124	0.460	0.183		0.945	0.150	0.296	0.137	0.066	0.661	0.464	0.378	-0.512	-0.279	-0.258	-0.300
un_amm	0.780	0.249	0.442	0.004	0.035	0.150	0.357	0.359	0.815	0.111	0.937	0.598	<.0001		-0.016	0.126	-0.027	0.121	0.683	0.458	0.397	-0.538	-0.295	-0.274	-0.314
Ca	0.492	0.277	0.886	0.047	<.0001	<.0001	0.443	0.185	<.0001	0.553	<.0001	<.0001	0.438	0.933		0.829	0.996	0.171	0.151	0.006	-0.096	-0.101	-0.118	-0.038	0.052
Mg	0.707	0.354	0.785	0.000	<.0001	<.0001	0.878	0.231	<.0001	0.858	<.0001	<.0001	0.119	0.515	<.0001		0.870	0.177	0.152	0.001	-0.099	-0.101	-0.115	-0.038	0.048
Cu	0.536	0.295	0.864	0.024	<.0001	<.0001	0.456	0.172	<.0001	0.611	<.0001	<.0001	0.478	0.890	<.0001	<.0001		0.180	0.148	-0.008	-0.108	-0.096	-0.117	-0.034	0.059
% cobble	0.007	0.138	0.224	0.470	0.132	0.077	0.217	0.981	0.349	0.489	0.346	0.336	0.739	0.541	0.367	0.349	0.340		0.448	0.252	0.098	-0.603	-0.468	-0.145	-0.186
% coarse gravel	0.016	0.867	0.128	0.186	0.079	0.054	0.328	0.648	0.416	0.088	0.419	0.388	0.000	<.0001	0.425	0.423	0.435	0.013		0.770	0.679	-0.931	-0.680	-0.258	-0.348
% fine gravel	0.003	0.258	0.178	0.859	0.717	0.334	0.891	0.331	0.967	0.197	0.987	0.968	0.013	0.014	0.977	0.997	0.966	0.179	<.0001		0.862	-0.872	-0.668	-0.128	-0.329
% very fine gravel	0.023	0.798	0.332	0.410	0.826	0.441	0.969	0.253	0.626	0.539	0.608	0.653	0.048	0.036	0.615	0.602	0.571	0.608	<.0001	<.0001		-0.775	-0.498	-0.262	-0.395
% sand, silt, clay	0.001	0.445	0.100	0.571	0.201	0.066	0.399	0.749	0.580	0.149	0.589	0.555	0.005	0.003	0.597	0.596	0.614	0.000	<.0001	<.0001	<.0001		0.733	0.249	0.385
% sand	0.003	0.100	0.168	0.231	0.389	0.016	0.207	0.986	0.508	0.000	0.502	0.582	0.151	0.128	0.536	0.543	0.537	0.009	<.0001	<.0001	0.005	<.0001		-0.342	-0.307
% silt	0.656	0.594	0.569	0.556	0.491	0.582	0.780	0.285	0.856	0.059	0.886	0.744	0.185	0.159	0.841	0.843	0.859	0.444	0.169	0.502	0.163	0.184	0.065		0.574
% clay	0.793	0.163	0.444	0.406	0.680	0.214	0.550	0.244	0.767	0.006	0.749	0.921	0.121	0.104	0.784	0.803	0.759	0.325	0.060	0.076	0.031	0.036	0.099	0.001	

[1] Ntaxa = total number invertebrate taxa; totsum = total number of individual invertebrates; simpdom = Simpson's Dominance Index.

[2] Substrate categories is described in Table 4

Table 42. Results of multiple stepwise regression analysis used to develop best predictive model of macroinvertbrate community parameters, February 1999 (n=20 samples).

Independent variable	**Dependent variable**	**# model variables**	**Partial model r^2**	**Total model r^2**	**Pr> F**
Total Numbers	Cu	1	0.2361	0.2361	0.0255
	pH	2	0.1890	0.4251	0.0256
	Velocity	3	0.1463	0.5714	0.0276
	Depth	4	0.1003	0.6717	0.0419
Number Taxa	% gravel	1	0.2736	0.2736	0.0150
	Velocity	2	0.1488	0.4224	0.0451
	Cu	3	0.2356	0.5480	0.0441
	pH	4	0.1129	0.6609	0.0347
	Unionized ammonia	5	0.0473	0.7082	0.1400
	Dissolved oxygen	6	0.0555	0.7637	0.0912
Simpson's Dominance	Depth	1	0.2060	0.2060	0.0387
	Alkalinity	2	0.1039	0.3099	0.1171
	pH	3	0.1096	0.4195	0.0911
	Conductivity	4	0.1067	0.5262	0.0758
	%sand	5	0.0764	0.6026	0.1102
	%clay	6	0.0719	0.6745	0.1005

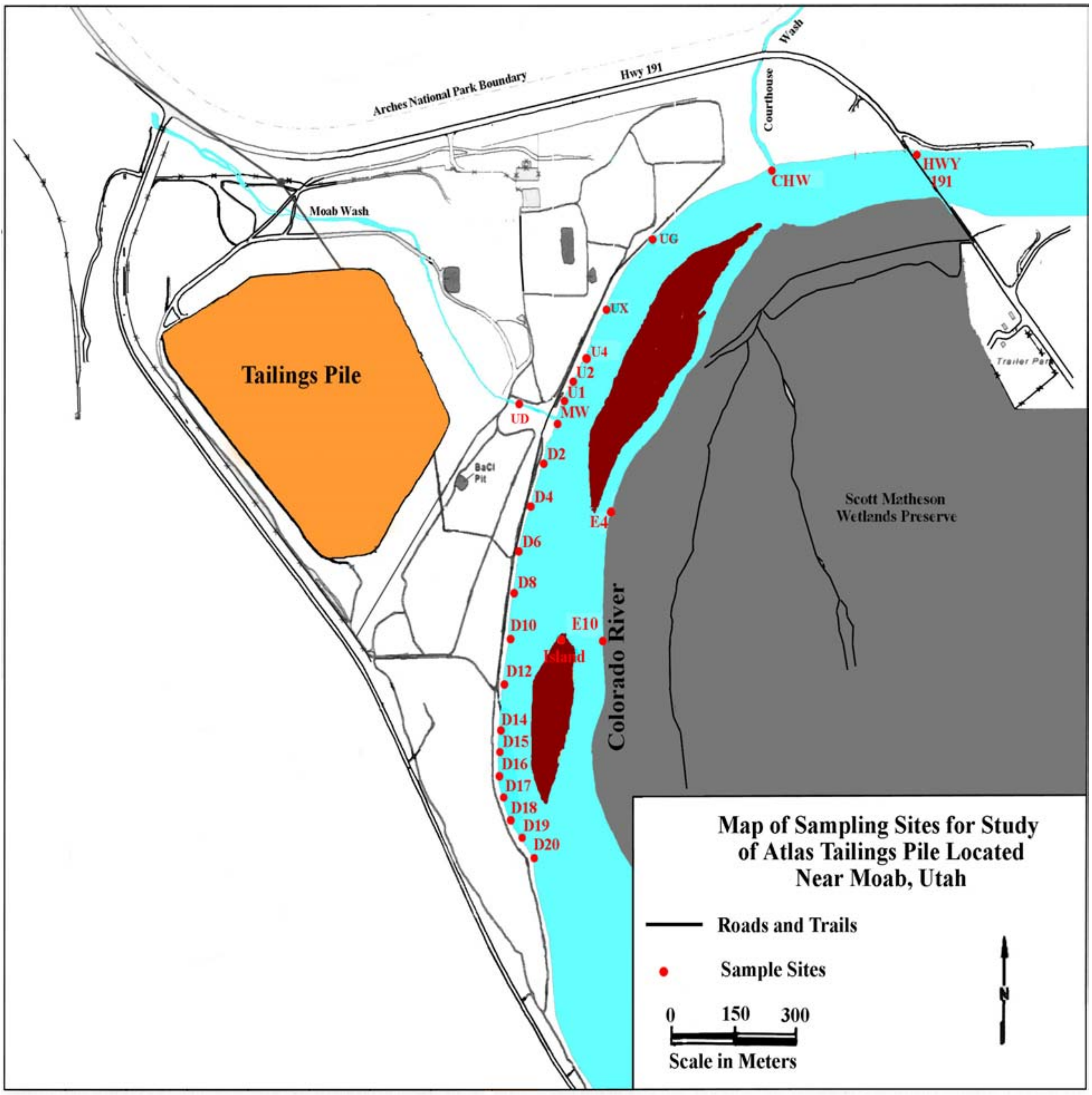


Figure 1. Spatial map of water and sediment sampling locations. Sites HWY 191, UX, CHW, E4, and E10 are considered reference locations. Note that in some cases the locations at MW, D2, and D4 were moved east under low water conditions and may be located as far as 50 m east of GPS location in Table 1. Those instances are discussed in the text.

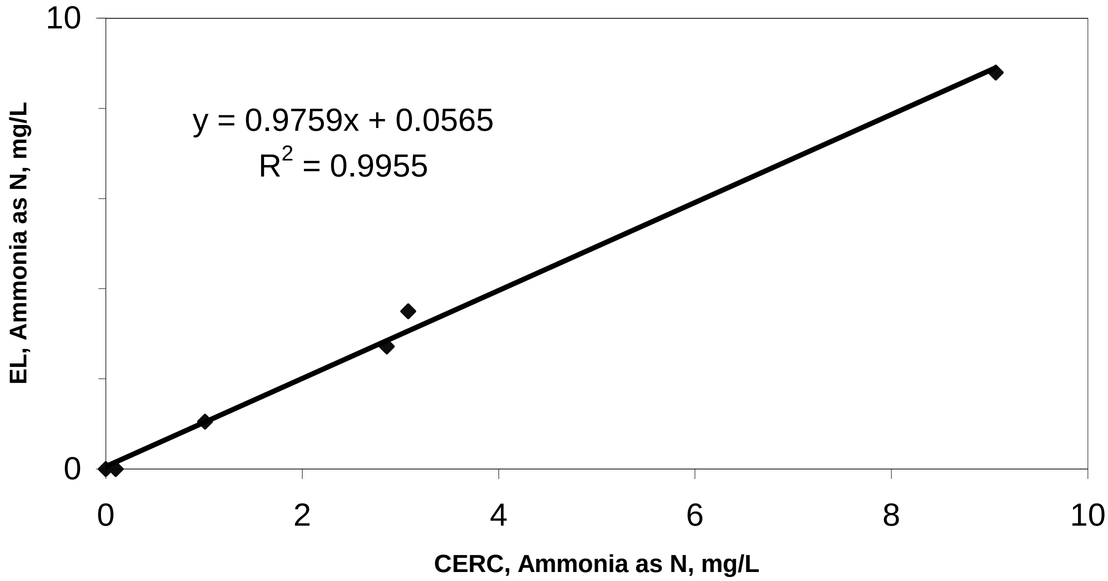


Figure 2. Plot of statistical relationship between total ammonia samples jointly measured by CERC and Energy Labs, Inc. (EL). Samples were sent to Energy Labs, Inc. in August 2000. Data is in Table 2. Samples exceeding 10 mg/L are not plotted.

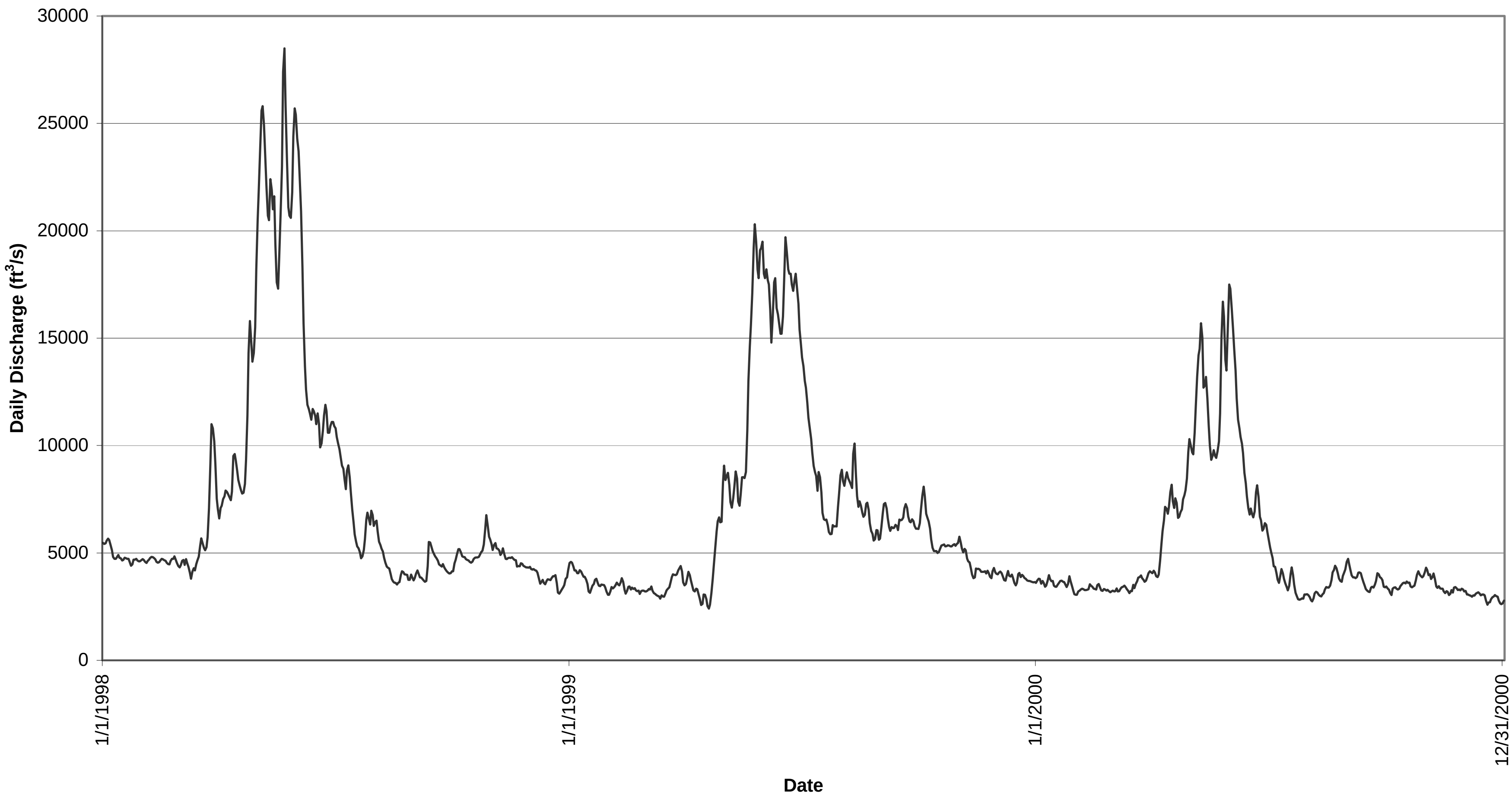
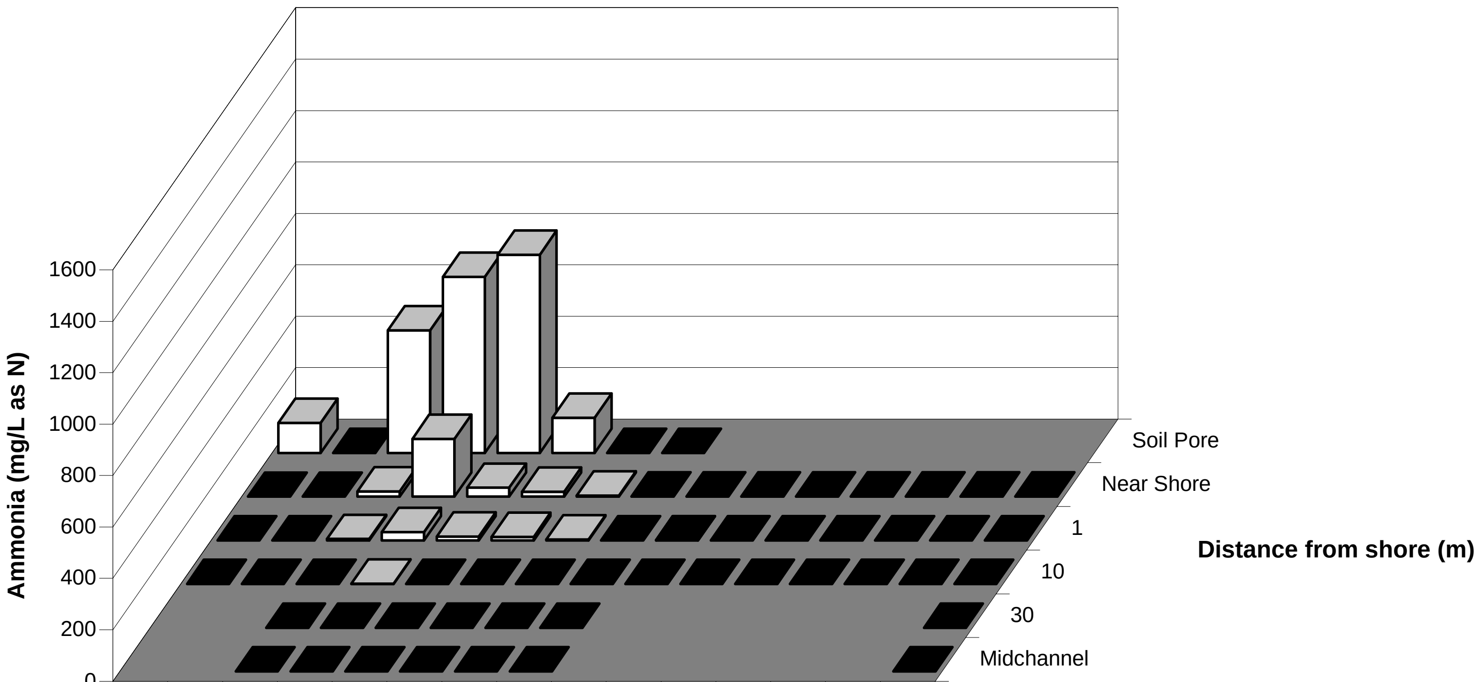


Figure 3. Hydrograph of Colorado River discharge (ft^3/sec) Cisco, Utah from 1998 to 2001.



	U2	U1	MW	D2	D4	D6	D8	D10	D14	D15	D16	D17	D18	D19	D20
Soil Pore	117	0	477	685	771	137	0	0							
Near Shore	0	0	20.5	224	34.9	19.1	4.59	0.65	0.46	0.37	0.21	0.25	0.40	0.32	0.15
1	0	0	5.38	32.8	14.5	13.6	3.83	0	0.48	0.38	0.21	0.22	0.26	0.15	0.13
10	0	0	0.09	1.96	0.25	0.24	0	0	0	0	0	0.02	0.06	0.07	0.07
30			0	0	0.02	0	0	0							0
Midchannel			0	0	0	0	0	0							0

Site

Figure 4. Spatial distribution of total ammonia (mg/L) in water (August 1998). MW= Moab Wash. Sites upstream and downstream of Moab Wash are represented with a "U" and "D", respectively. Numbers represent multiples of 50 m (e.g., D4 = 200 m downstream). Utah 30-d chronic ammonia criteria is 1.21 mg/L as N based on pH = 8.0 and temperature = 25 °C for class 3B rivers.

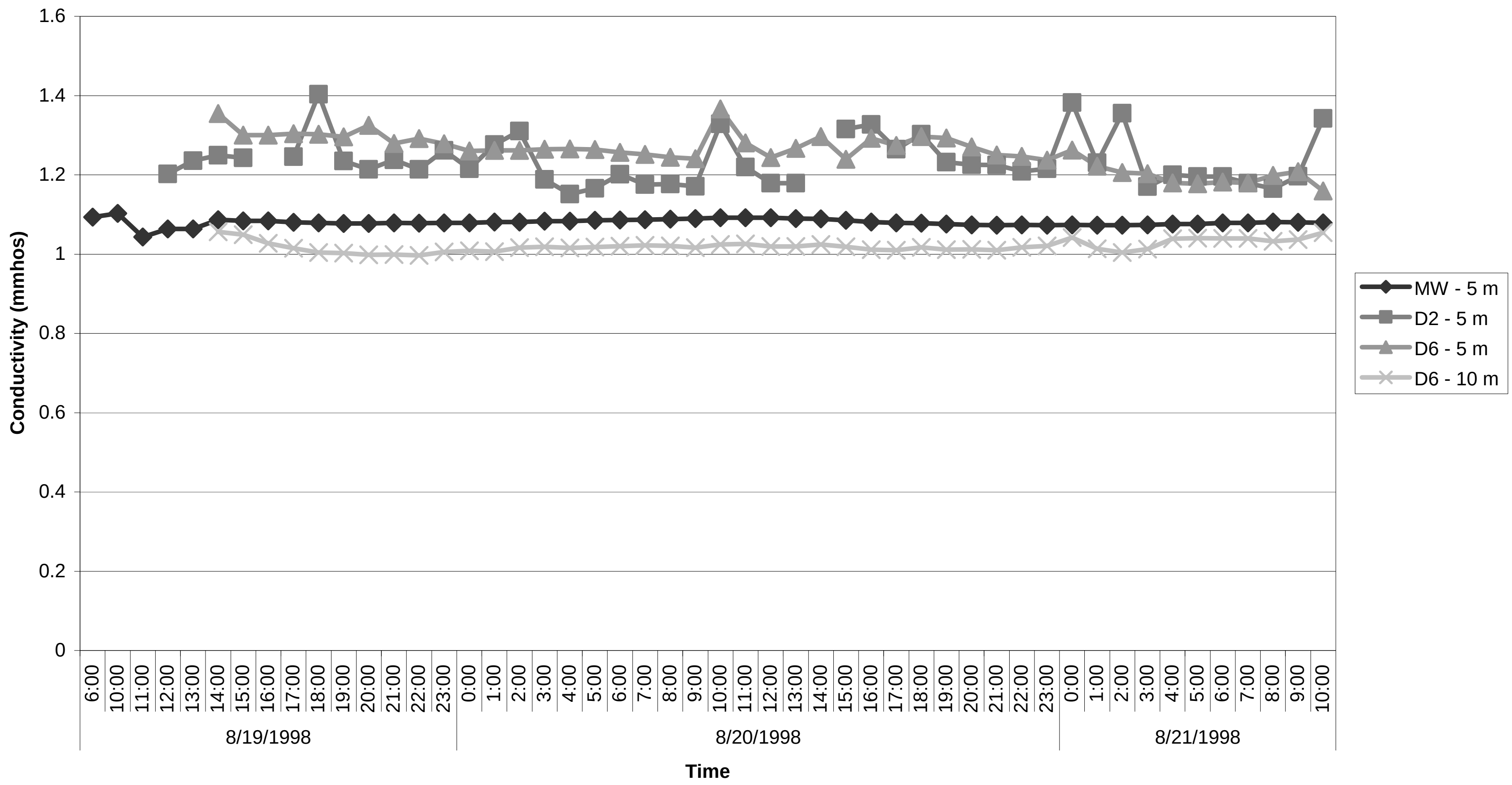


Figure 5. Diurnal changes in conductivity at four locations in the Upper Colorado River, August 1998. The first designation (e.g., MW; Moab Wash) indicates site; the second designation indicates distance from shore in meters (e.g., 5 m). Measurements were taken hourly over a 48-h interval using an in situ Hydrolab Water Quality Monitor. Each measurement represents a mean of n = 2 replicates.

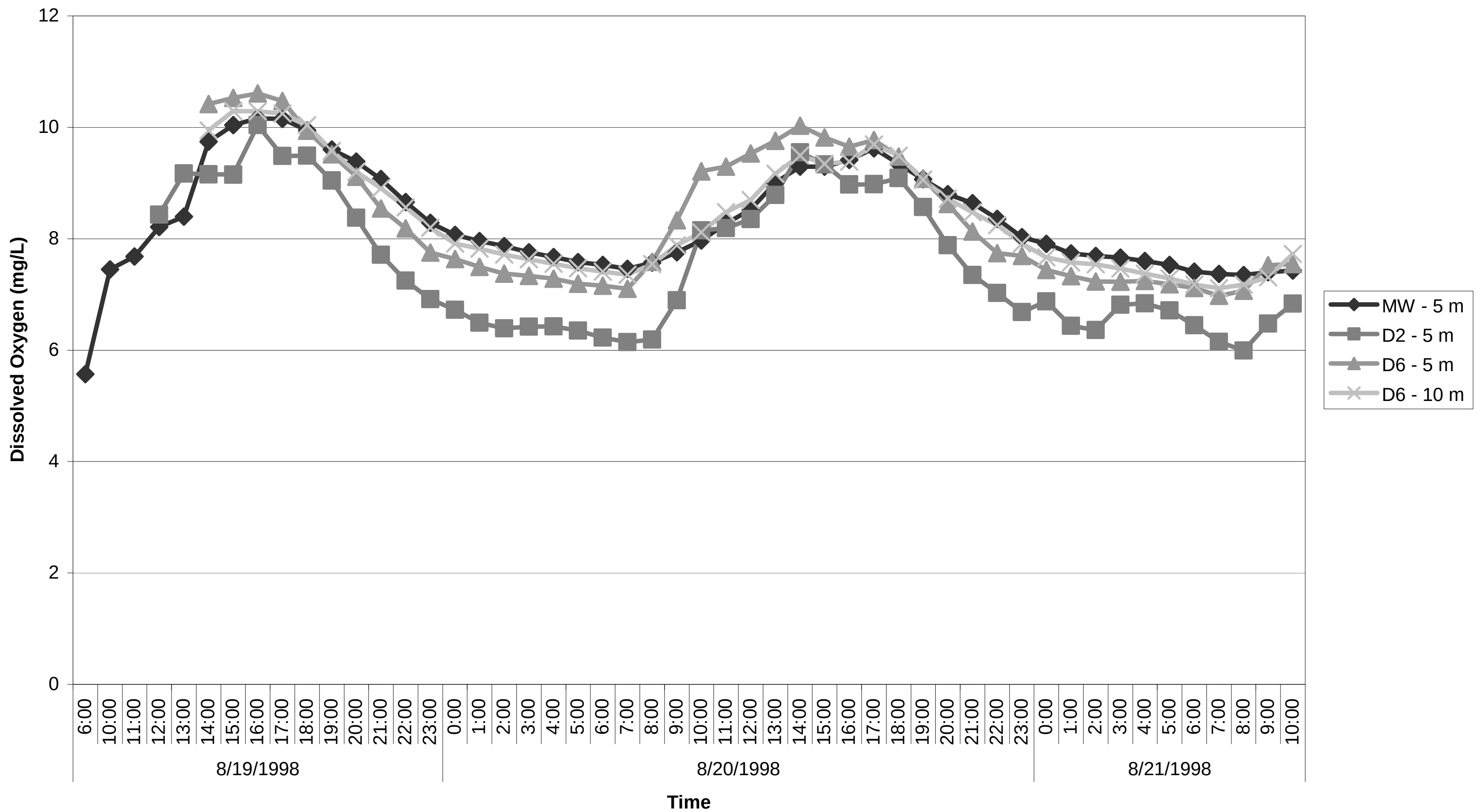


Figure 6. Diurnal changes in dissolved oxygen at four locations in the Upper Colorado River, August 1998. The first designation (e.g., MW; Moab Wash) indicates site; the second designation indicates distance from shore in meters (e.g., 5 m). Measurements were taken hourly over a 48-h interval using an in situ Hydrolab Water Quality Monitor. Each measurement represents a mean of n = 2 replicates.

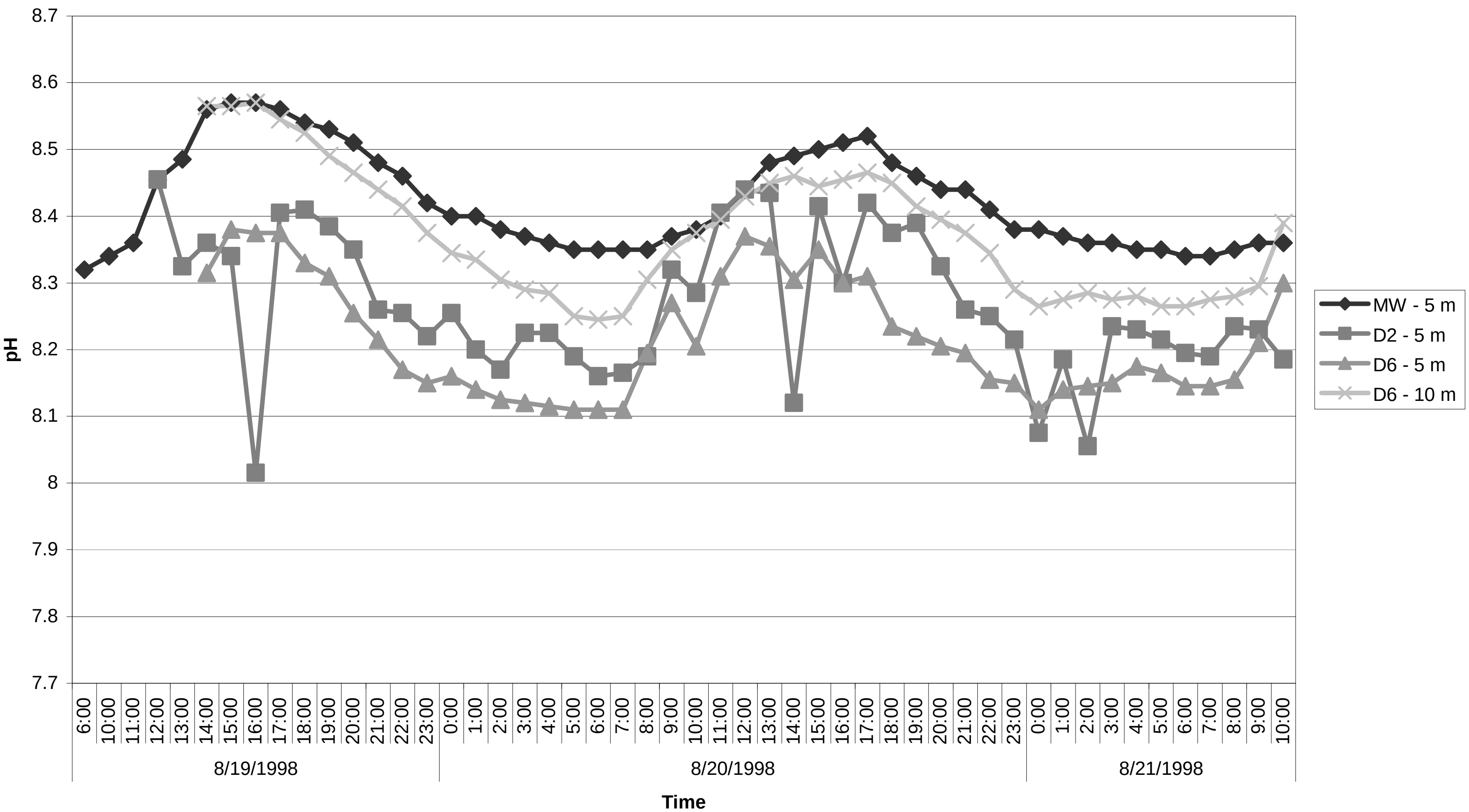


Figure 7. Diurnal changes in pH at four locations in the Upper Colorado River, August 1998. The first half of the legend (e.g., MW; Moab Wash) indicates site; the second designation indicates distance from shore in meters (e.g., 5 m). Measurements were taken hourly over a 48-h interval using an in situ Hydrolab Water Quality Monitor. Each measurement represents a mean of n = 2 replicates.

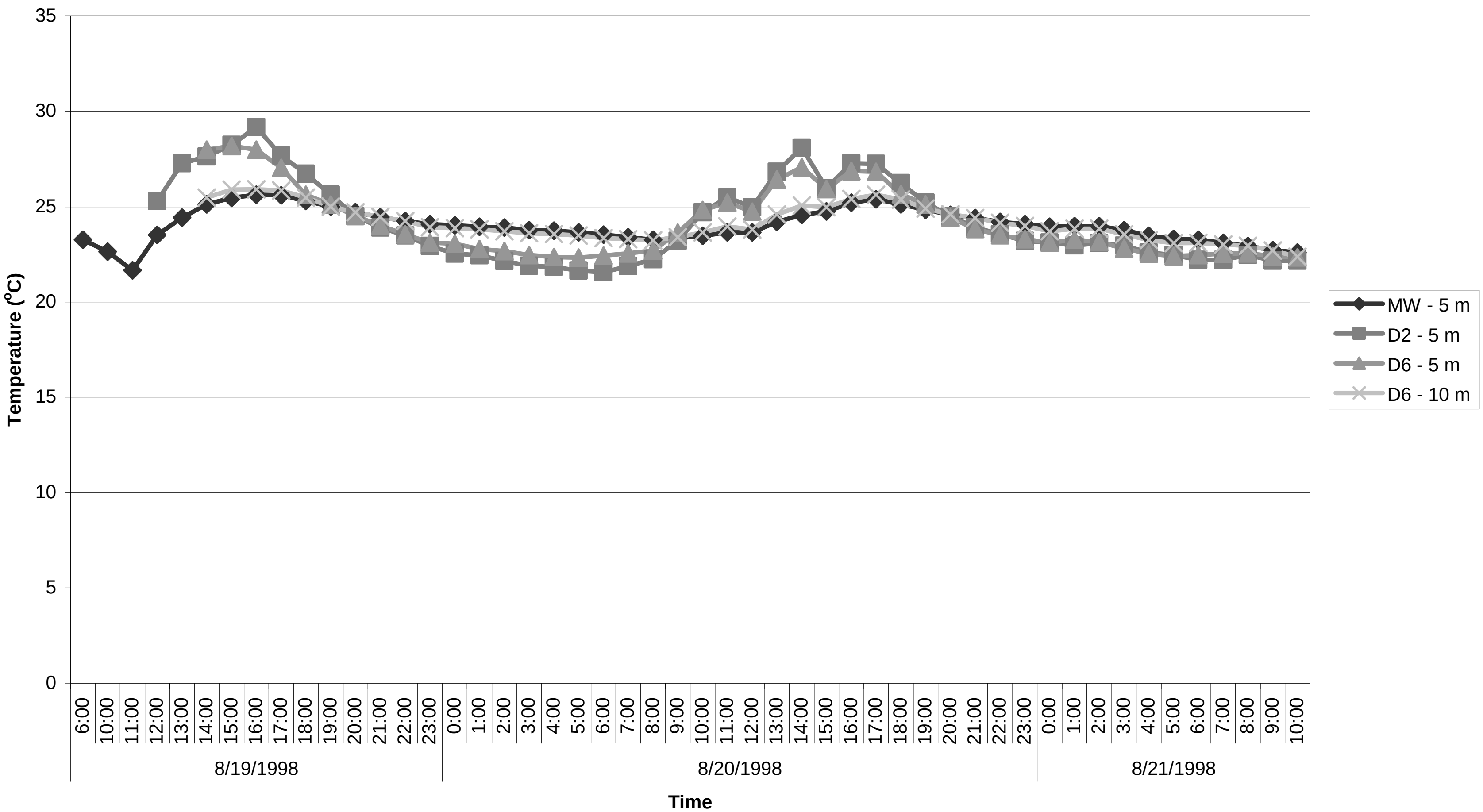


Figure 8. Diurnal changes in temperature at four locations in the Upper Colorado River, August 1998. The first half of the legend (e.g., MW; Moab Wash) indicates site; the second designation indicates distance from shore in meters (e.g., 5 m). Measurements were taken hourly over a 48-h interval using an in situ Hydrolab Water Quality Monitor. Each measurement represents a mean of n = 2 replicates.

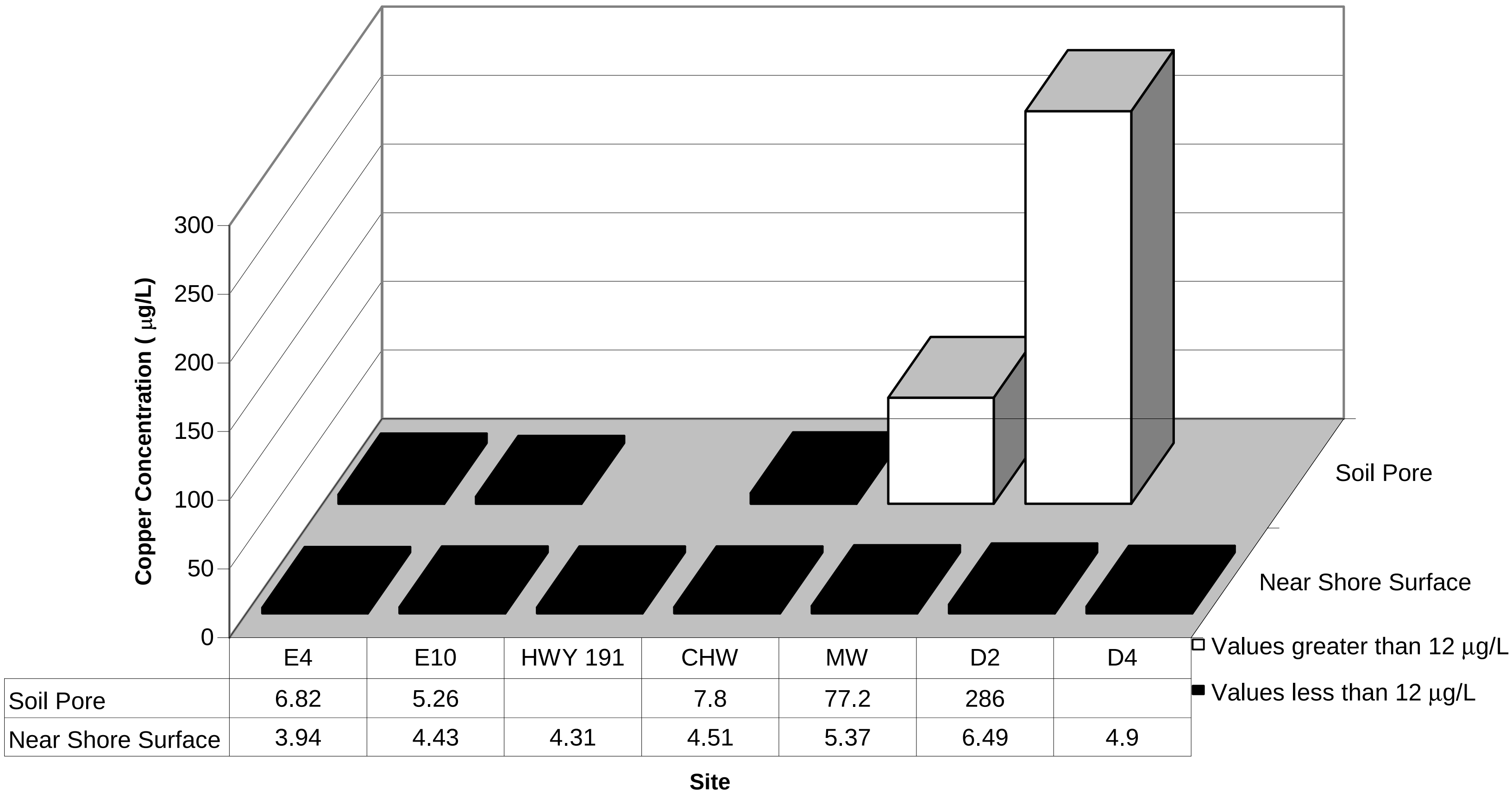
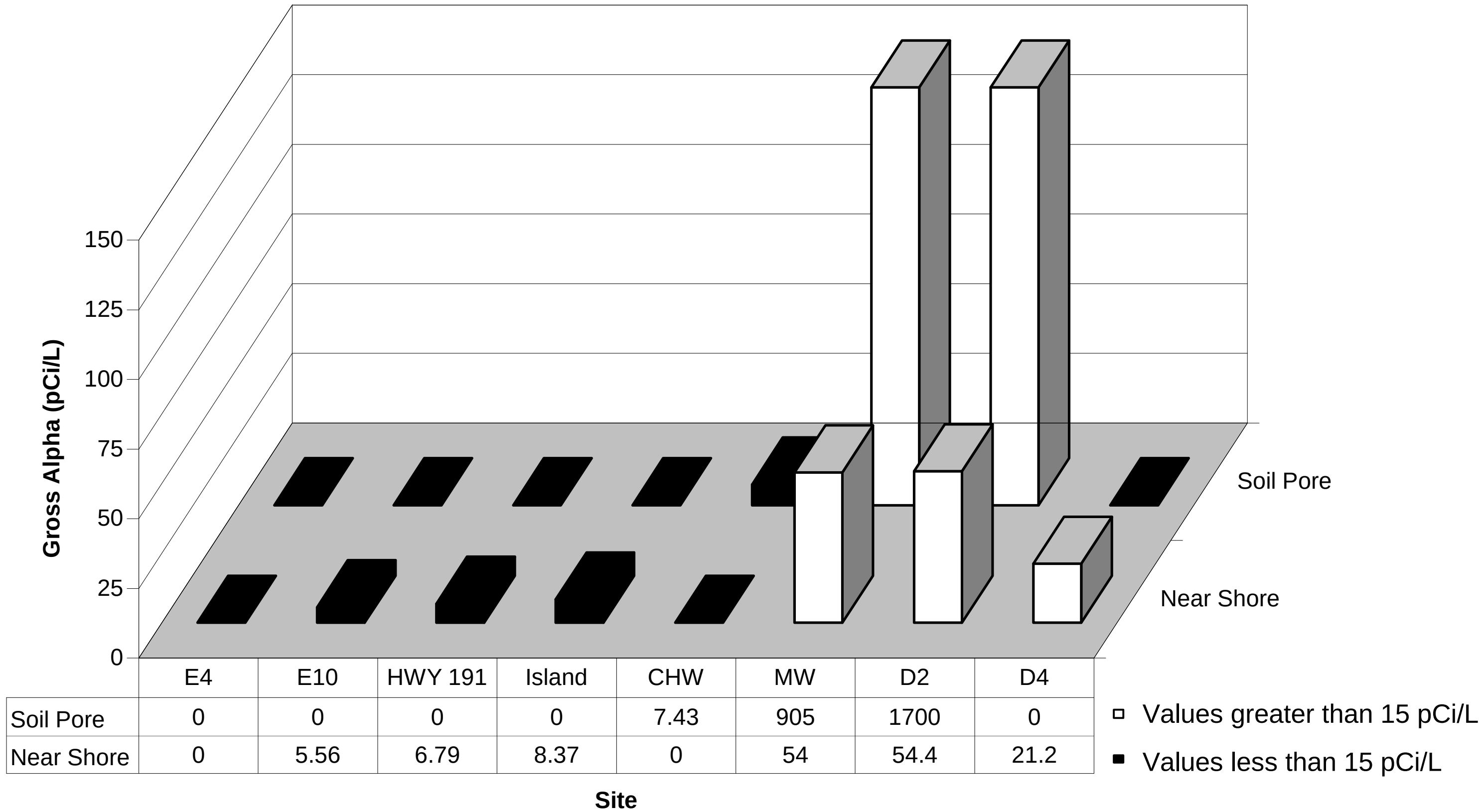
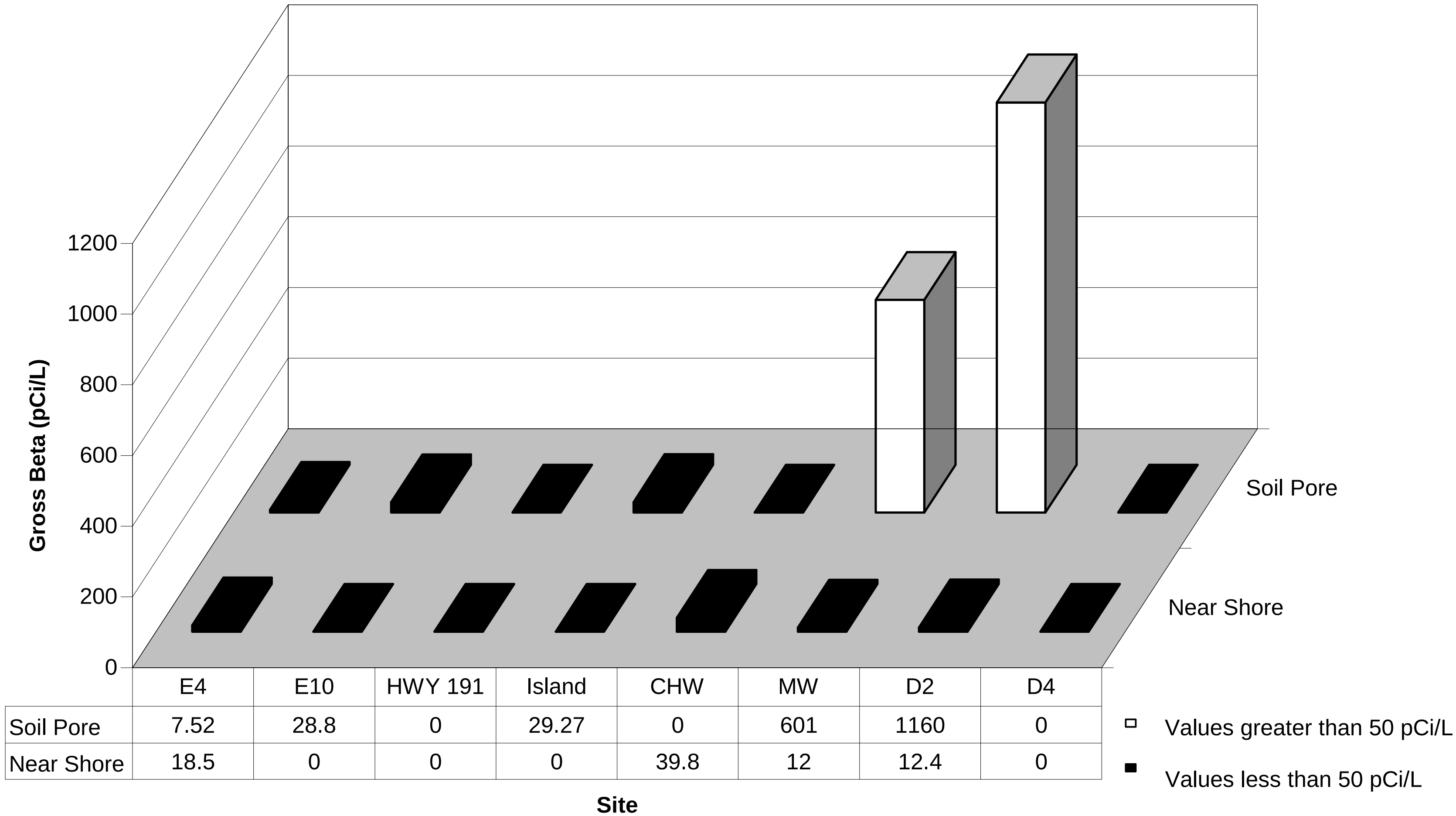


Figure 9. Spatial distribution of dissolved copper (µg/L) in water, August 1998. MW = Moab Wash. Sites downstream of Moab Wash are represented with a "D". Numbers represent multiples of 50 m (e.g., D4 = 200 m downstream). CHW = Courthouse Wash. E4 and E10 are described in text. Utah water quality criteria for copper is 12 µg/L based on water hardness of 100 mg/L.



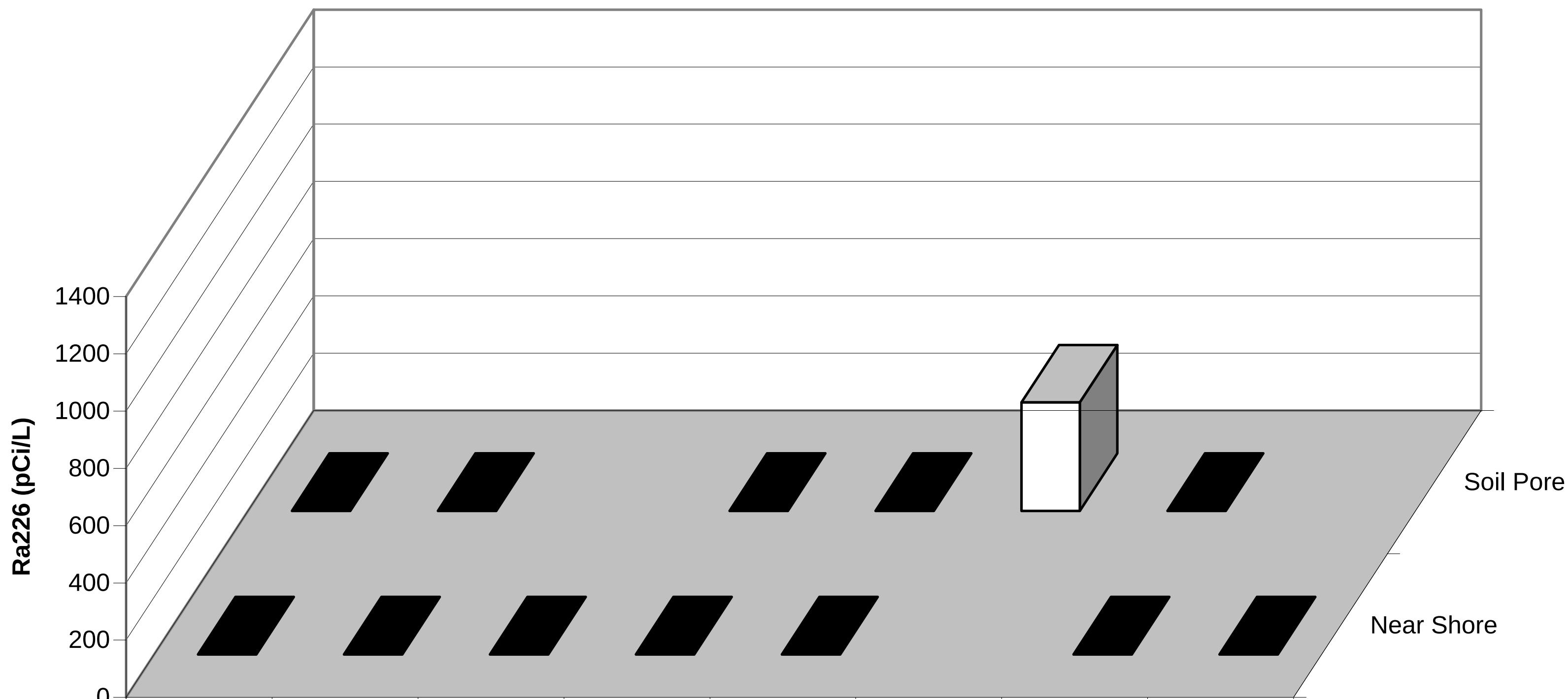
	E4	E10	HWY 191	Island	CHW	MW	D2	D4
Soil Pore	0	0	0	0	7.43	905	1700	0
Near Shore	0	5.56	6.79	8.37	0	54	54.4	21.2

Figure 10. Spatial distribution of gross alpha (pCi/L) in water, August 1998. MW = Moab Wash. Sites downstream of Moab Wash are represented with a "D". Numbers represent multiples of 50 m (e.g., D4 = 200 m downstream). CHW = Courthouse Wash. E4 and E10 are described in text. Utah water quality criteria for gross alpha is 15 pCi/L.



	E4	E10	HWY 191	Island	CHW	MW	D2	D4
Soil Pore	7.52	28.8	0	29.27	0	601	1160	0
Near Shore	18.5	0	0	0	39.8	12	12.4	0

Figure 11. Spatial distribution of gross beta (pCi/L) in water (August 1998). MW = Moab Wash. Sites downstream of Moab Wash are represented with a "D". Numbers represent multiples of 50 m (e.g., D4 = 200 m downstream). CHW = Courthouse Wash. E4 and E10 are described in text. Utah water quality criteria for gross beta is 50 pCi/L.



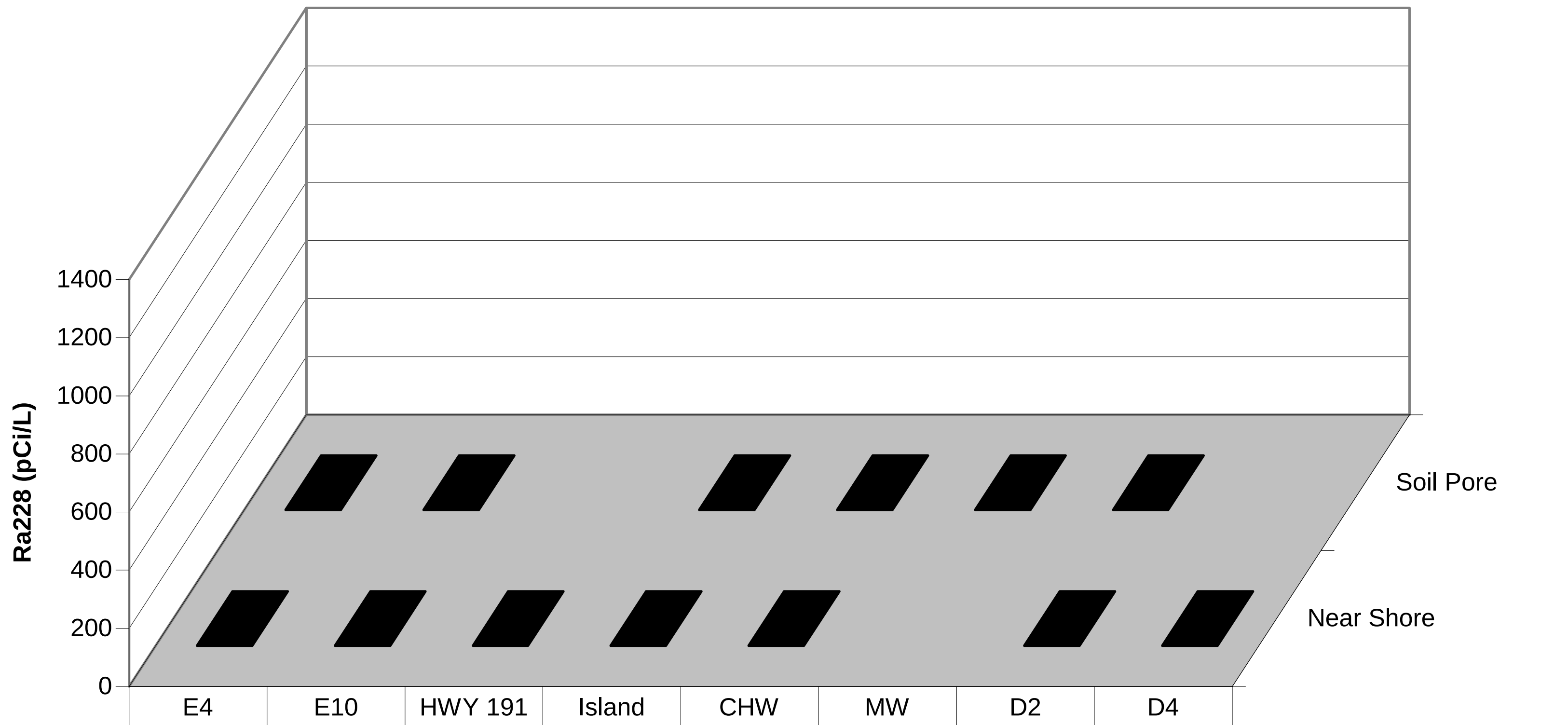
	E4	E10	HWY 191	Island	CHW	MW	D2	D4
Soil Pore	0	0		0	0	379	0	
Near Shore	0	0	0	0	0		0	0

□ Values greater than 5 pCi/L

■ Values less than 5 pCi/L

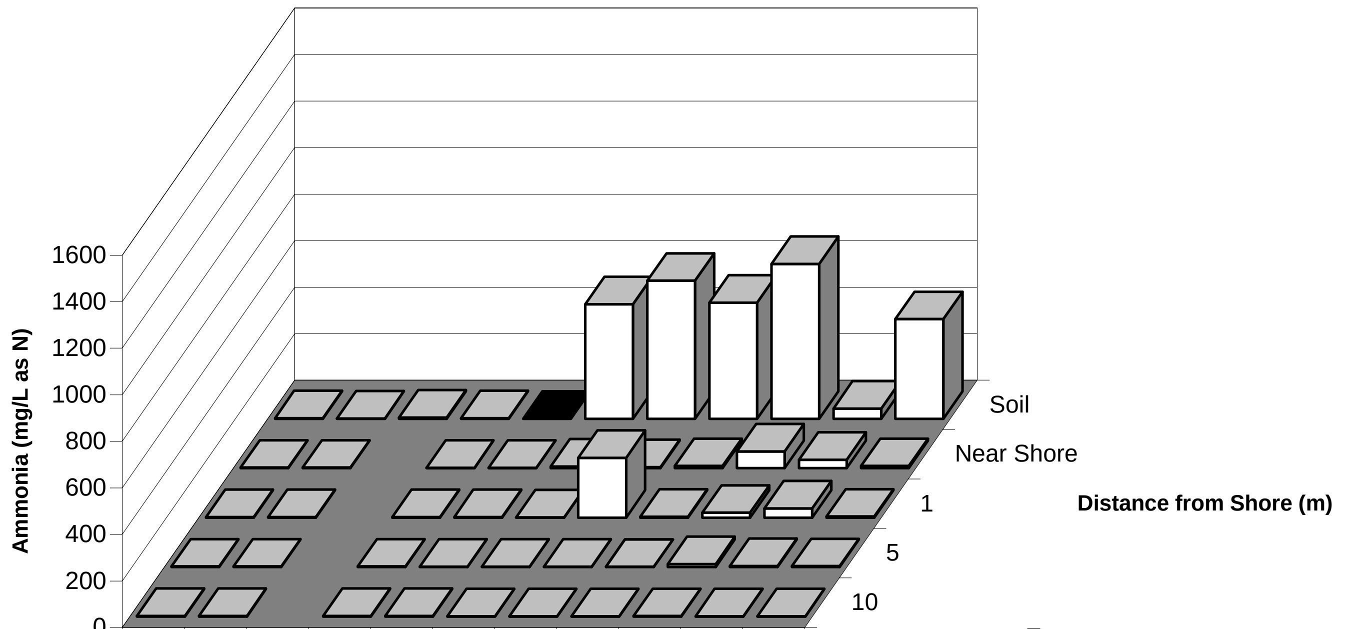
Site

Figure 12. Spatial distribution of Ra226 (pCi/L) in water (August 1998). MW = Moab Wash. Sites upstream and downstream of Moab Wash are represented with a “D”. Numbers represent multiples of 50 m (e.g., D4 = 200 m downstream). CHW = Courthouse Wash. E4 and E10 are described in text.



	E4	E10	HWY 191	Island	CHW	MW	D2	D4
Soil Pore	0	0		0	0	0	0	
Near Shore	0	0	0	0	0		0	0

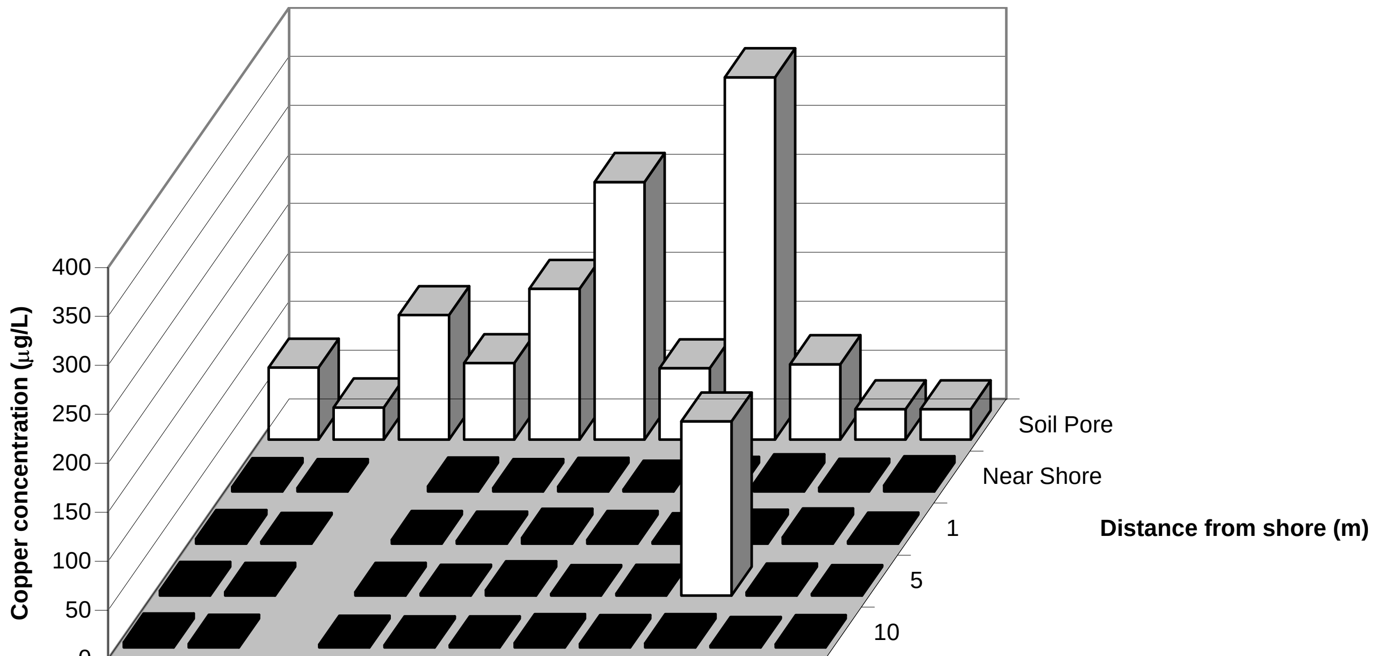
Figure 13. Spatial distribution of Ra228 (pCi/L) in water (August 1998). MW = Moab Wash. Sites upstream and downstream of Moab Wash are represented with a "D". Numbers represent multiples of 50 m (e.g., D4 = 200 m downstream). CHW = Courthouse Wash. E4 and E10 are described in text.



	E4	E10	CHW	UX	U4	MW	D2	D4	D6	D8	D10
Soil	1.98	1.17	4.78	2.25	0	492	593	499	665	43.7	428
Near Shore	2.22	2.26		2.17	1.81	6.57	4.44	9.26	71.5	35.7	9.29
1	2.21	2.21		2.22	2.08	1.28	257	3.55	21.0	39.9	4.50
5	2.27	2.27		2.29	1.96	1.28	1.28	1.08	12.6	4.51	3.85
10	2.22	2.01		2.27	2.29	1.28	1.28	1.38	1.79	1.24	1.24

Site

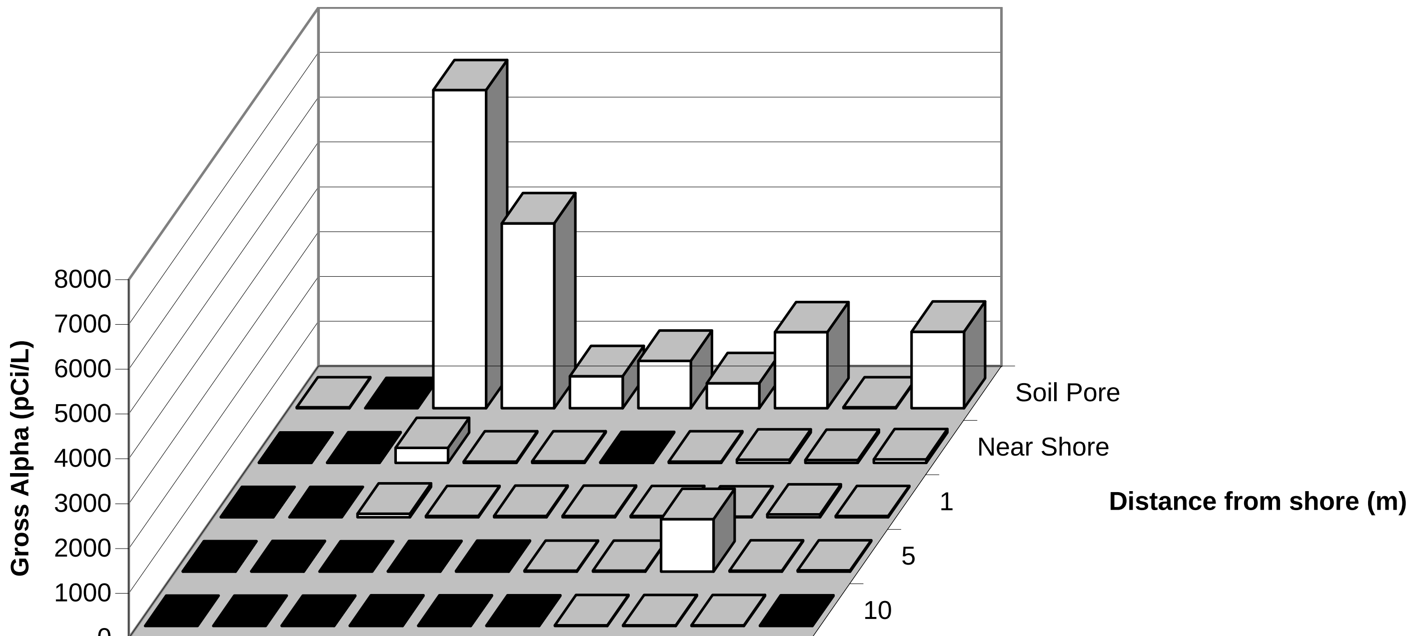
Figure 14. Spatial distribution of total ammonia (mg/L) in water (February 1999). MW = Moab Wash. Sites upstream and downstream of Moab Wash are represented with a "U" and "D", respectively. Numbers represent multiples of 50 m (e.g., D4 = 200 m downstream). CHW = Courthouse Wash. E4 and E10 are described in text. Utah 30-d chronic ammonia criteria is 1.21 mg/L as N based on pH = 8.0 and temperature = 25 °C for class 3B rivers.



	E4	E10	CHW	UX	U4	MW	D2	D4	D6	D8	D10
Soil Pore	73.5	32.7	127	78.1	154	263	72.9	370	76.9	30.9	30.9
Near Shore	4.67	3.59		5.3	3.49	4.84	1.91	5.9	8.36	3.84	6.36
1	4.65	1.77		3.84	2.96	5.85	3.69	1.37	4.86	6.28	2.15
5	4.86	3.7		3.71	1.81	5.4	1.5	1.93	178	3.34	1.69
10	5.39	3.87		3.16	2.3	2.4	4.63	3.63	4.54	1.74	3.56

Site

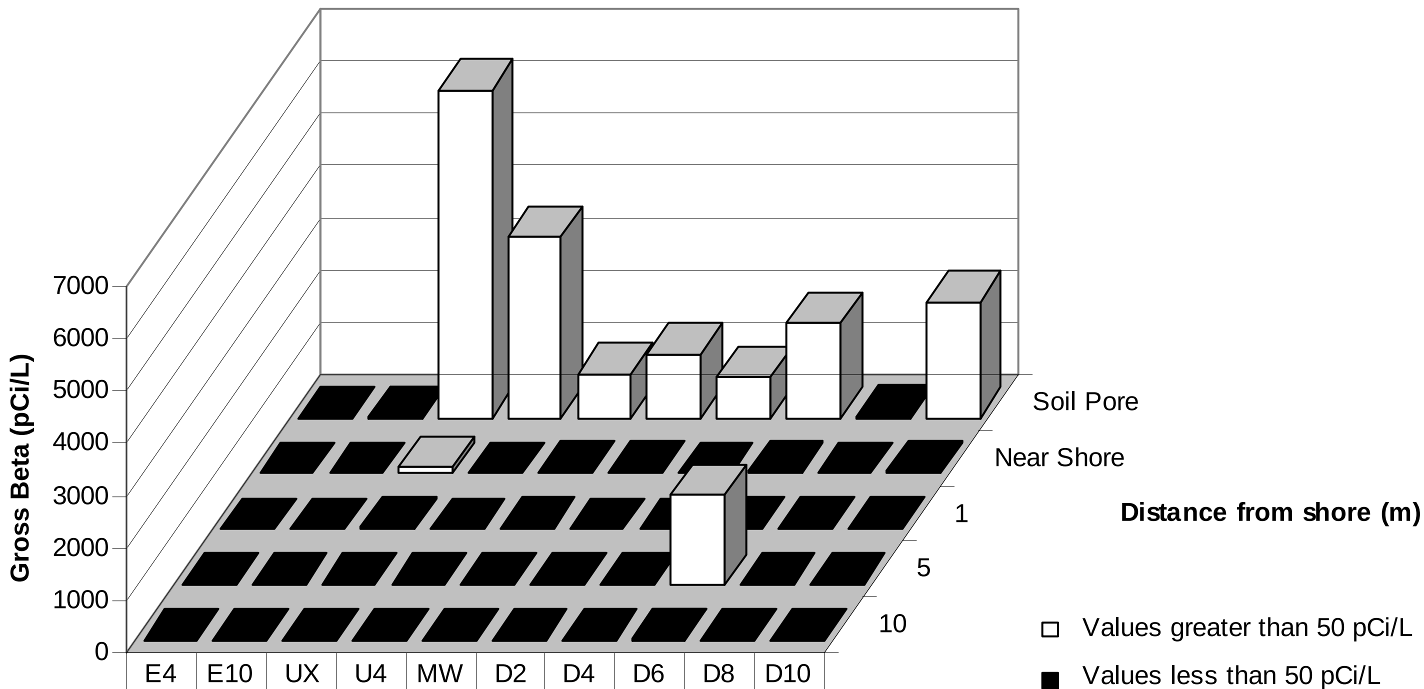
Figure 15. Spatial distribution of dissolved copper (µg/L) in water (February 1999). MW = Moab Wash. Sites upstream and downstream of Moab Wash are represented with a "U" and "D", respectively. Numbers represent multiples of 50 m (e.g., D4 = 200 m downstream). CHW = Courthouse Wash. E4 and E10 are described in text. Utah water quality criteria for copper is 12 µg/L based on water hardness of 100 mg/L.



	E4	E10	UX	U4	MW	D2	D4	D6	D8	D10
Soil Pore	24.0	6.00	7100	4130	720	1060	560	1700	21.3	1710
Near Shore	2.30	6.10	330	27.3	32.0	14.6	26.3	72.0	65.0	78.0
1	3.10	0	77.0	24.0	29.0	28.6	24.0	16.6	60.0	25.8
5	6.20	9.00	0.70	9.90	14.3	18.6	18.1	1170	22.3	27.3
10	3.60	2.10	5.80	8.70	11.2	11.5	17.7	18.6	17.0	10.9

Site

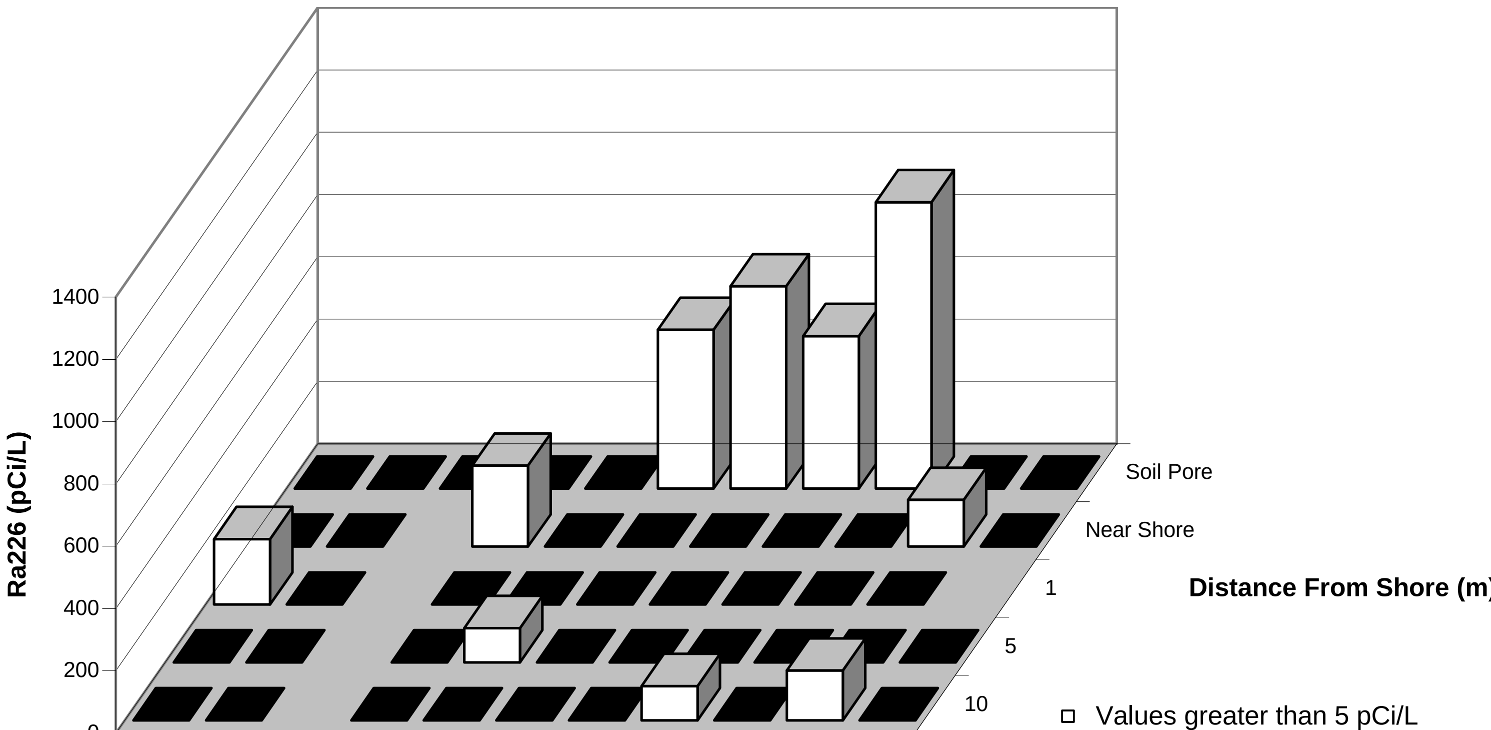
Figure 16. Spatial distribution of gross alpha (pCi/L) in water (February 1999). MW = Moab Wash. Sites upstream and downstream of Moab Wash are represented with a "U" and "D", respectively. Numbers represent multiples of 50 m (e.g., D4 = 200 m downstream). E4 and E10 are described in text. UX = 400 m upstream of Moab Wash. Utah water quality criteria for gross alpha is 15 pCi/L.



	E4	E10	UX	U4	MW	D2	D4	D6	D8	D10
Soil Pore	-9.00	9.80	6270	3460	845	1220	790	1820	42.4	2210
Near Shore	5.10	4.30	125	19.2	22.7	20.6	14.8	48.6	18.7	37.9
1	5.20	7.30	31.7	10.2	17.1	11.4	9.90	26.7	26.2	14.9
5	6.10	5.80	6.00	16.8	8.70	11.7	15.2	1720	10.7	15.4
10	4.30	5.30	3.80	7.20	6.10	11.6	9.40	13.8	4.00	8.90

Site

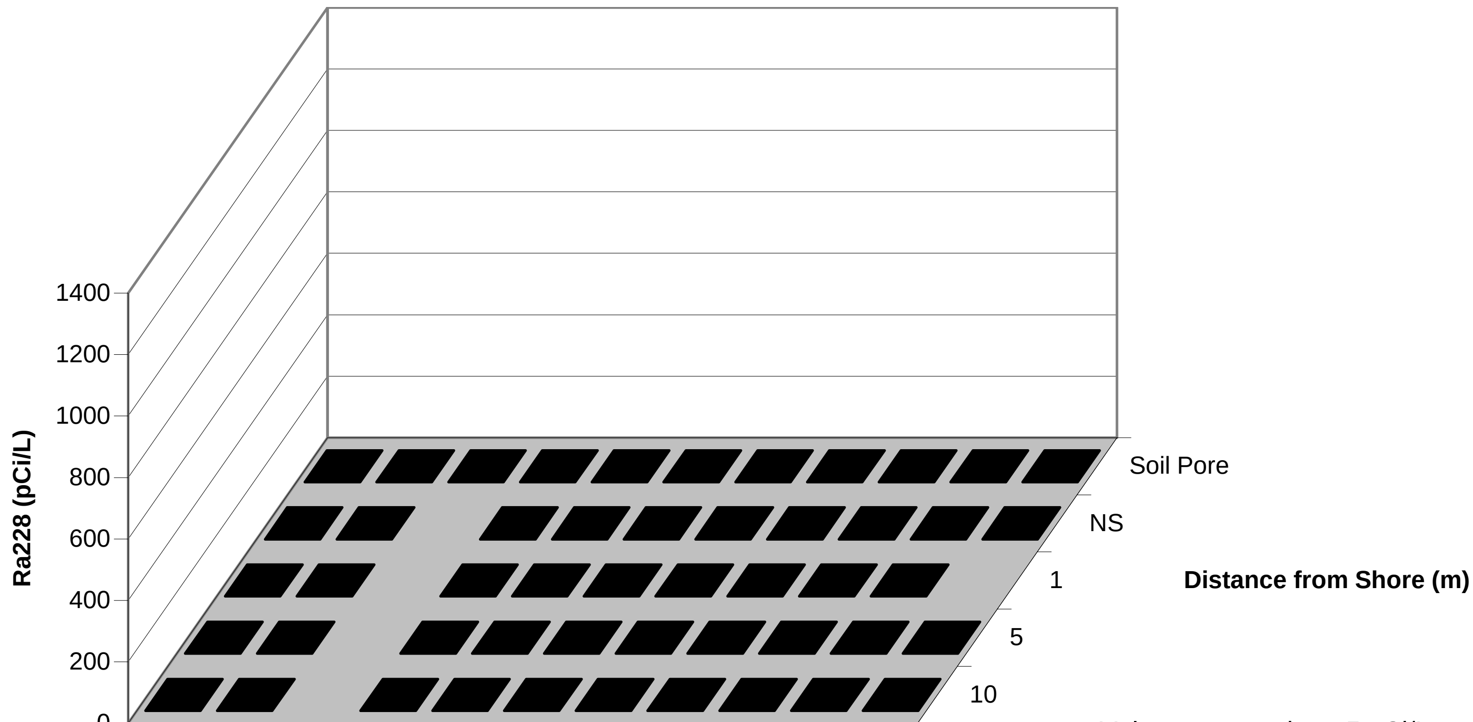
Figure 17. Spatial distribution of gross beta (pCi/L) in water (February 1999). MW = Moab Wash and sites upstream and downstream of Moab Wash are represented with a "U" and "D", respectively. Numbers represent multiples of 50 m (e.g., D4 = 200 m downstream). E4 and E10 are described in text. UX = 400 m upstream of Moab Wash. Utah water quality criteria for gross beta is 50 pCi/L.



	E4	E10	CHW	UX	U4	MW	D2	D4	D6	D8	D10
Soil Pore	0	0	0	0	0	510	650	490	920	0	0
Near Shore	0	0		260	0	0	0	0	0	150	0
1	210	0		0	0	0	0	0	0	0	
5	0	0		0	110	0	0	0	0	0	0
10	0	0		0	0	0	0	110	0	160	0

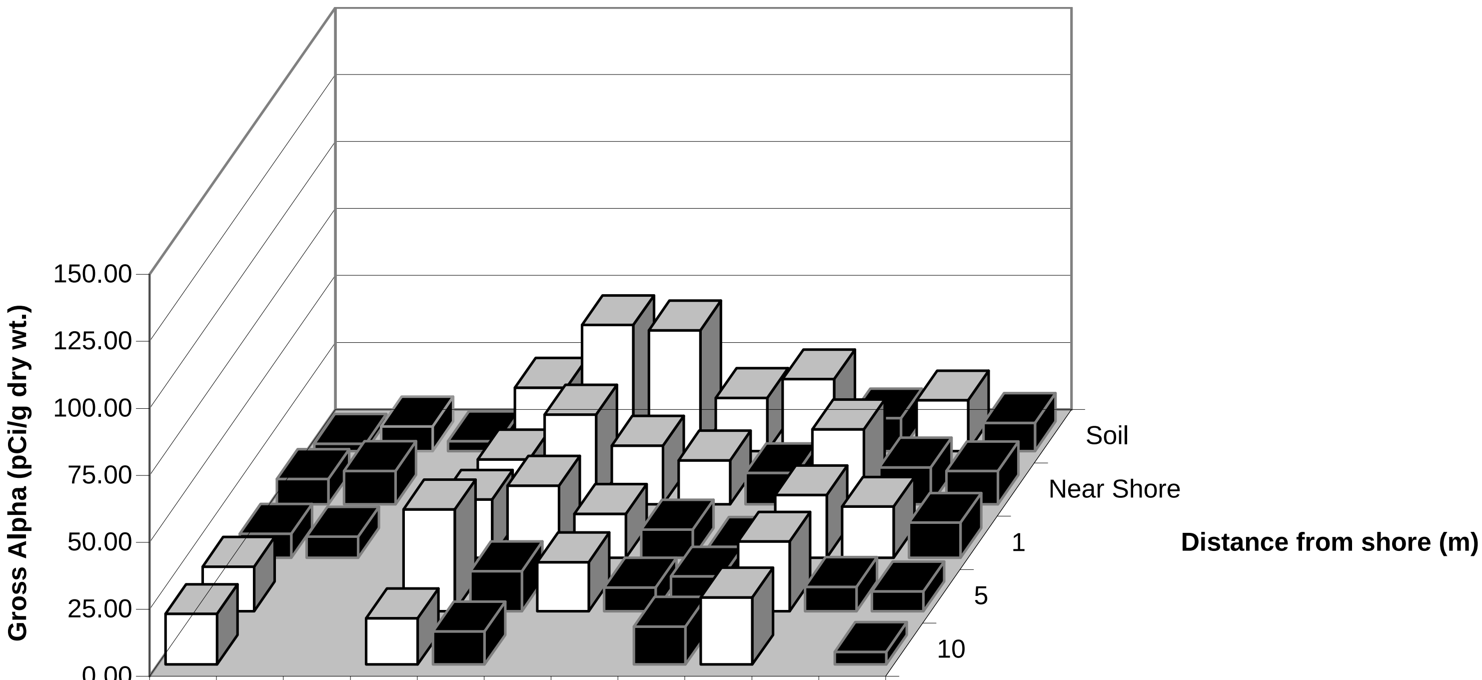
Site

Figure 18. Spatial distribution of Ra226 (pCi/L) in water (February 1999). MW = Moab Wash. Sites upstream and downstream of Moab Wash are represented with a "U" and "D", respectively. Numbers represent multiples of 50 m (e.g., D4 = 200 m downstream). CHW = Courthouse Wash. E4 and E10 are described in text. UX = 400 m upstream of Moab Wash.



	E4	E10	CHW	UX	U4	MW	D2	D4	D6	D8	D10
Soil Pore	0	0	0	0	0	0	0	0	0	0	0
NS	0	0		0	0	0	0	0	0	0	0
1	0	0		0	0	0	0	0	0	0	
5	0	0		0	0	0	0	0	0	0	0
10	0	0		0	0	0	0	0	0	0	0

Figure 19. Spatial distribution of Ra228 (pCi/L) in water (February 1999). MW = Moab Wash. Sites upstream and downstream of Moab Wash are represented with a "U" and "D", respectively. Numbers represent multiples of 50 m (e.g., D4 = 200 m downstream). CHW = Courthouse Wash. E4 and E10 are described in text. UX = 400 m upstream of Moab Wash.



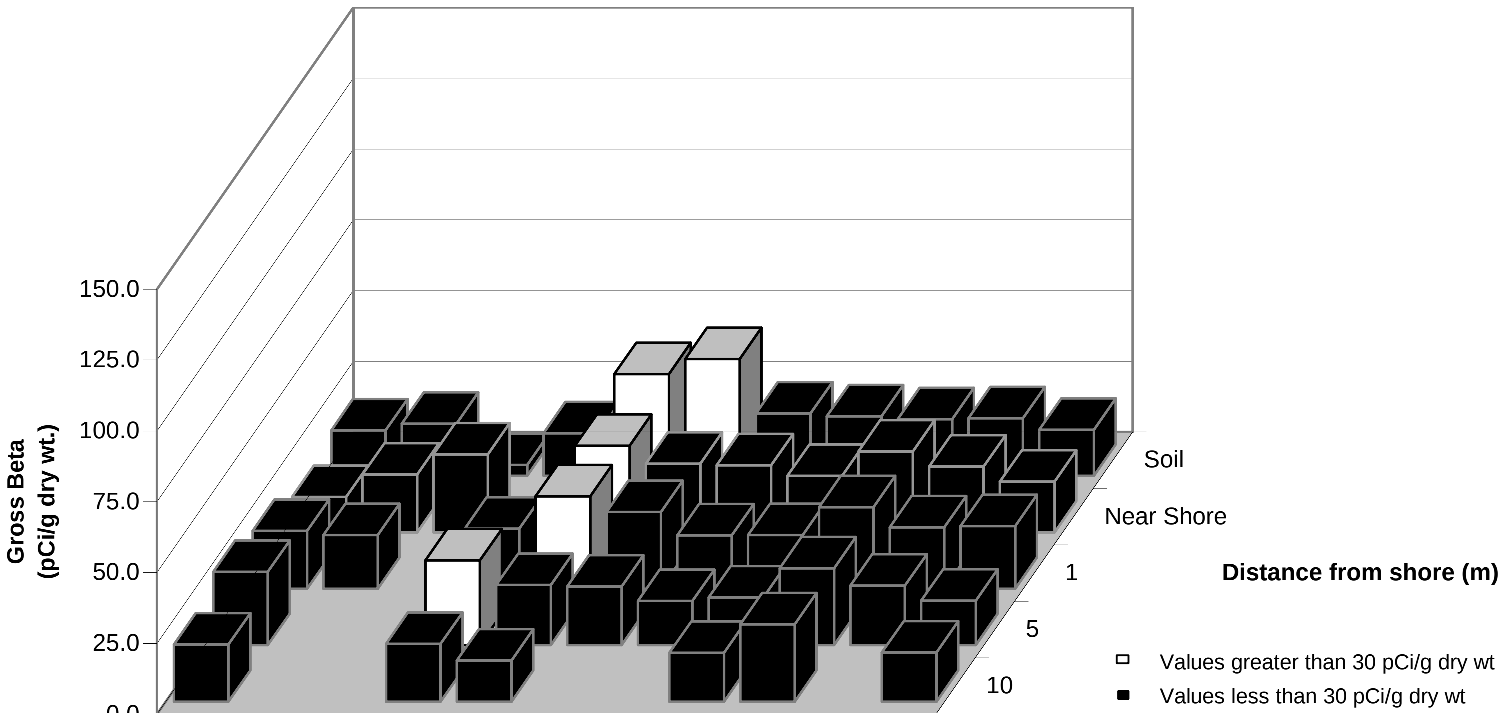
	E4	E10	CHW	UX	U4	MW	D2	D4	D6	D8	D10
Soil	2.70	9.20	3.70	23.7	47.0	45.0	19.8	26.8	12.2	18.9	10.5
Near Shore	9.50	12.5		16.8	33.5	21.9	16.4	11.7	28.0	13.8	12.4
1	9.00	7.90		21.7	26.9	16.4	10.5	4.10	23.4	19.1	13.1
5	16.6			38.0	14.9	18.3	8.80	13.0	26.0	9.10	7.40
10	18.9			17.3	12.4			14.2	25.1		4.70

Site

□ Values greater than 15 pCi / g dry wt.

■ Values less than 15 pCi / g dry wt.

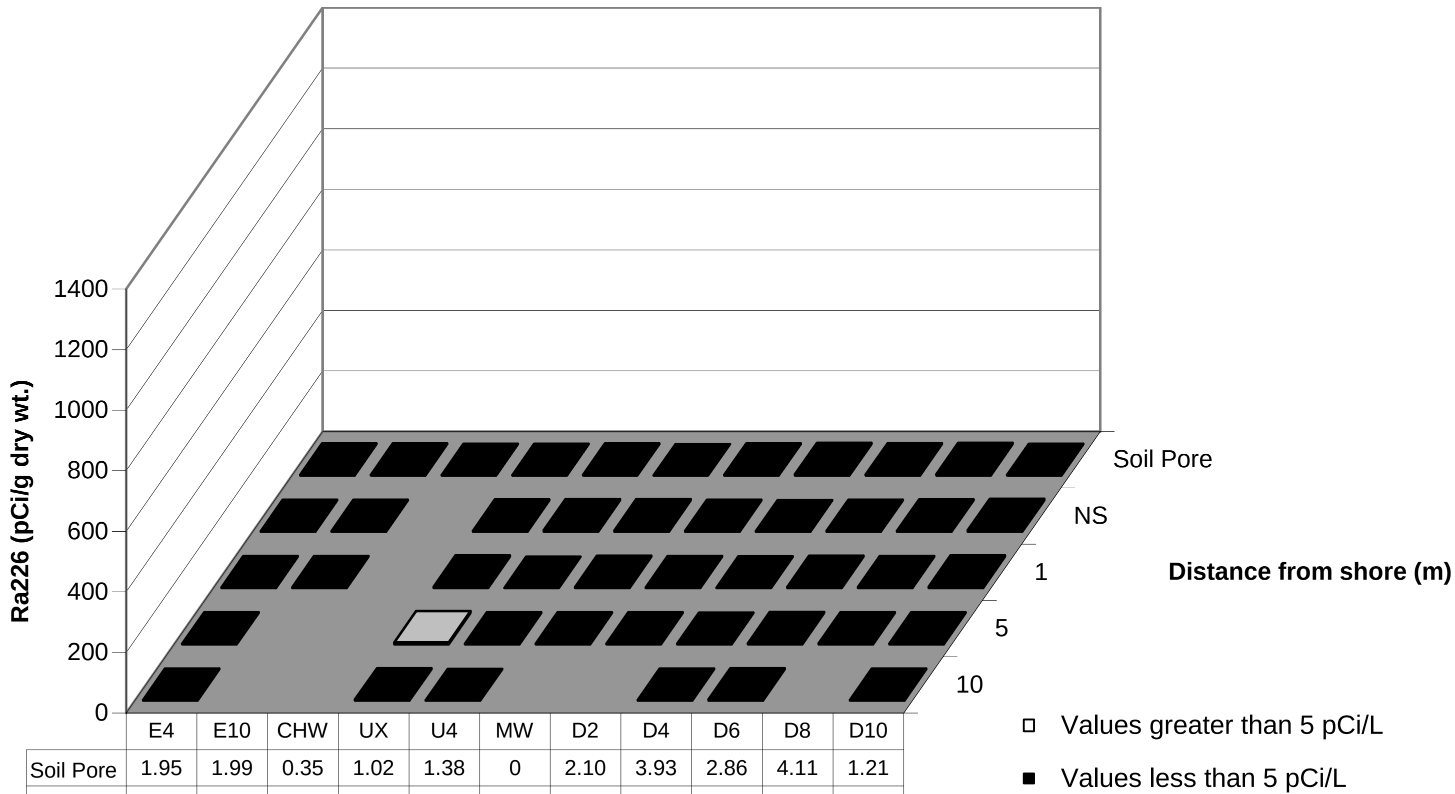
Figure 20. Spatial distribution of gross alpha (pCi/g dry wt.) in sediment (February 1999). MW = Moab Wash. Sites upstream and downstream of Moab Wash are represented with a "U" and "D", respectively. Numbers represent multiples of 50 m (e.g., D4 = 200 m downstream). CHW = Courthouse Wash. E4 and E10 are described in text. UX = 400 m upstream of Moab Wash. Utah water quality criteria for gross alpha is 15 pCi/g dry wt..



	E4	E10	CHW	UX	U4	MW	D2	D4	D6	D8	D10
Soil	16.1	18.4	3.90	15.1	36.0	41.2	22.1	21.0	20.0	20.4	16.3
Near Shore	12.5	20.4	27.5		30.6	24.2	23.7	20.0	28.5	23.2	17.9
1	20.5	19.0		21.2	32.6	27.1	18.9	19.0	28.8	21.7	22.1
5	25.9			30.0	21.3	20.7	15.6	16.9	27.2	21.0	15.7
10	20.2			20.4	14.5			17.3	27.4		17.4

Site

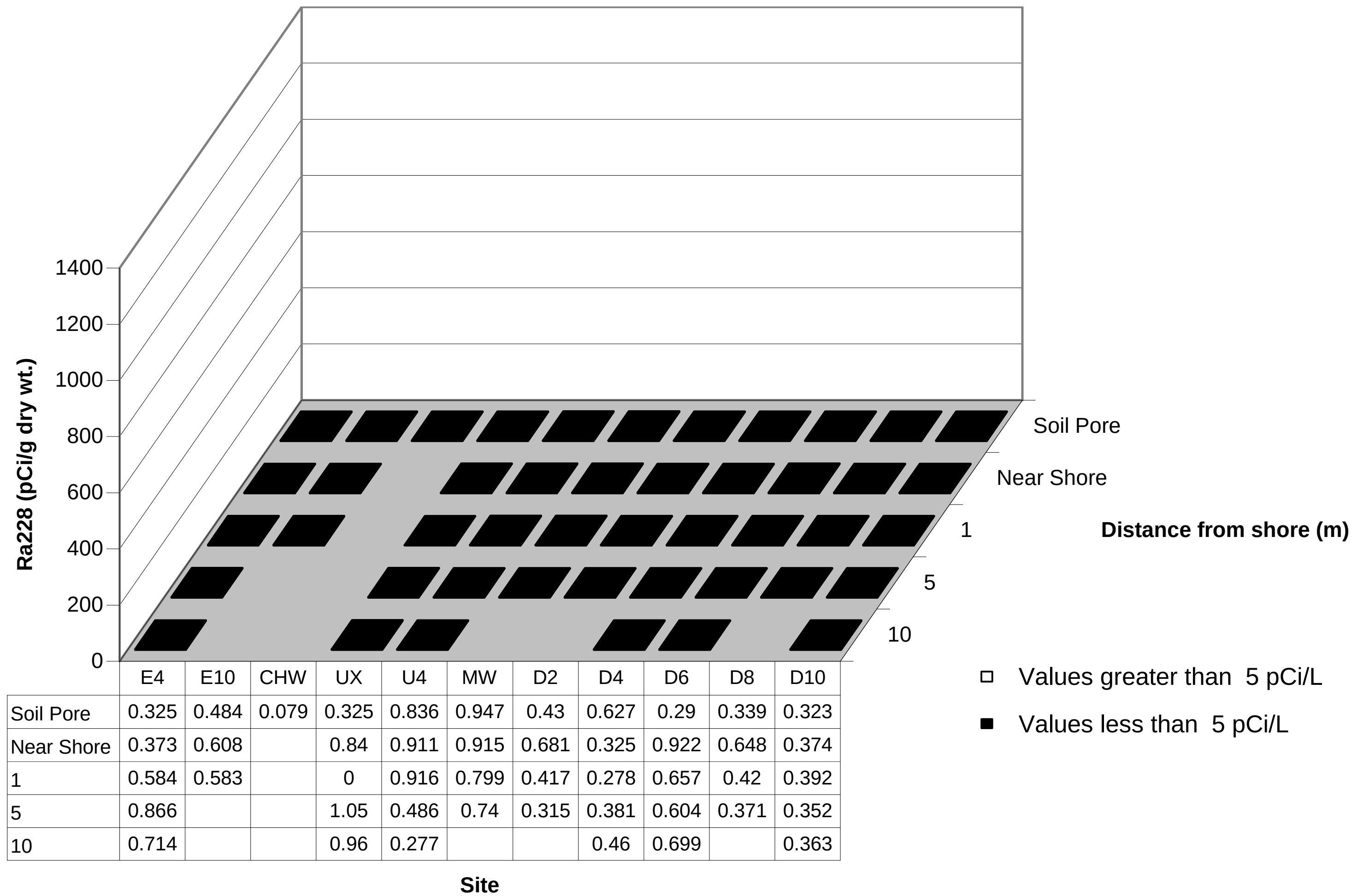
Figure 21. Spatial distribution of gross beta (pCi/g dry wt.) in sediment (February 1999). MW = Moab Wash. Sites upstream and downstream of Moab Wash are represented with a "U" and "D", respectively. Numbers represent multiples of 50 m (e.g., D4 = 200 m downstream). CHW = Courthouse Wash. E4 and E10 are described in text. UX = 400 m upstream of Moab Wash. Utah water quality criteria for gross beta is 50 pCi/g dry wt.



	E4	E10	CHW	UX	U4	MW	D2	D4	D6	D8	D10
Soil Pore	1.95	1.99	0.35	1.02	1.38	0	2.10	3.93	2.86	4.11	1.21
NS	1.97	2.19		2.96	4.45	4.13	2.36	1.30	2.29	3.29	4.61
1	1.92	1.79		4.10	0	3.29	1.60	1.19	3.19	2.50	3.75
5	2.38			6.60	2.17	2.40	1.94	1.49	3.39	2.62	2.27
10	2.13			3.60	0.79			1.67	3.65		1.77

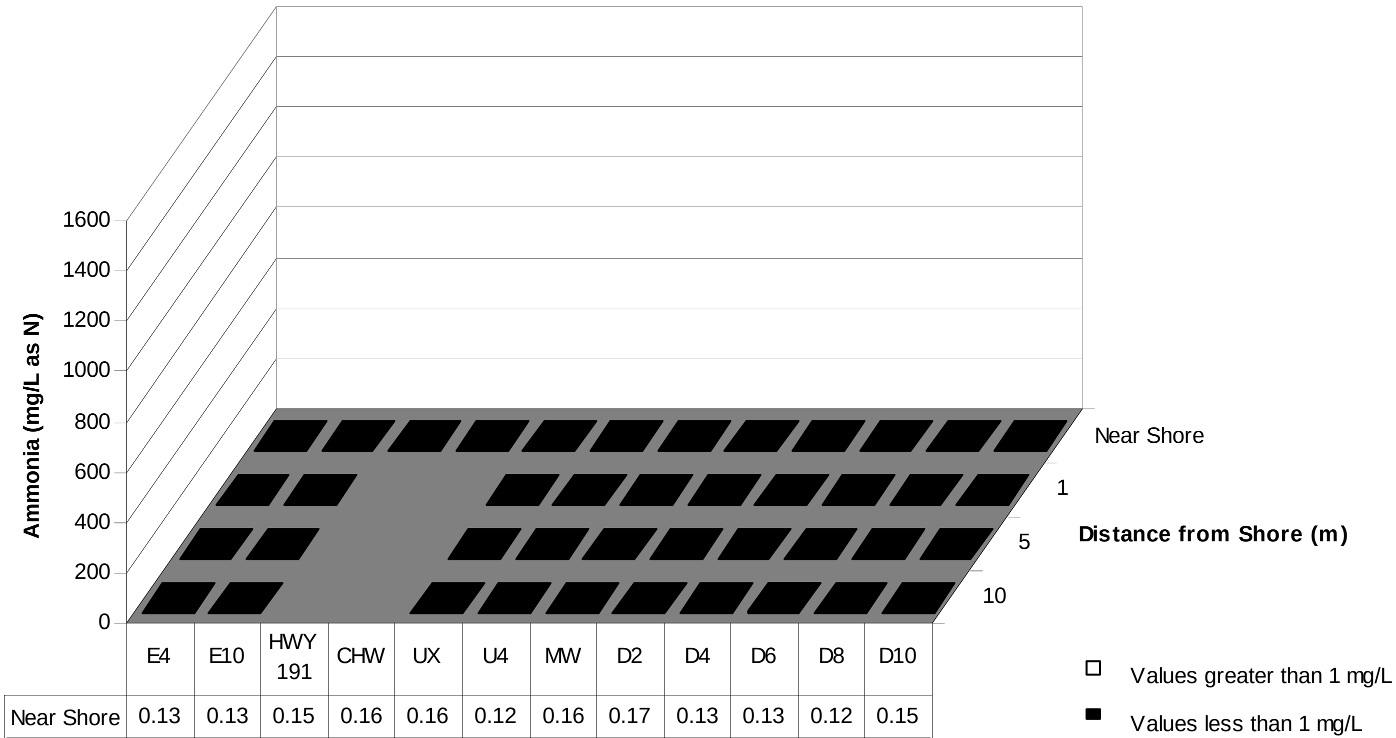
Site

Figure 22. Spatial distribution of Ra226 (pCi /g dry wt.) in sediment (February 1999). MW = Moab Wash. Sites upstream and downstream of Moab Wash are represented with a "U" and "D", respectively. Numbers represent multiples of 50 m (e.g., D4 = 200 m downstream). CHW = Courthouse Wash. E4 and E10 are described in text. UX = 400 m upstream of Moab Wash.



	E4	E10	CHW	UX	U4	MW	D2	D4	D6	D8	D10
Soil Pore	0.325	0.484	0.079	0.325	0.836	0.947	0.43	0.627	0.29	0.339	0.323
Near Shore	0.373	0.608		0.84	0.911	0.915	0.681	0.325	0.922	0.648	0.374
1	0.584	0.583		0	0.916	0.799	0.417	0.278	0.657	0.42	0.392
5	0.866			1.05	0.486	0.74	0.315	0.381	0.604	0.371	0.352
10	0.714			0.96	0.277			0.46	0.699		0.363

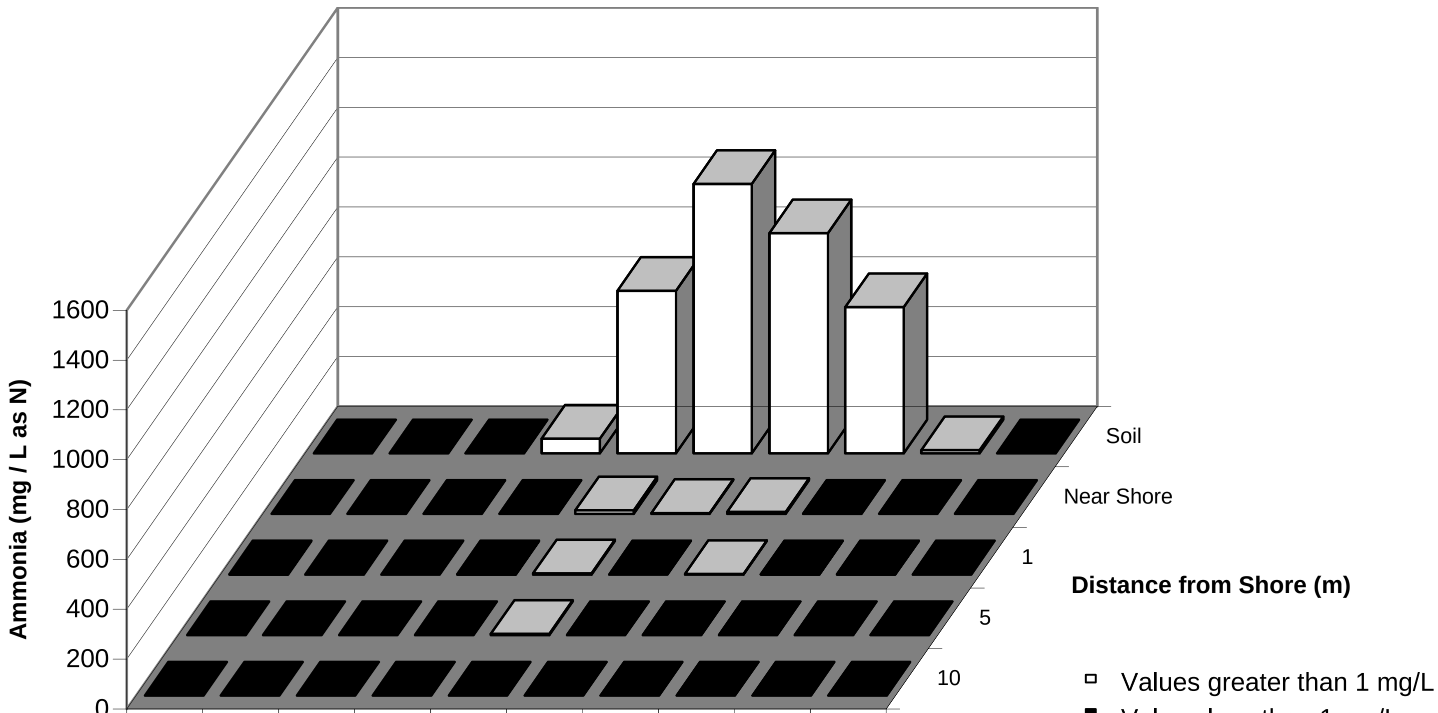
Figure 23. Spatial distribution of Ra228 (pCi /g dry wt.) in sediment (February 1999). MW = Moab Wash. Sites upstream and downstream of Moab Wash are represented with a "U" and "D", respectively. Numbers represent multiples of 50 m (e.g., D4 = 200 m downstream). CHW = Courthouse Wash. E4 and E10 are described in text. UX = 400 m upstream of Moab Wash.



	E4	E10	HWY 191	CHW	UX	U4	MW	D2	D4	D6	D8	D10
Near Shore	0.13	0.13	0.15	0.16	0.16	0.12	0.16	0.17	0.13	0.13	0.12	0.15
1	0.13	0.13			0.15	0.13	0.17	0.15	0.13	0.40	0.17	0.15
5	0.13	0.13			0.14	0.13	0.17	0.15	0.13	0.40	0.17	0.14
10	0.13	0.13			0.16	0.13	0.17	0.14	0.13	0.40	0.17	0.15

Site

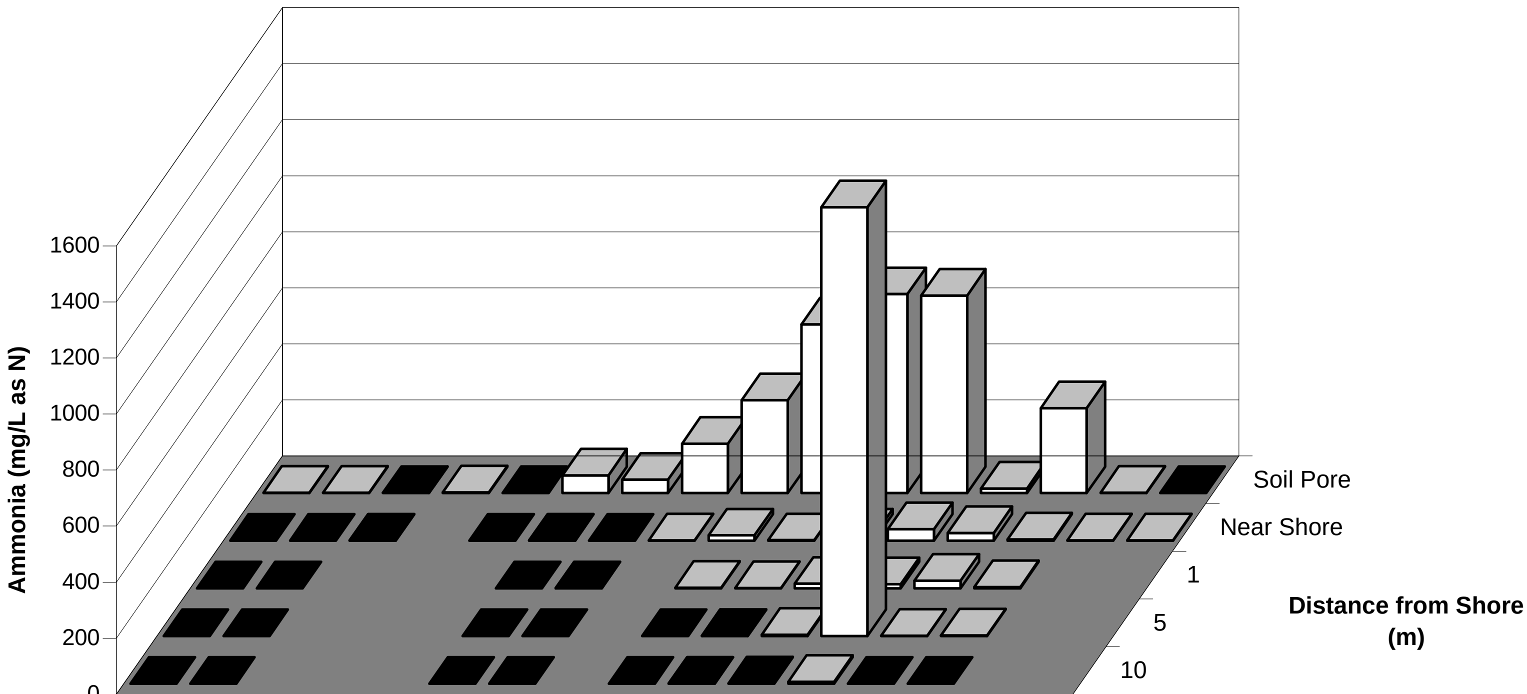
Figure 24. Spatial distribution of total ammonia (mg/L) in water (June 1999). MW = Moab Wash. Sites upstream and downstream of Moab Wash are represented with a "U" and "D", respectively. Numbers represent multiples of 50 m (e.g., D4 = 200 m downstream). CHW = Courthouse Wash. E4 and E10 are described in text. Utah 30-d chronic ammonia criteria is 1.21 mg/L as N based on pH = 8.0 and temperature = 25 °C for class 3B rivers.



	E4	E10	UX	U4	MW	D2	D4	D6	D8	D10
Soil	0.11	0.17	0.14	58.89	652.67	1082	884.01	587.31	13.35	0.11
Near Shore	0.21	0.14	0.21	0.29	14.95	3.81	8.12	0.43	0.36	0.11
1	0.21	0.18	0.18	0.21	5.16	0.08	1.89	0.36	0.31	0.11
5	0.23	0.18	0.23	0.23	4.47	0.12	0.12	0.1	0.11	0.12
10	0.23	0.19	0.24	0.23	0.12	0.14	0.17	0.16	0.18	0.21

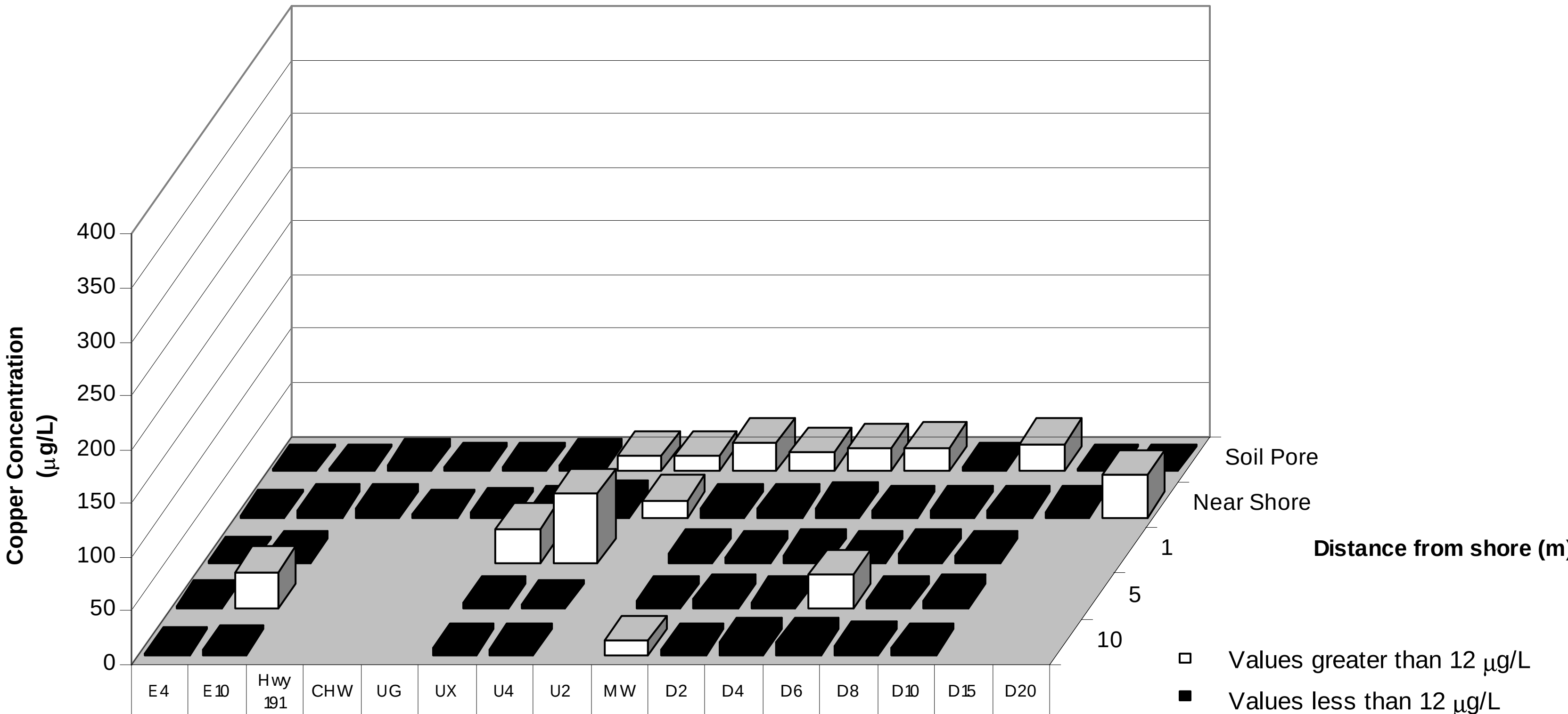
Site

Figure 25. Spatial distribution of total ammonia (mg/L) in water (September 1999). MW = Moab Wash. Sites upstream and downstream of Moab Wash are represented with a "U" and "D", respectively. Numbers represent multiples of 50 m (e.g., D4 = 200 m downstream). UX = 400 meters upstream of Moab Wash. E4 and E10 described in text. Utah 30-d chronic ammonia criteria is 1.21 mg/L as N based on pH = 8.0 and temperature = 25 ^{o}C for class 3B rivers.



	E4	E10	Hwy 191	CHW	UG	UX	U4	U2	MW	D2	D4	D6	D8	D10	D15	D20
Soil Pore	1.30	1.23	0.91	3.05	0.34	63.6	47.8	177	332	602	710	705	16.7	303	1.31	0.41
Near Shore	0.21	0.24	0.84		0.20	0.85	0.38	1.19	19.1	2.02	18.5	41.4	27.9	5.18	1.26	1.17
1	0.21	0.26				0.29	0.95		2.41	1.39	16.8	15.1	27.9	5.15		
5	0.11	0.35				0.14	0.17		0.42	0.83	4.68	1530	1.01	2.45		
10	0.09	0.20				0.37	0.14		0.26	0.57	0.95	6.74	0.44	0.45		

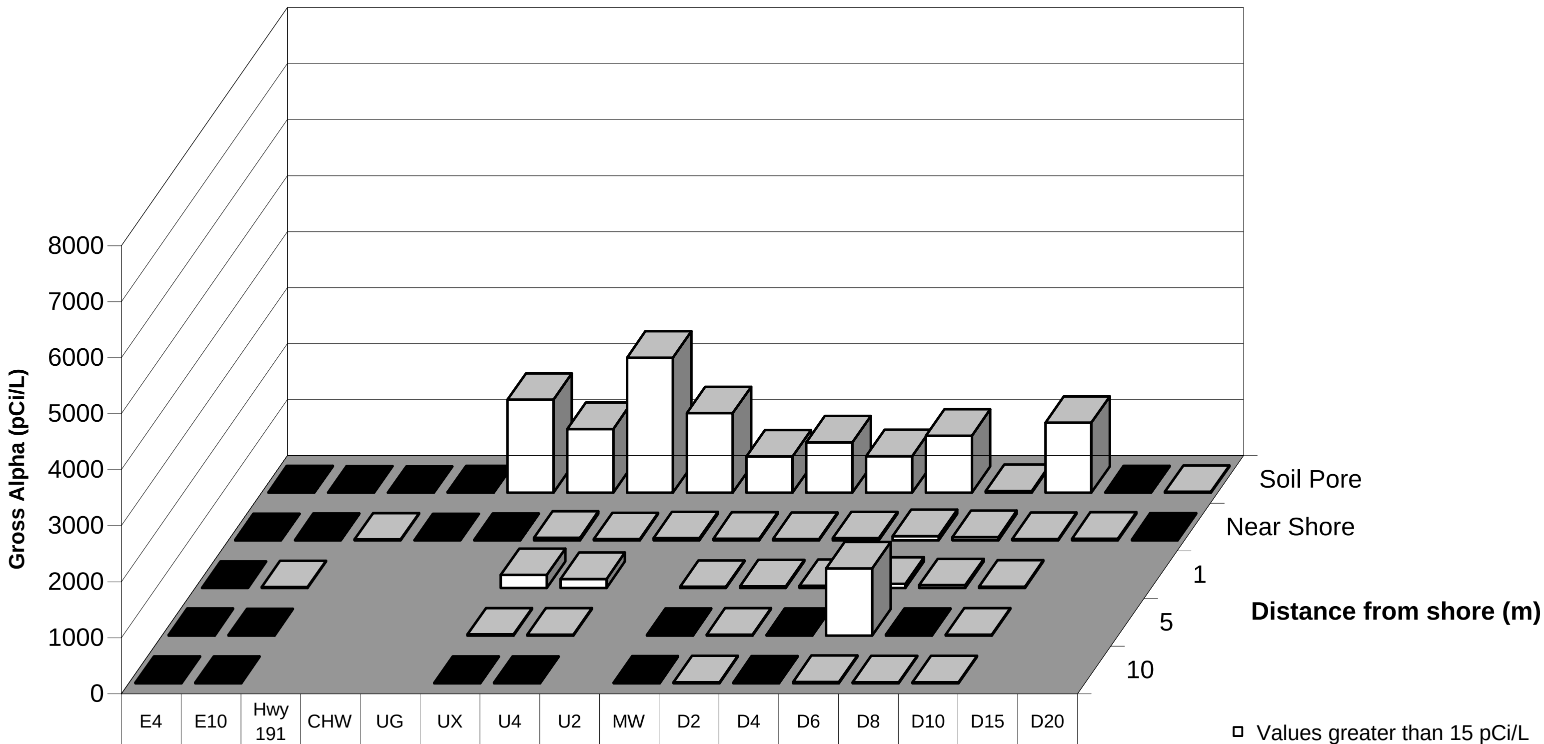
Figure 26. Spatial distribution of total ammonia (mg/L) in water (February 2000). MW = Moab Wash. Sites upstream and downstream of Moab Wash are represented with a "U" and “D”, respectively. Numbers represent multiples of 50 m (e.g., D4 = 200 m downstream). CHW = Courthouse Wash. E4 and E10 are described in text. UX = 400 m upstream of Moab Wash. Utah 30-d chronic ammonia criteria is 1.21 mg/L as N based on pH = 8.0 and temperature = 25 °C for class 3B rivers.



	E4	E10	Hwy 191	CHW	UG	UX	U4	U2	MW	D2	D4	D6	D8	D10	D15	D20
Soil Pore	195	177	6.14	3.89	4.20	6.34	14.5	14.5	26.3	17.5	21.1	21.9	4.22	25.9	2.46	170
Near Shore	190	6.60	7.78	2.60	4.75	5.10	11.5	16.0	7.49	8.25	9.10	6.09	6.48	6.21	5.28	39.4
1	176	6.53				32.4	64.0		8.43	5.76	7.16	5.68	7.99	6.25		
5	2.33	34.3				6.28	3.41		7.54	9.12	6.40	31.7	7.08	7.65		
10	144	4.52				5.67	5.05		12.7	4.70	11.3	10.8	7.68	5.86		

Site

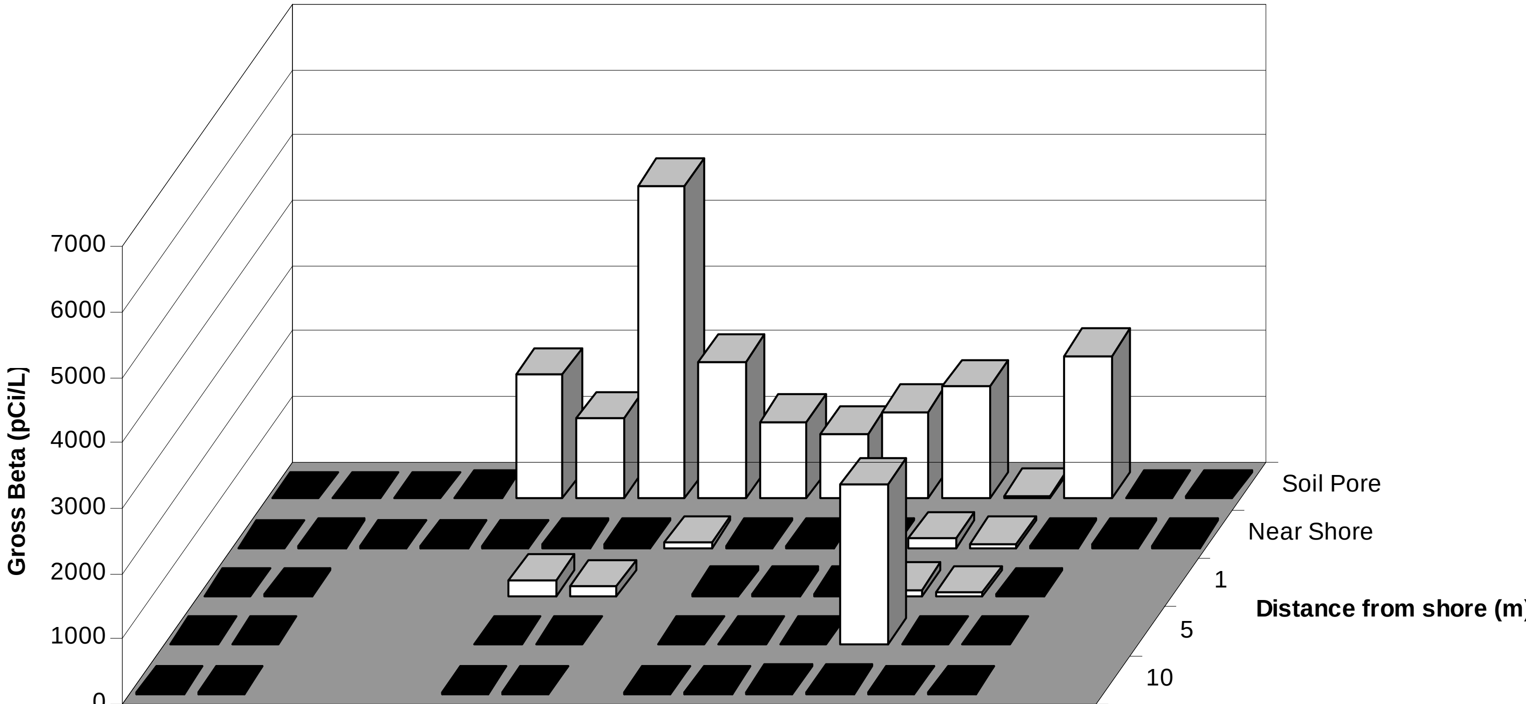
Figure 27. Spatial distribution of dissolved copper (μg/L) in water (February 2000). MW = Moab Wash. Sites upstream and downstream of Moab Wash are represented with a "U" and “D”, respectively. Numbers represent multiples of 50 m (e.g., D4 = 200 m downstream). CHW = Courthouse Wash. E4 and E10 are described in text. UX = 400 m upstream of Moab Wash. Utah water quality criteria for copper is 12 μg/L based on water hardness of 100 mg/L.



	E4	E10	Hwy 191	CHW	UG	UX	U4	U2	MW	D2	D4	D6	D8	D10	D15	D20
Soil Pore	6.06	1.71	-8.21	10.5	1660	1140	2410	1420	649	898.5	652	1020	31.9	1250	8.22	16
Near Shore	4.48	5.35	16.1	3.25	7.94	47.1	22.25	38.4	29	22.5	38	74.6	60.4	25	30.2	12.7
1	2.05	15				232	159		21.8	30.9	37.8	73.9	49.5	23.2		
5	11.9	8.6				19.7	16.3		13.3	16.7	13.2	1200	14.2	18.4		
10	8.02	8.31				7.02	8.1		10.4	16	11.1	21.1	16.1	16.05		

Site

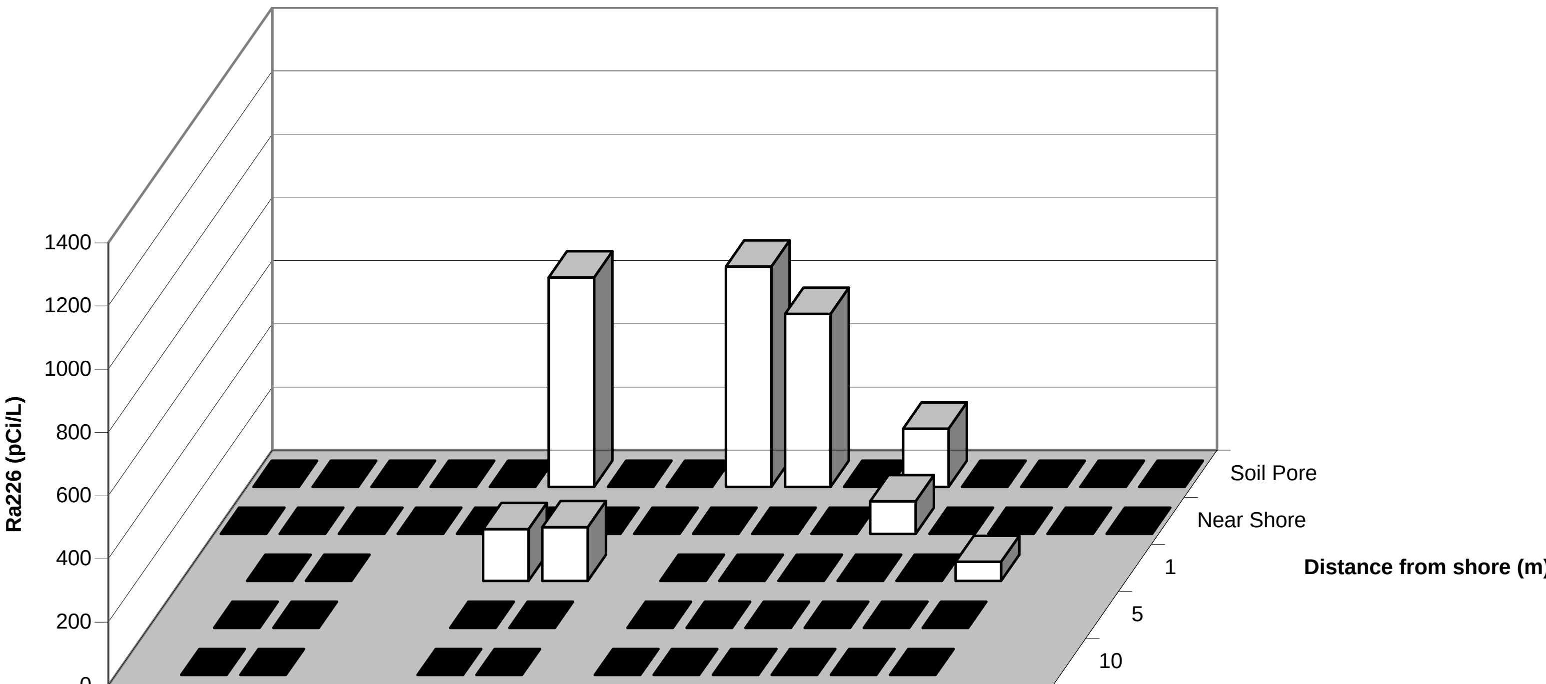
Figure 28. Spatial distribution of gross alpha (pCi/L) in water (February 2000). MW = Moab Wash. Sites upstream and downstream of Moab Wash are represented with a "U" and "D", respectively. Numbers represent multiples of 50 m (e.g., D4 = 200 m downstream). CHW = Courthouse Wash. E4 and E10 are described in text. UX = 400 m upstream of Moab Wash. Utah water quality criteria for gross alpha is 15 pCi/L.



	E4	E10	Hwy 191	CHW	UG	UX	U4	U2	MW	D2	D4	D6	D8	D10	D15	D20
Soil Pore	7.99	6.23	-2.5	24.3	1900	1230	4800	2100	1190	997.5	1340	1720	56.2	2190	20.9	37.1
Near Shore	5.99	23.2	13.5	8.69	12.7	412	40.25	76	28	32.6	43.5	143	67.6	35.5	38.1	24
1	5.89	24.9				243	140		34.2	36	44.2	94.5	55.5	31		
5	9.99	15.1				17.7	21.1		18	16.6	19.4	2450	20.6	16.6		
10	15.4	9.56				15.95	11.3		14.1	15.6	26.4	31.6	15.3	15.9		

Site

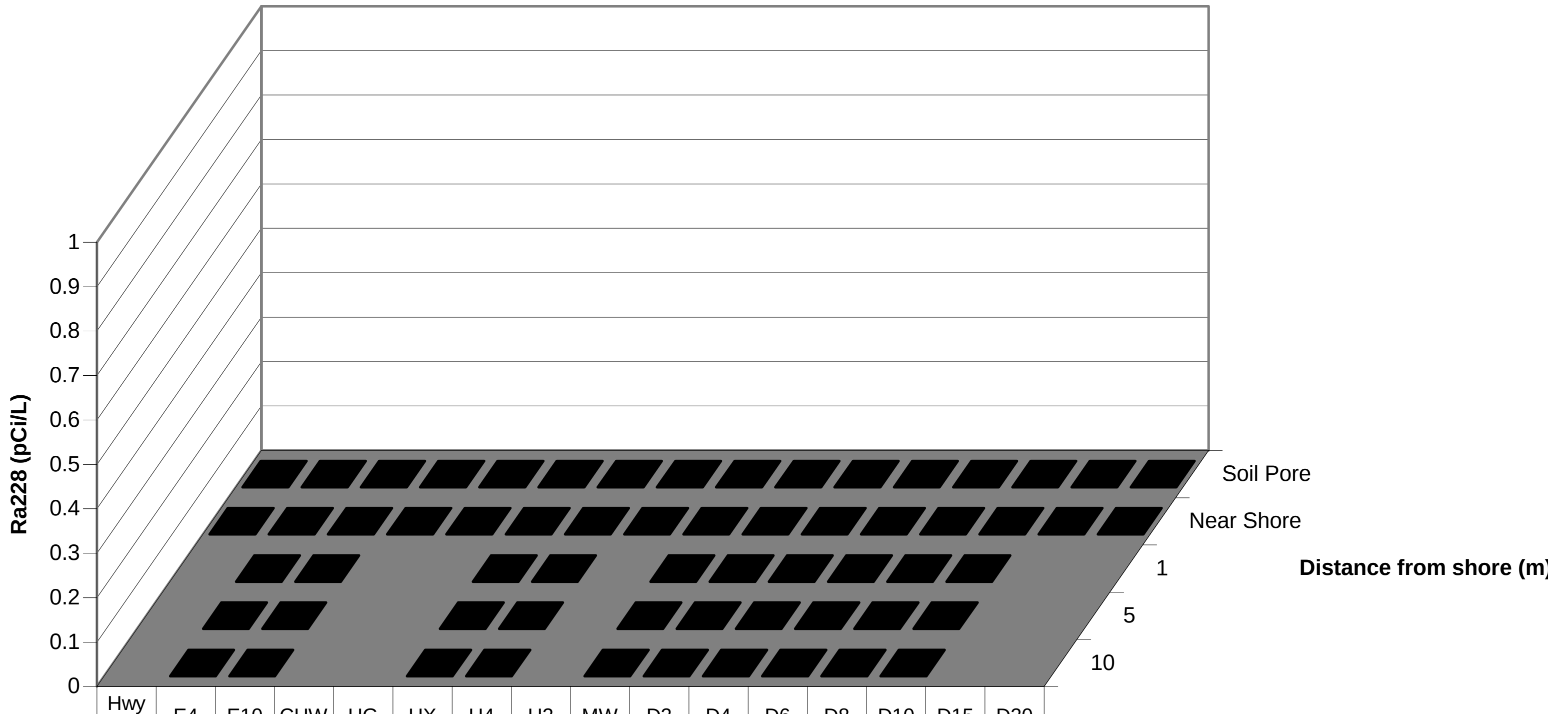
Figure 29. Spatial distribution of gross beta (pCi/L) in water (February 2000). MW = Moab Wash. Sites upstream and downstream of Moab Wash are represented with a "U" and “D”, respectively. Numbers represent multiples of 50 m (e.g., D4 = 200 m downstream). CHW = Courthouse Wash. E4 and E10 are described in text. UX = 400 m upstream of Moab Wash. Utah water quality criteria for gross beta is 50 pCi/L.



	Hwy 191	E4	E10	CHW	UG	UX	U4	U2	MW	D2	D4	D6	D8	D10	D15	D20
Soil Pore	0	0	0	0	0	662	0	0	697	547	0	184	0	0	0	0
Near Shore	0	0	0	0	0	0	0	0	0	0	0	103	0	0	0	0
1		0	0			164	170		0	0	0	0	0	60.3		
5		0	0			0	0		0	0	0	0	0	0		
10		0	0			0	0		0	0	0	0	0	0		

Site

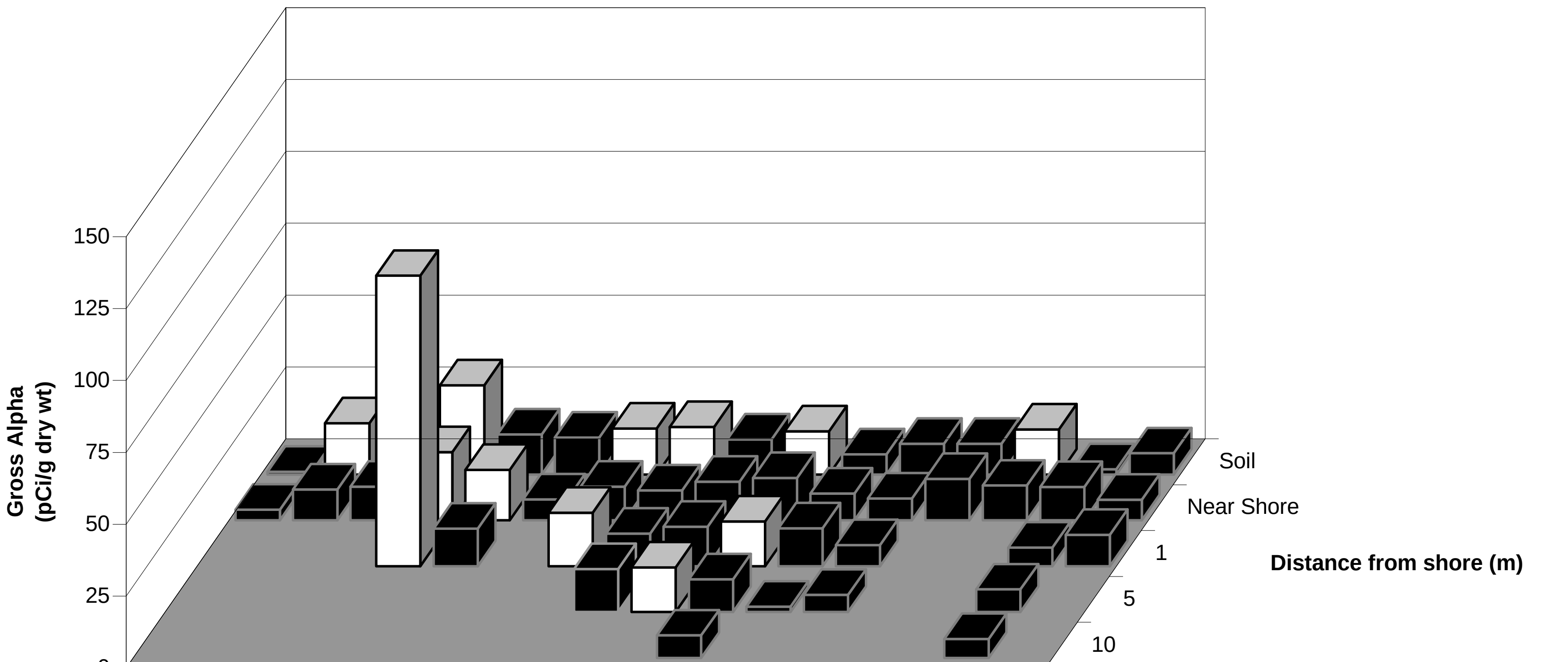
Figure 30. Spatial distribution of Ra226 (pCi/L) in water (February 2000). MW = Moab Wash. Sites upstream and downstream of Moab Wash are represented with a "U" and “D”, respectively. Numbers represent multiples of 50 m (e.g., D4 = 200 m downstream). CHW = Courthouse Wash. E4 and E10 are described in text. UX = 400 m upstream of Moab Wash.



	Hwy 191	E4	E10	CHW	UG	UX	U4	U2	MW	D2	D4	D6	D8	D10	D15	D20
Soil Pore	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Near Shore	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
1		0	0			0	0		0	0	0	0	0	0		
5		0	0			0	0		0	0	0	0	0	0		
10		0	0			0	0		0	0	0	0	0	0		

Site

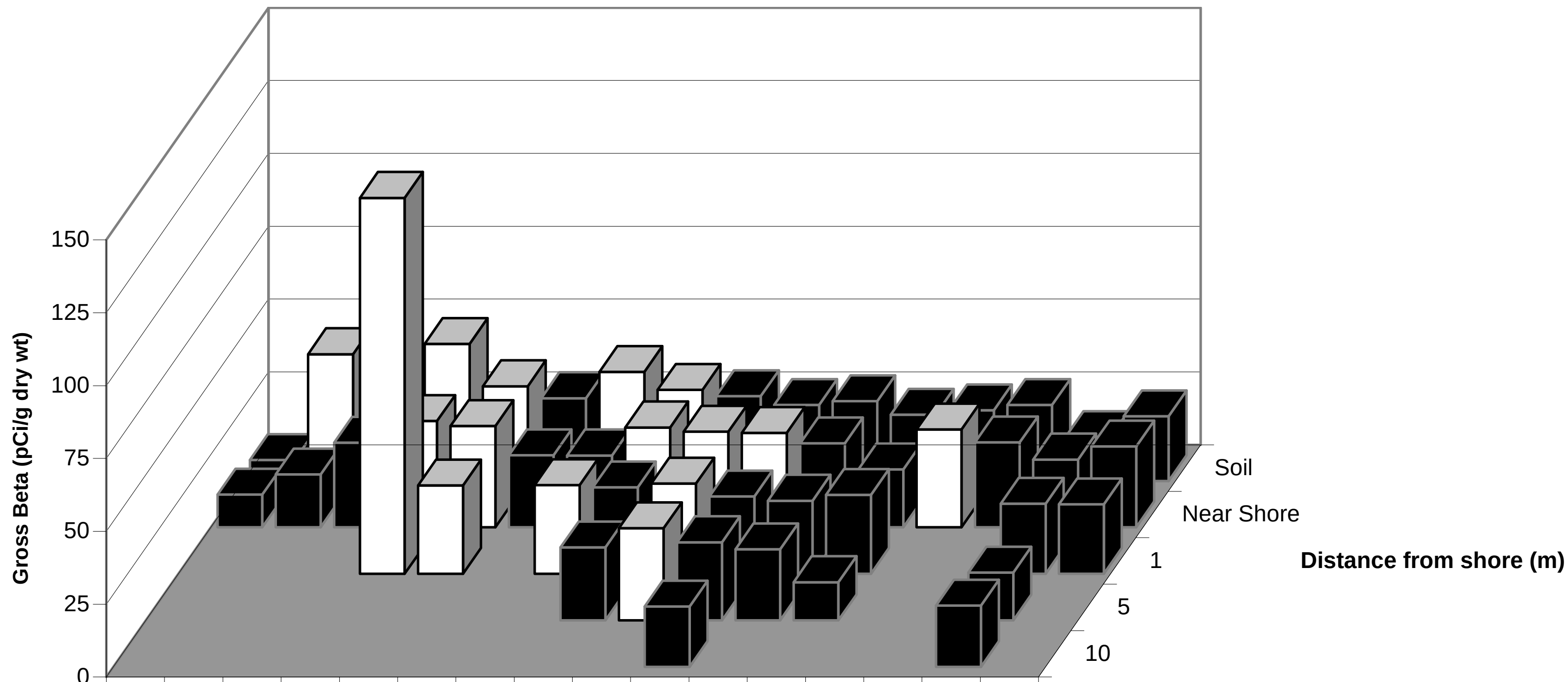
Figure 31. Spatial distribution of Ra228 (pCi/L) in water (February 2000). MW = Moab Wash. Sites upstream and downstream of Moab Wash are represented with a "U" and "D", respectively. Numbers represent multiples of 50 m (e.g., D4 = 200 m downstream). CHW = Courthouse Wash. E4 and E10 are described in text. UX = 400 m upstream of Moab Wash.



	E4	E10	Hwy 191	CHW	UG	UX	U4	U2	MW	D2	D4	D6	D8	D10	D15	D20
Soil	0.975	17.8	5.9	31	13.9	12.9	16	16.5	12.2	15	6.99	10.7	10.7	15.7	1.85	7.4
Near Shore	3.69	10.75	11.7	23.7	17.6	7.28	11.7	10.4	13.45	14.7	9.3	7.63	14.4	12.1	11.6	7.16
1				101	13.1		18.6	11.3	13.7	15.6	13.2	7.32			6.47	10.9
5								14.95	15.5	11.4	1.86	5.98			7.87	
10										7.81					6.57	

Site

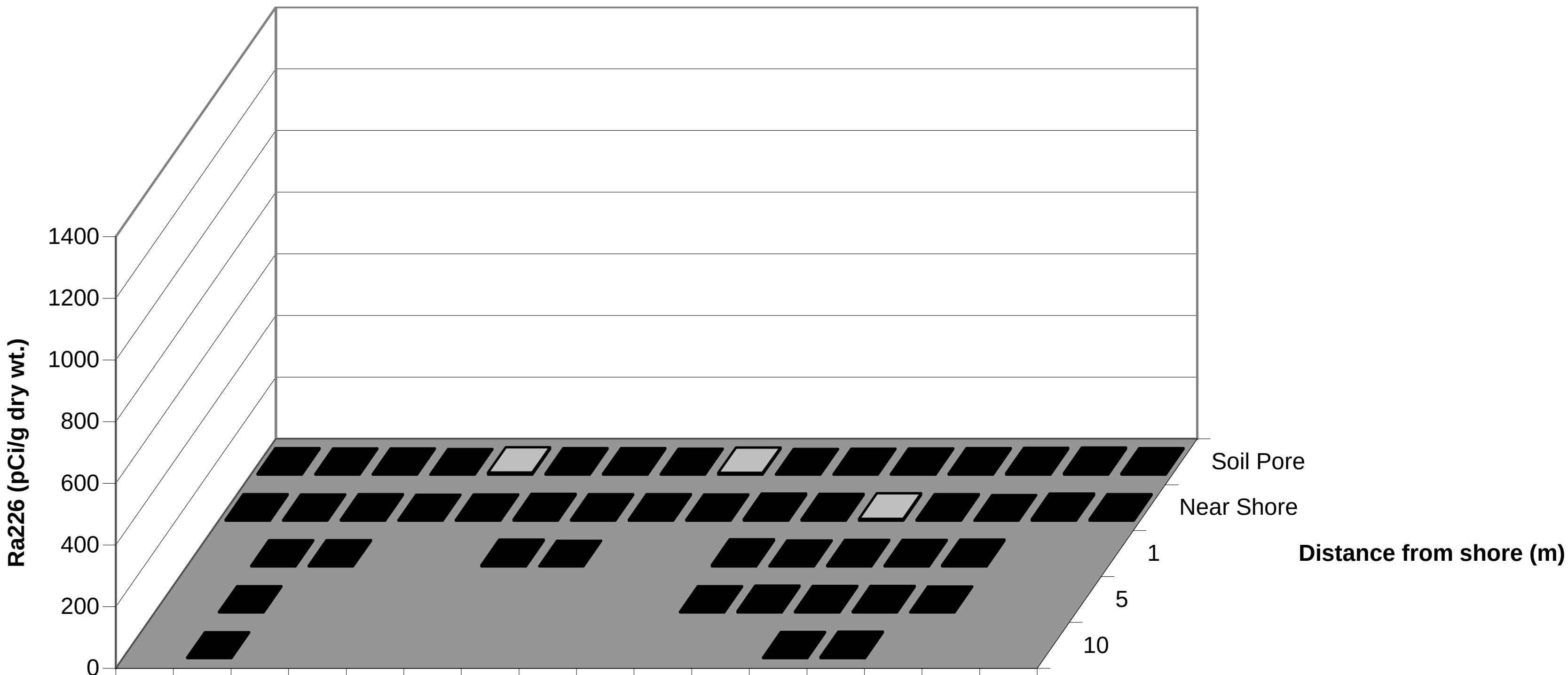
Figure 32. Spatial distribution of gross alpha (pCi/g dry wt.) in sediment (February 2000). MW = Moab Wash. Sites upstream and downstream of Moab Wash are represented with a "U" and "D", respectively. Numbers represent multiples of 50 m (e.g., D4 = 200 m downstream). CHW = Courthouse Wash. E4 and E10 are described in text. UX = 400 m upstream of Moab Wash. Utah water quality criteria for gross alpha is 15 pCi/g dry wt..



	E4	E10	Hwy 191	CHW	UG	UX	U4	U2	MW	D2	D4	D6	D8	D10	D15	D20
Soil	7.23	43.5	26.8	47	32.5	28.4	37.4	31.3	29.1	26.1	27.4	22.7	24.2	26.1	15.1	22.2
Near Shore	11.3	18.17	29.1	36.5	34.8	24.8	24.7	34.3	32.8	32.4	28.9	19.9	33.6	29.2	23.3	27.8
1				129	30.3		30.4	29.7	31	26.6	25.1	27.1			24.1	23.9
5								25	31.6	26.8	24.4	13.1			16.4	
10										20.7					21	

Site

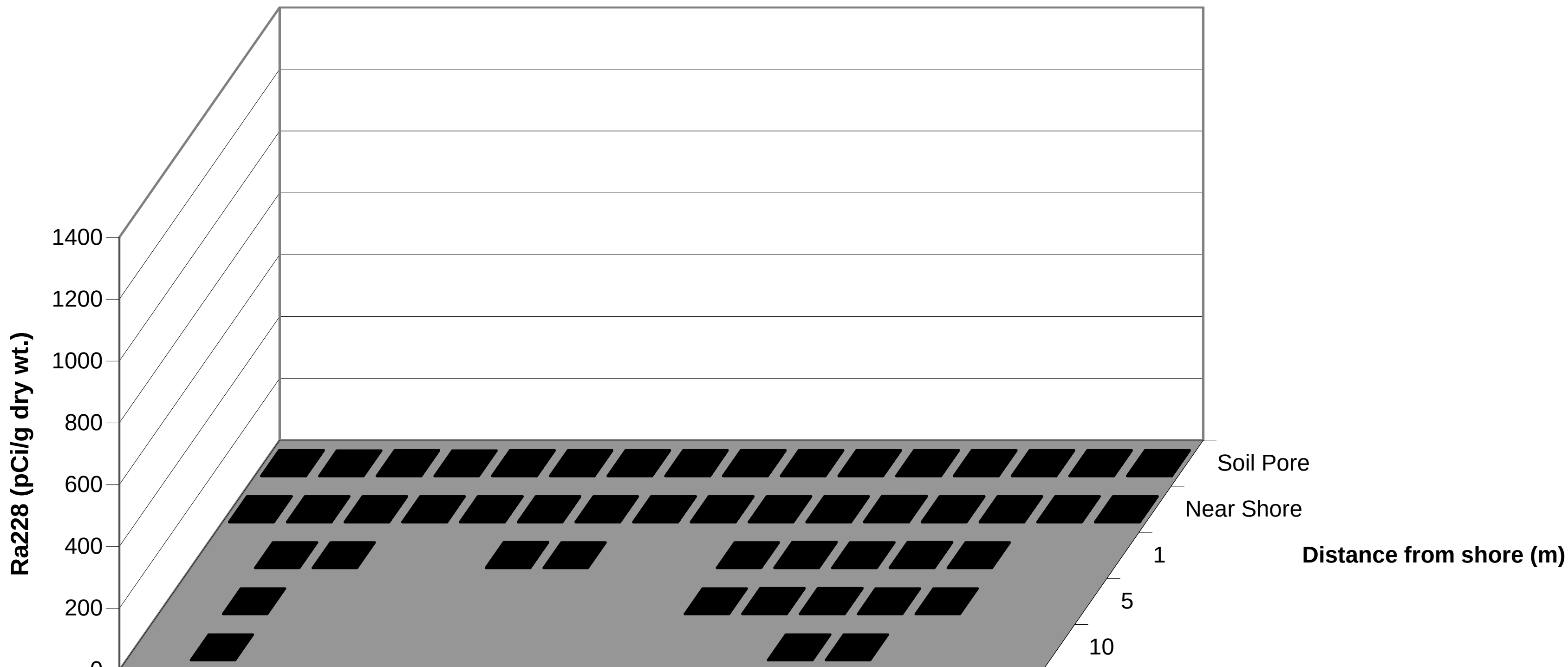
Figure 33. Spatial distribution of gross beta (pCi/g dry wt.) in sediment (February 2000). MW = Moab Wash and sites upstream and downstream of Moab Wash are represented with a "U" and “D”, respectively. Numbers represent multiples of 50 m (e.g., D4 = 200 m downstream). CHW = Courthouse Wash. E4 and E10 are described in text. UX = 400 m upstream of Moab Wash. Utah water quality criteria for gross beta is 50 pCi/g dry wt.



	Hwy 191	E4	E10	CHW	UG	UX	U4	U2	MW	D2	D4	D6	D8	D10	D15	D20
Soil Pore	3.09	2.20	2.10	0.24	6.52	3.11	2.86	1.73	5.65	0.61	1.15	1.95	3.19	4.49	4.76	3.55
Near Shore	2.41	2.00	2.27	0.59	1.40	3.38	2.62	2.62	2.19	4.35	3.90	5.62	2.64	0.00	4.47	3.03
1		2.13	2.07			4.04	0.00			4.82	1.49	3.08	3.28	4.45		
5		1.50								1.75	3.76	2.97	2.62	0.54		
10		1.61										2.00	3.61			

Site

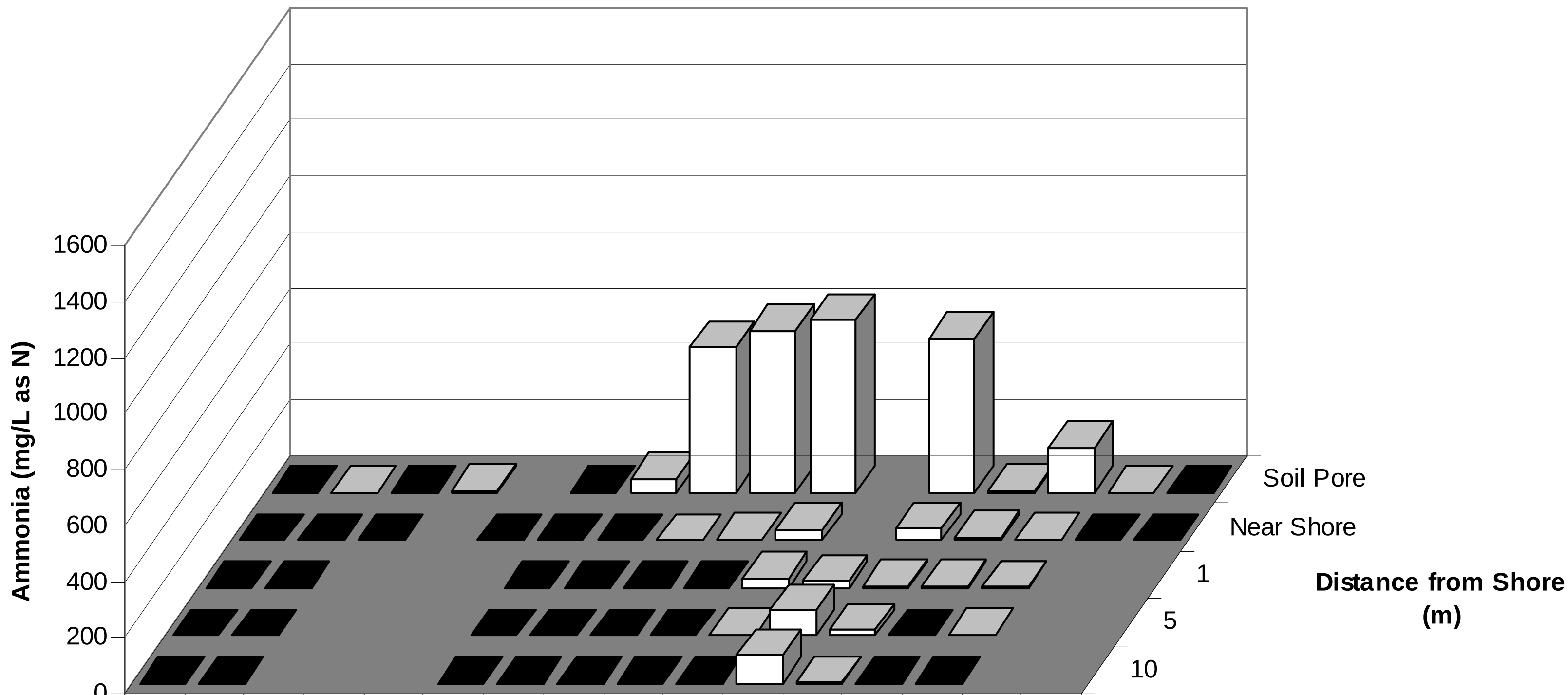
Figure 34. Spatial distribution of Ra226 (pCi/g dry wt.) in sediment (February 2000). MW = Moab Wash. Sites upstream and downstream of Moab Wash are represented with a "U" and “D”, respectively. Numbers represent multiples of 50 m (e.g., D4 = 200 m downstream). CHW = Courthouse Wash. E4 and E10 are described in text. UX = 400 m upstream of Moab Wash.



	Hwy 191	E4	E10	CHW	UG	UX	U4	U2	MW	D2	D4	D6	D8	D10	D15	D20
Soil Pore	1.16	0.29	0.40	0.10	1.16	0.91	0.85	0.94	1.17	1.16	0.91	0.74	0.55	0.42	0.39	0.95
Near Shore	0.96	0.71	0.79	0.20	0.97	0.96	0.94	1.02	0.52	1.01	1.11	1.22	0.58	0.44	0.93	0.61
1		0.64	0.82			1.03	0.83			0.95	1.13	0.60	1.02	0.81		
5		0.36								0.39	1.18	0.81	0.32	0.36		
10		0.30										0.39	0			

Site

Figure 35. Spatial distribution of Ra228 (pCi/g dry wt.) in sediment (February 2000). MW = Moab Wash. Sites upstream and downstream of Moab Wash are represented with a "U" and “D”, respectively. Numbers represent multiples of 50 m (e.g., D4 = 200 m downstream). CHW = Courthouse Wash. E4 and E10 are described in text. UX = 400 m upstream of Moab Wash.



	E4	E10	HWY 191	CHW	UG	UX	U4	MW	D2	D4	D5	D6	D8	D10	D15	D20
Soil Pore	0.91	2.25	0.30	7.48		0.39	49.3	519	577	617		549	6.66	162	101	0.21
Near Shore	0.10	0.12	0.11		0.13	0.08	0.33	139	3.08	39.1		43.3	9.06	2.86	0.70	0.53
1	0.11	0.08				0.09	0.15	0.70	0.99	35.1	24.8	6.65	7.98	2.59		
5	0.05	0.06				0.07	0.06	0.17	0.54	120	88.6	20.9	0.38	186		
10	0.05	0.05				0.05	0.06	0.10	0.25	0.39	99.7	3.21	0.18	0.41		

Site

Figure 36. Spatial distribution of total ammonia (mg/L) in water (August 2000). MW = Moab Wash. Sites upstream and downstream of Moab Wash are represented with a "U" and "D", respectively. Numbers represent multiples of 50 m (e.g., D4 = 200 m downstream). CHW = Courthouse Wash. E4 and E10 are described in text. UX = 400 m upstream of Moab Wash. Utah 30-d chronic ammonia criteria is 1.21 mg/L as N based on pH = 8.0 and temperature = 25 °C for class 3B rivers.

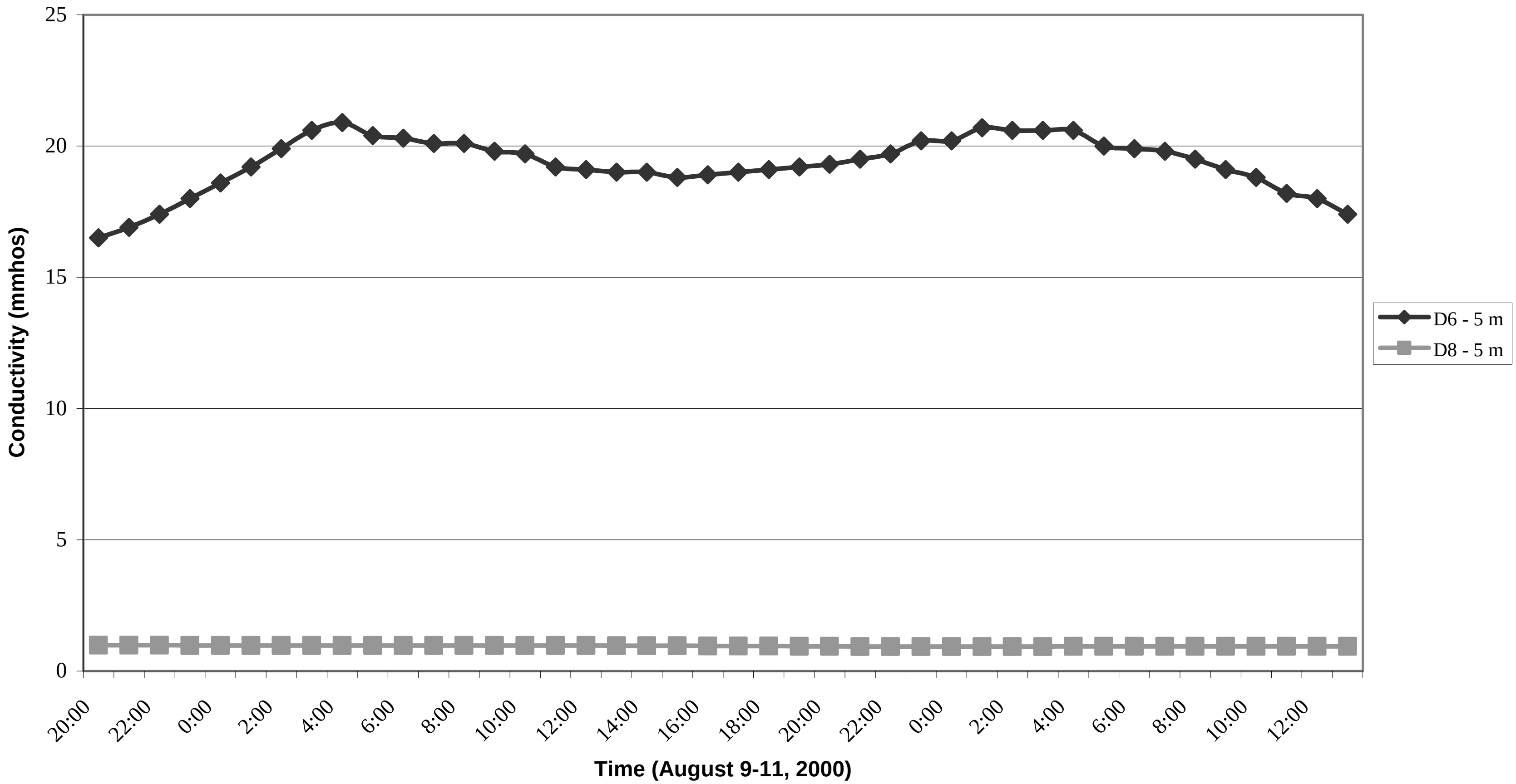


Figure 37. Diurnal changes in conductivity at two locations in the Upper Colorado River, August 2000. The first designation (e.g., D6) indicates site; the second designation indicates distance from shore in meters (e.g., 5 m). Measurements were taken hourly over a 32-h interval using an *in situ* Hydrolab Water Quality Monitor.

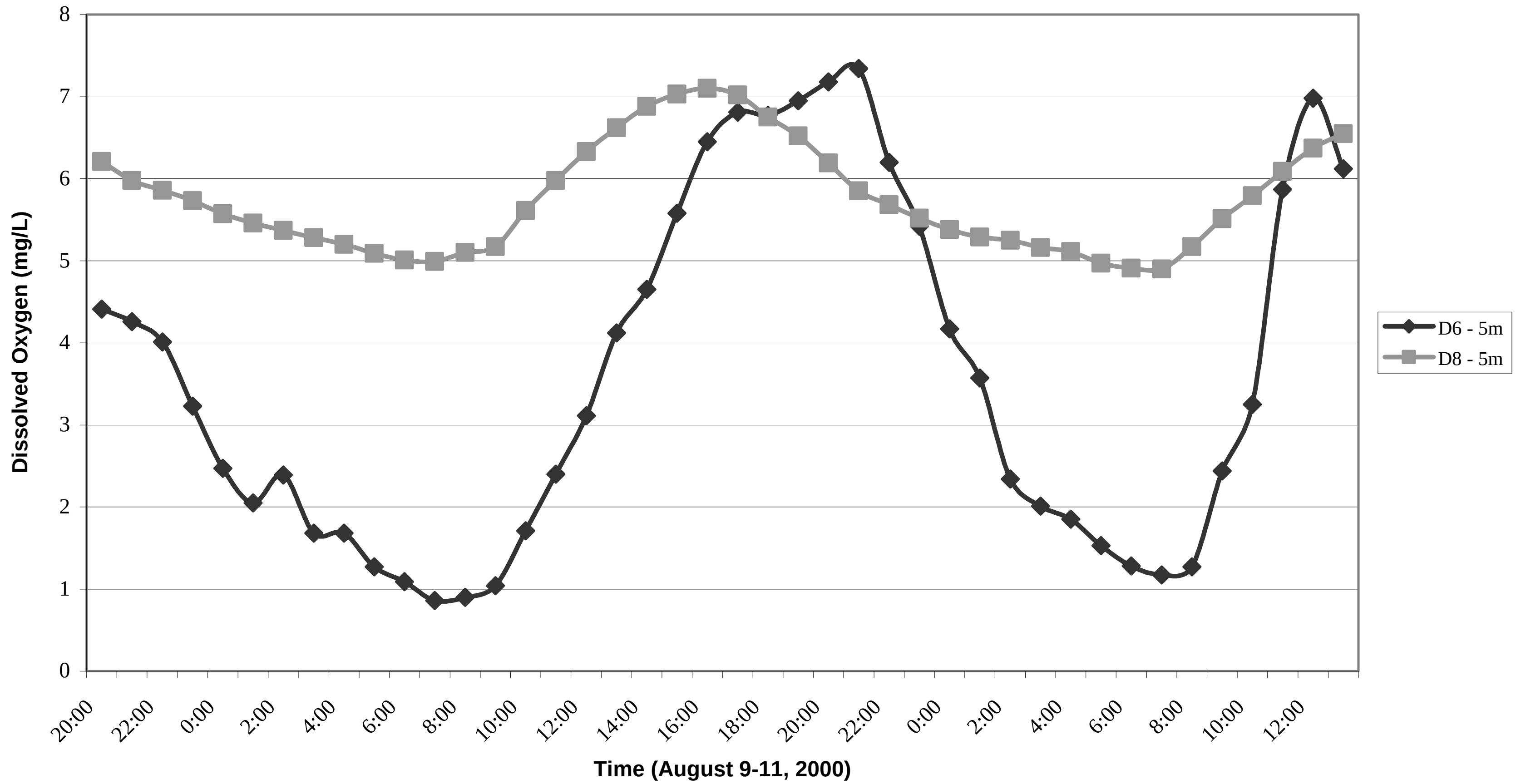


Figure 38. Diurnal changes in dissolved oxygen at two locations in the Upper Colorado River, August 2000. The first designation (e.g., D6) indicates site; the second designation indicates distance from shore in meters (e.g., 5 m). Measurements were taken hourly over a 32-h interval using an *in situ* Hydrolab Water Quality Monitor.

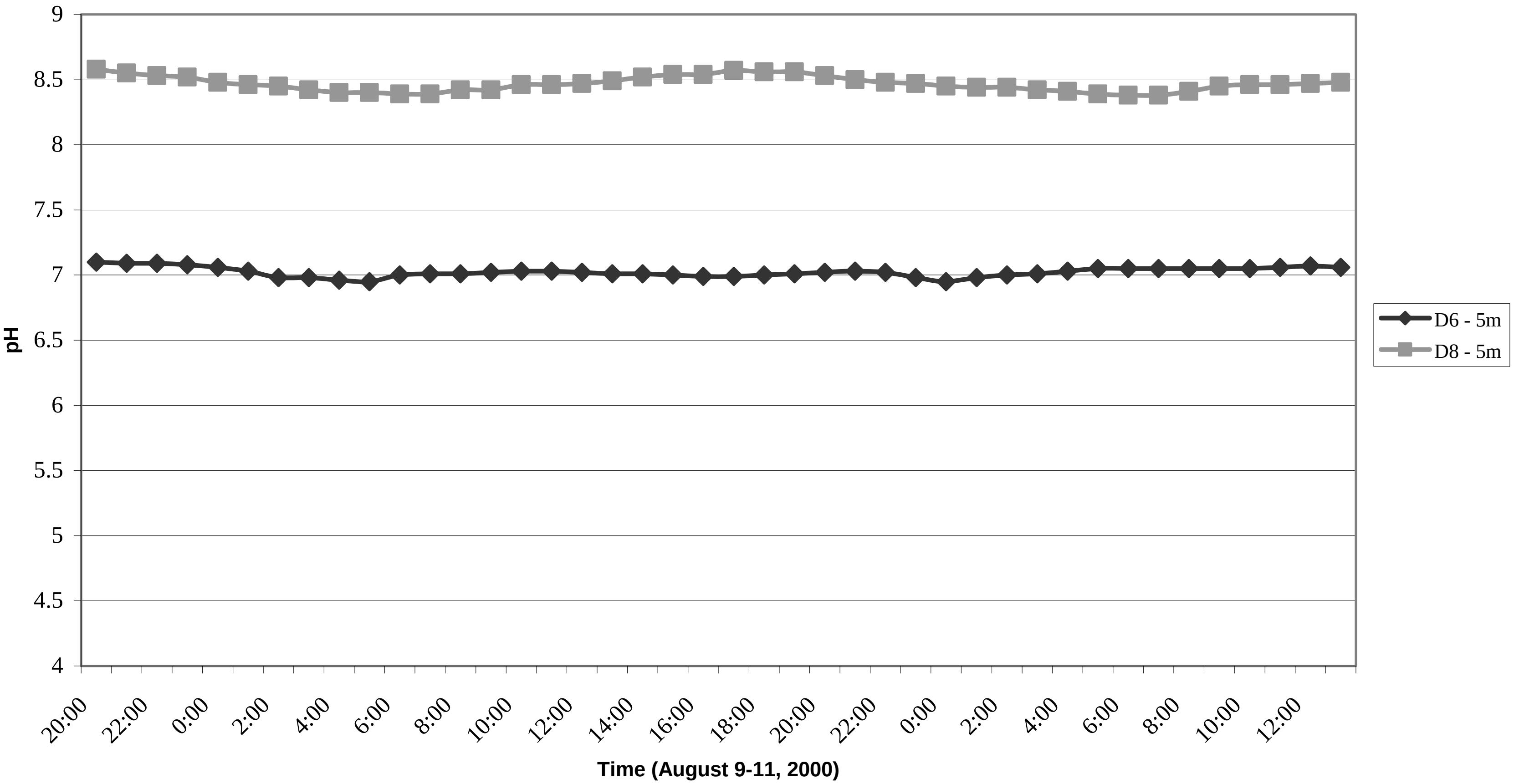


Figure 39. Diurnal changes in pH at two locations in the Upper Colorado River, August 2000. The first half of the legend (e.g., D6) indicates site; the second designation indicates distance from shore in meters (e.g., 5 m). Measurements were taken hourly over a 32-h interval using an *in situ* Hydrolab Water Quality Monitor.

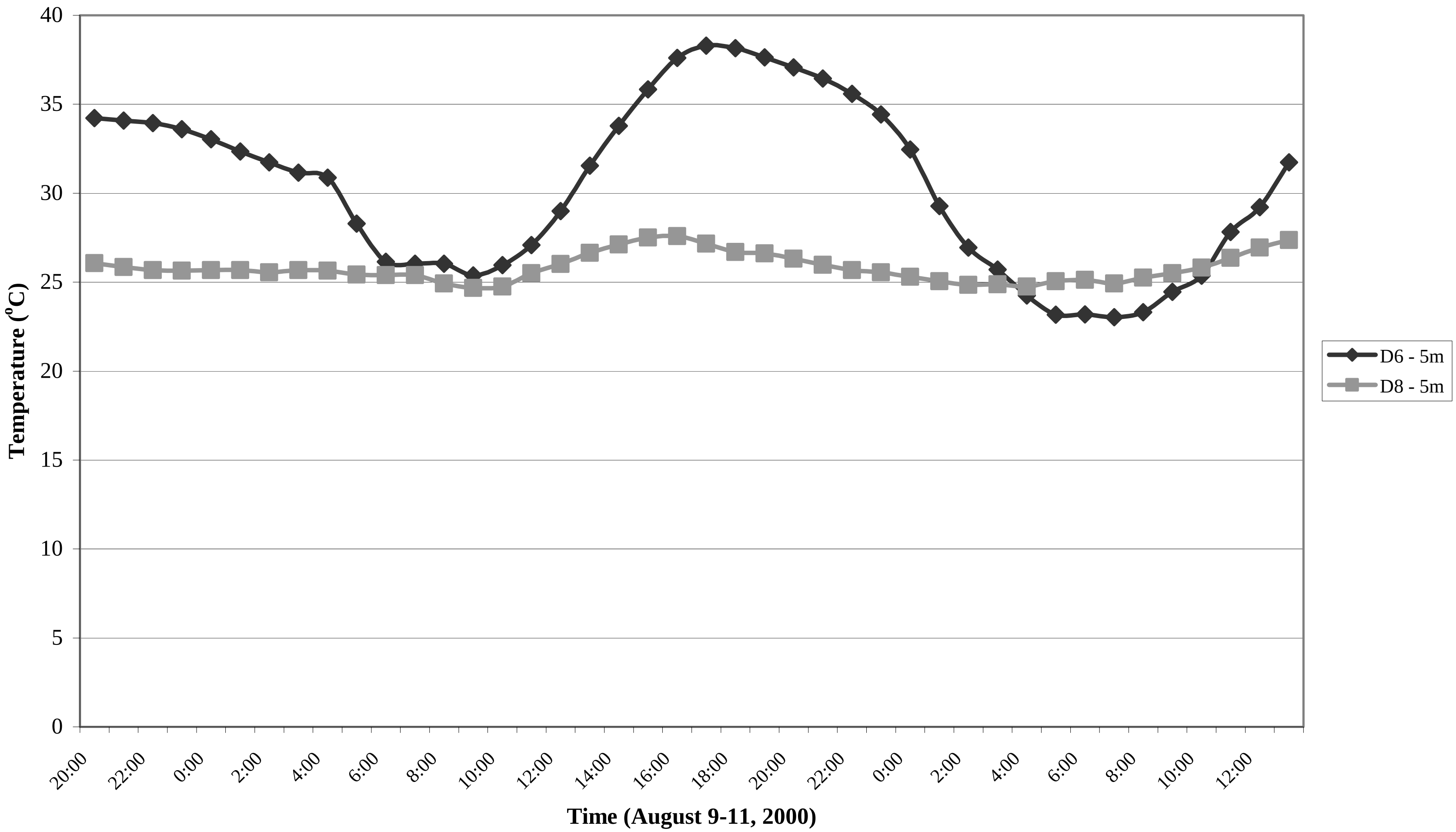


Figure 40. Diurnal changes in temperature at two locations in the Upper Colorado River, August 2000. The first designation (e.g., D6) indicates site; the second designation indicates distance from shore in meters (e.g., 5 m). Measurements were taken hourly over a 32-h interval using an in situ Hydrolab Water Quality Monitor.

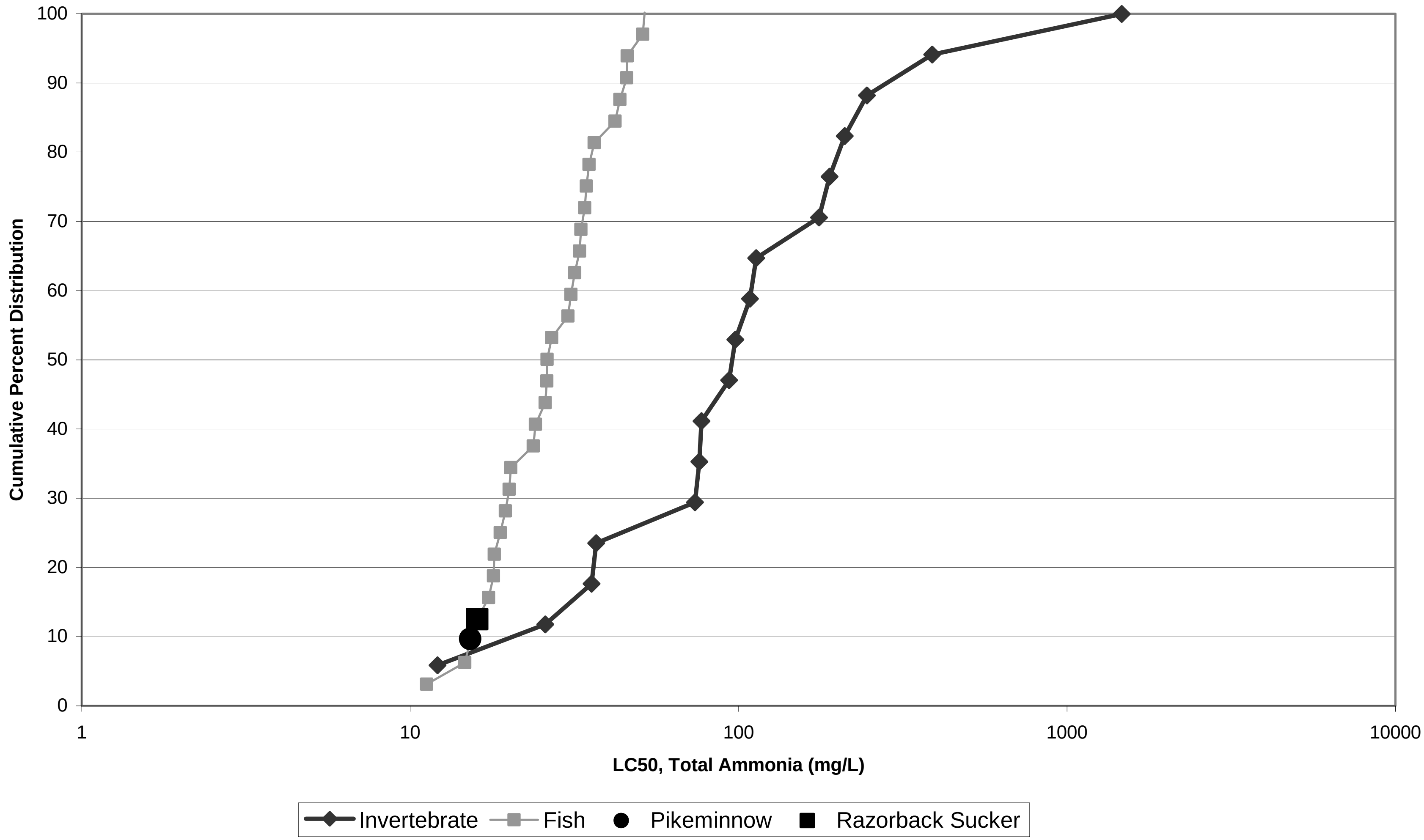


Figure 41. Cumulative species sensitivity profile for the acute toxicity of total ammonia. Toxicity values are LC50s adjusted to standard conditions (pH = 8.0; temperature = 25 ^{0}C). Data are from USEPA (1999); pikeminnow value from CERC July 1999 study; razorback sucker value from CERC April 1999 study.

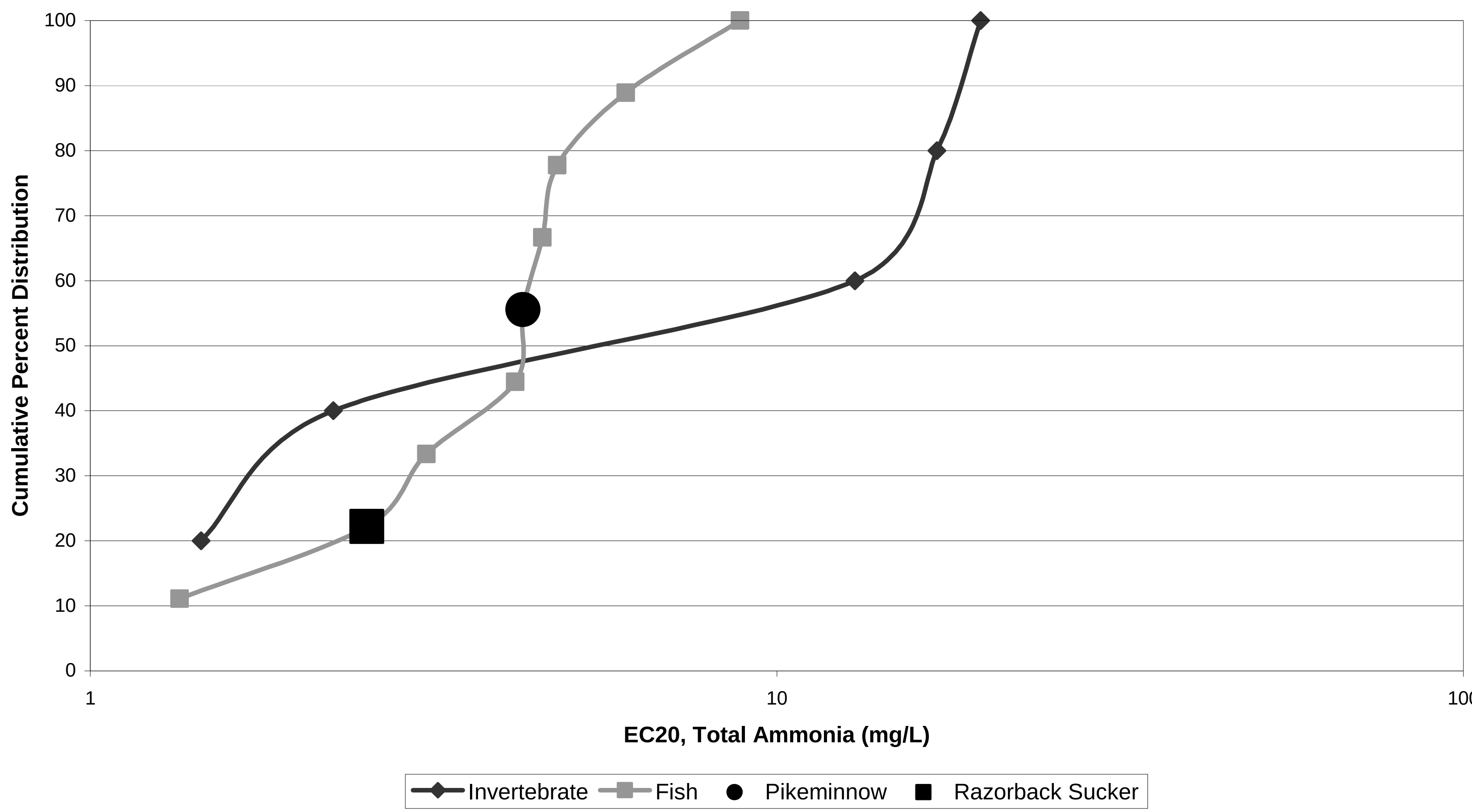


Figure 42. Cumulative species sensitivity profile for chronic toxicity of total ammonia. Toxicity values are EC20s adjusted to standard conditions (pH = 8.0; temperature = 25 ^{0}C). Data are from USEPA (1999); pikeminnow data from CERC July 1999 study. Razorback sucker data from CERC April 1999 study.

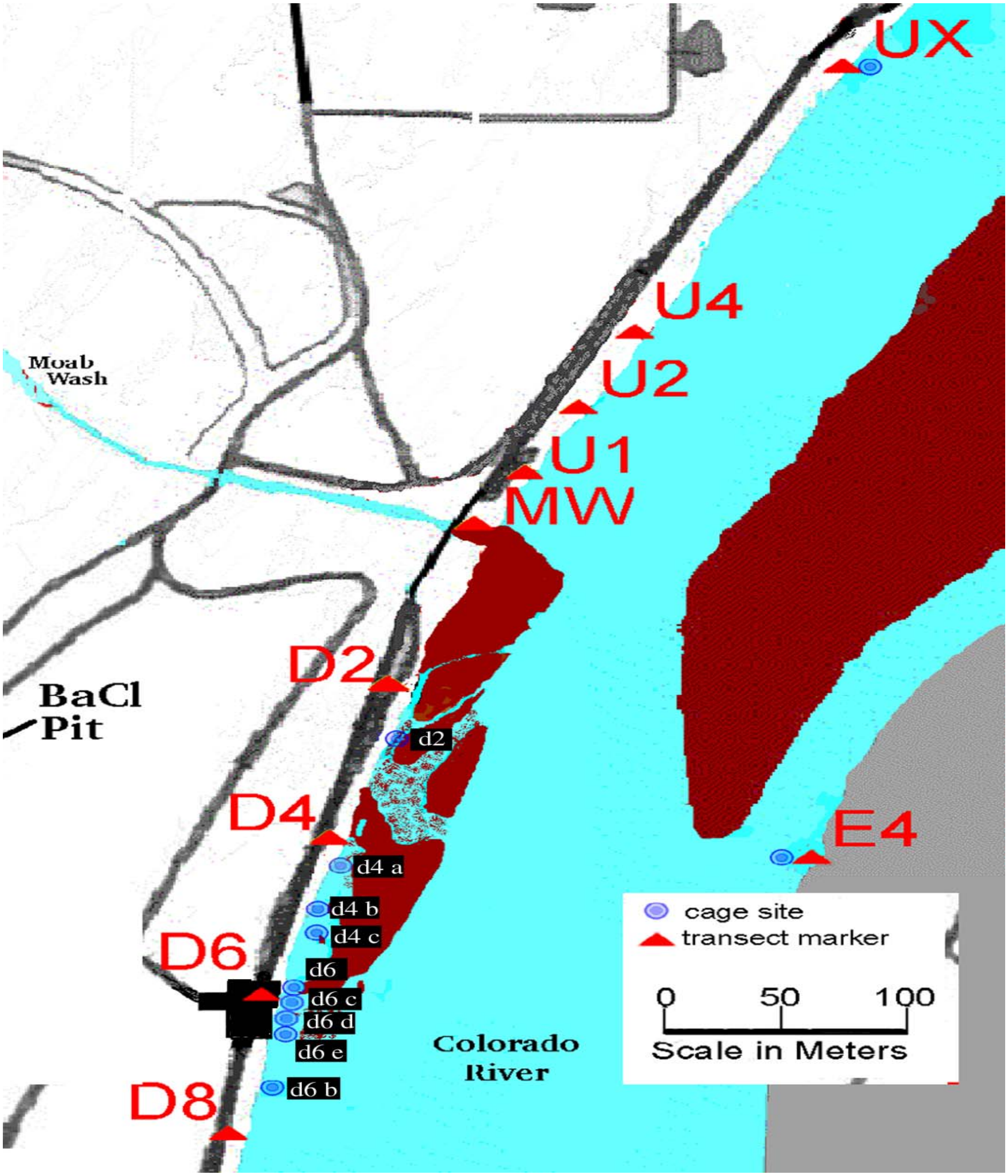
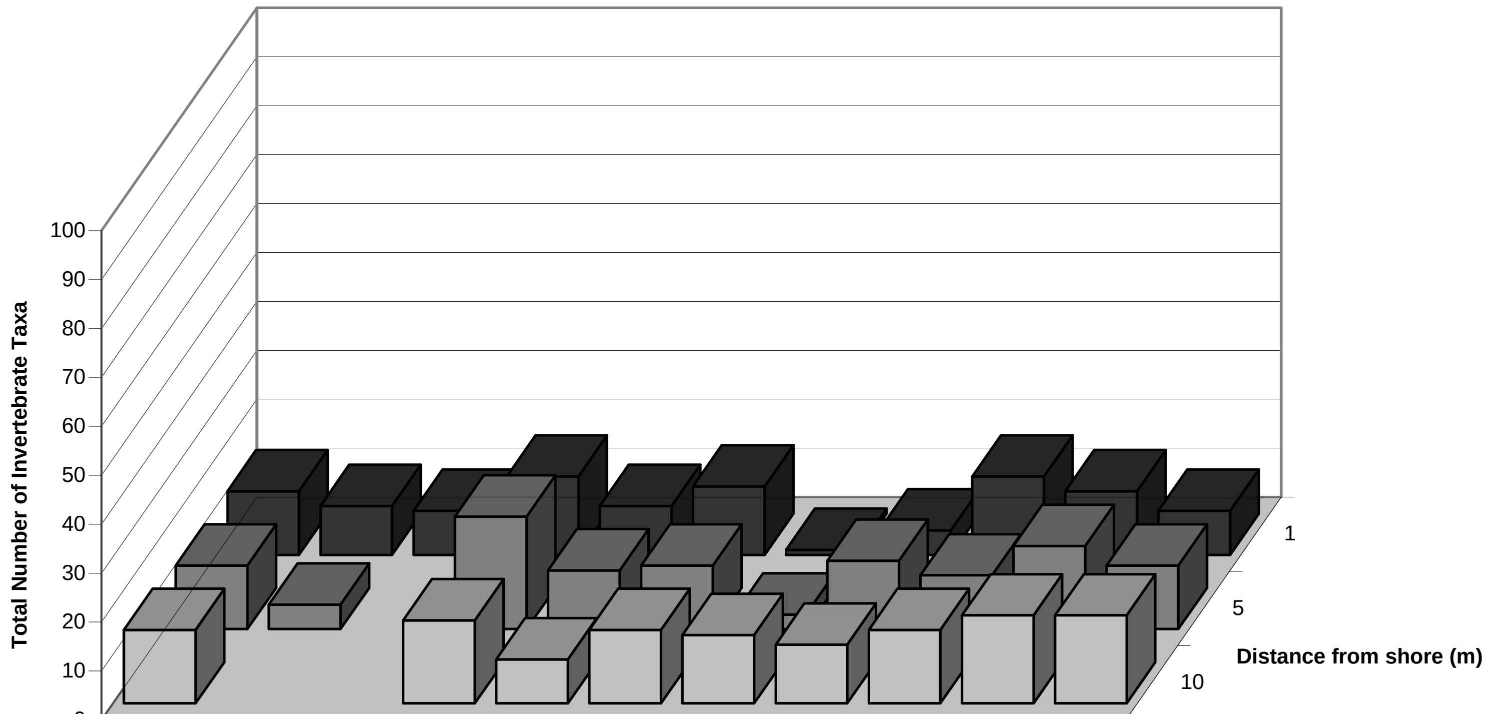
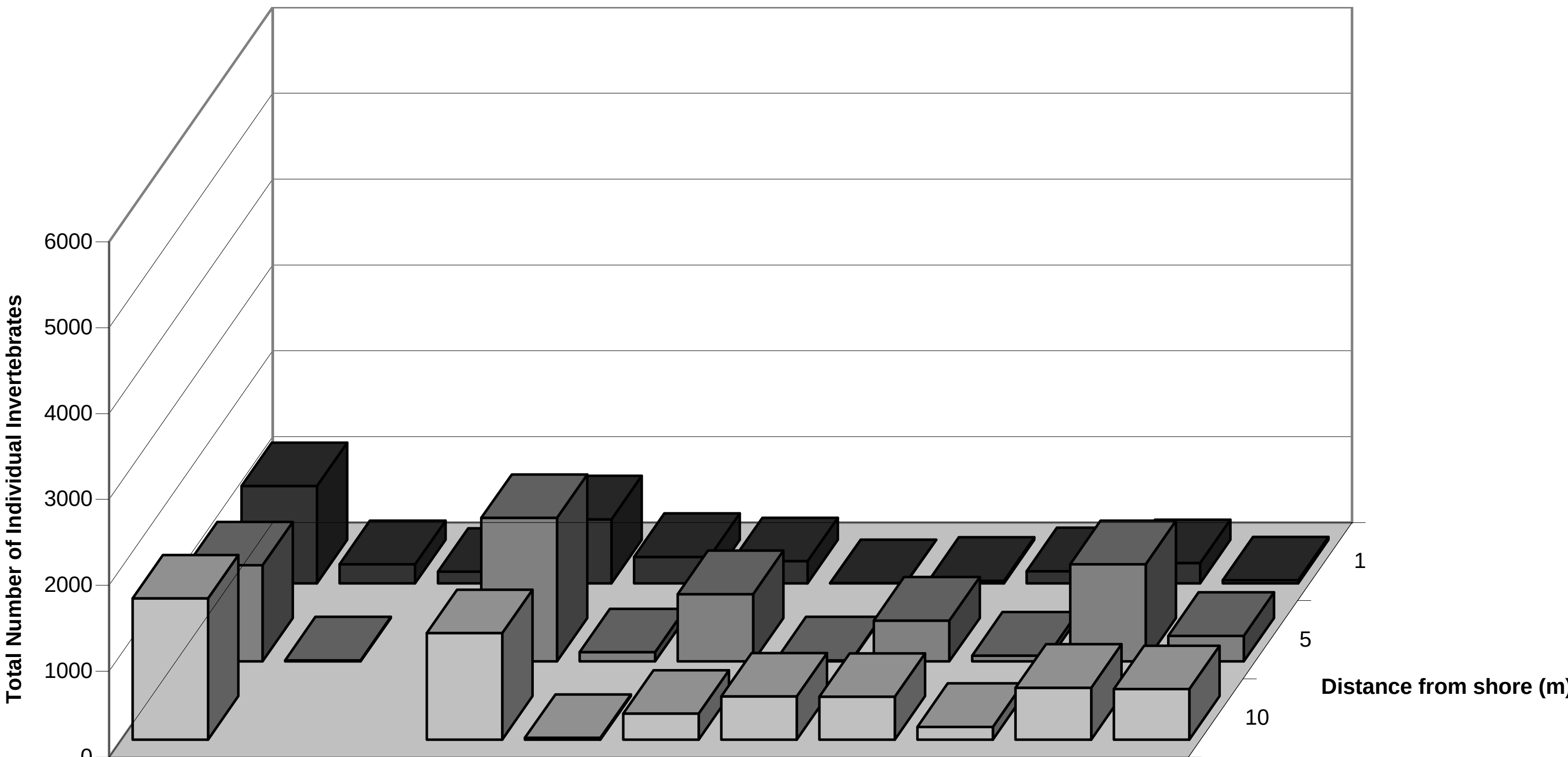


Figure 43. Spatial map of February 2000 *in situ* bioassay locations. Small circles denote locations of fish cages. Sites UX and E4 represent reference locations.



	E4	E10	CHW	UX	U4	MW	D2	D4	D6	D8	D10
1	13	10	9	16	10	14	1	5	16	13	9
5	13	5		23	12	13	3	14	11	17	13
10	15			17	9	15	14	12	15	18	18

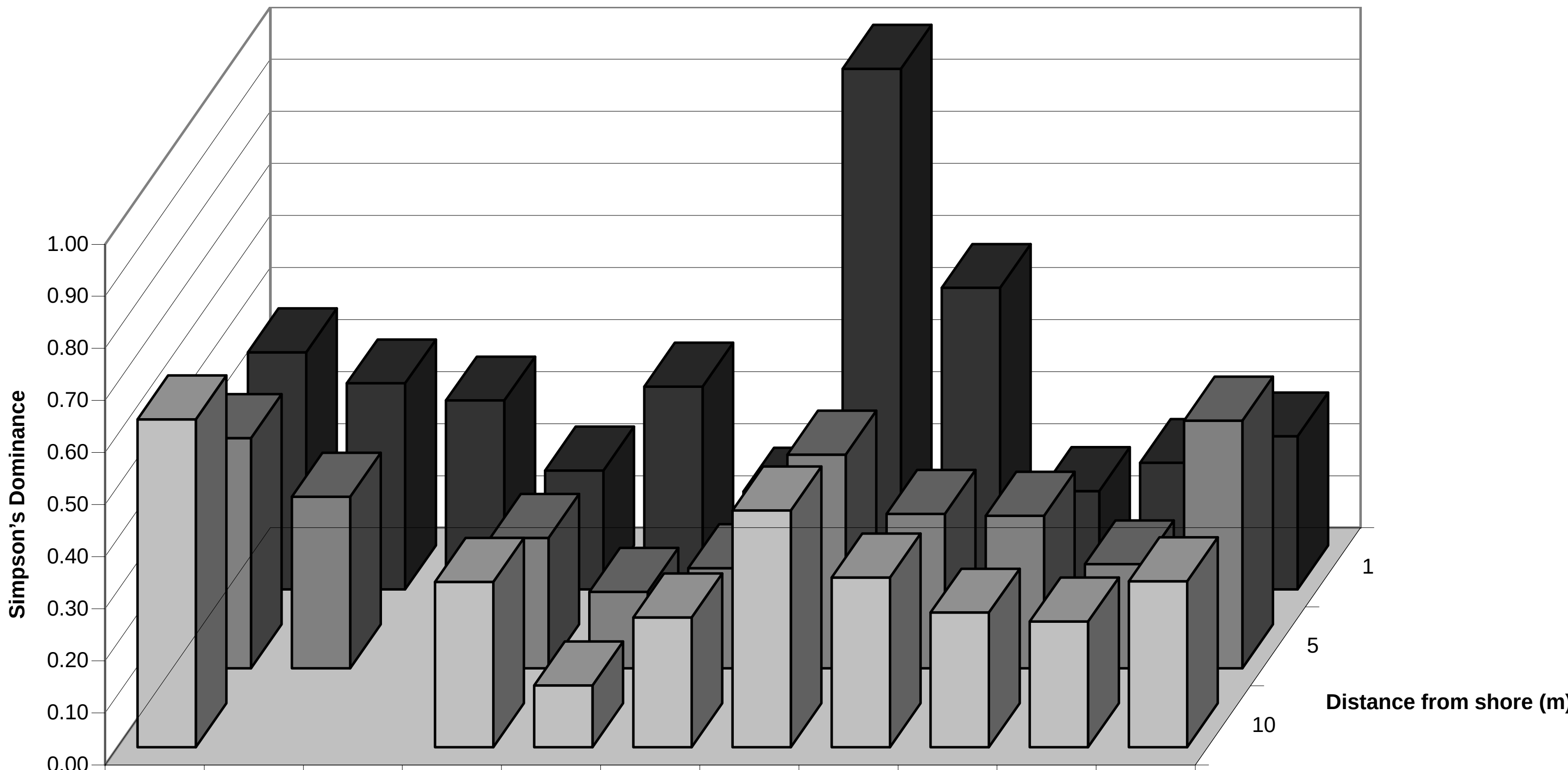
Figure 44. Spatial distribution of total number of invertebrate taxa (February 1999). MW = Moab Wash. Sites upstream and downstream of Moab Wash are represented with a "U" and “D”, respectively. Numbers represent multiples of 50 m (e.g., D4 = 200 m downstream). CHW = Courthouse Wash. E4 and E10 are described in text. UX = 400 m upstream of Moab Wash.



	E4	E10	CHW	UX	U4	MW	D2	D4	D6	D8	D10
1	1132	223	136	745	307	258	2	29	141	235	36
5	1121	14		1672	107	784	13	477	69	1133	298
10	1645			1241	21	302	504	499	149	605	589

Site

Figure 45. Spatial distribution of total number of individual invertebrates (February 1999). MW = Moab Wash. Sites upstream and downstream of Moab Wash are represented with a "U" and “D”, respectively. Numbers represent multiples of 50 m (e.g., D4 = 200 m downstream). CHW = Courthouse Wash. E4 and E10 are described in text. UX = 400 m upstream of Moab Wash.



	E4	E10	CHW	UX	U4	MW	D2	D4	D6	D8	D10
1	0.46	0.40	0.36	0.23	0.39	0.19	1	0.58	0.19	0.24	0.29
5	0.44	0.33		0.25	0.15	0.19	0.41	0.30	0.29	0.20	0.48
10	0.63			0.32	0.12	0.25	0.45	0.33	0.26	0.24	0.32

Site

Figure 46. Spatial distribution of Simpson's Dominance. Index of invertebrates collected in February 1999. MW = Moab Wash. Sites upstream and downstream of Moab Wash are represented with a "U" and "D", respectively. Numbers represent multiples of 50 m (e.g., D4 = 200 m downstream). CHW = Courthouse Wash. E4 and E10 are described in text. UX = 400 m upstream of Moab Wash.