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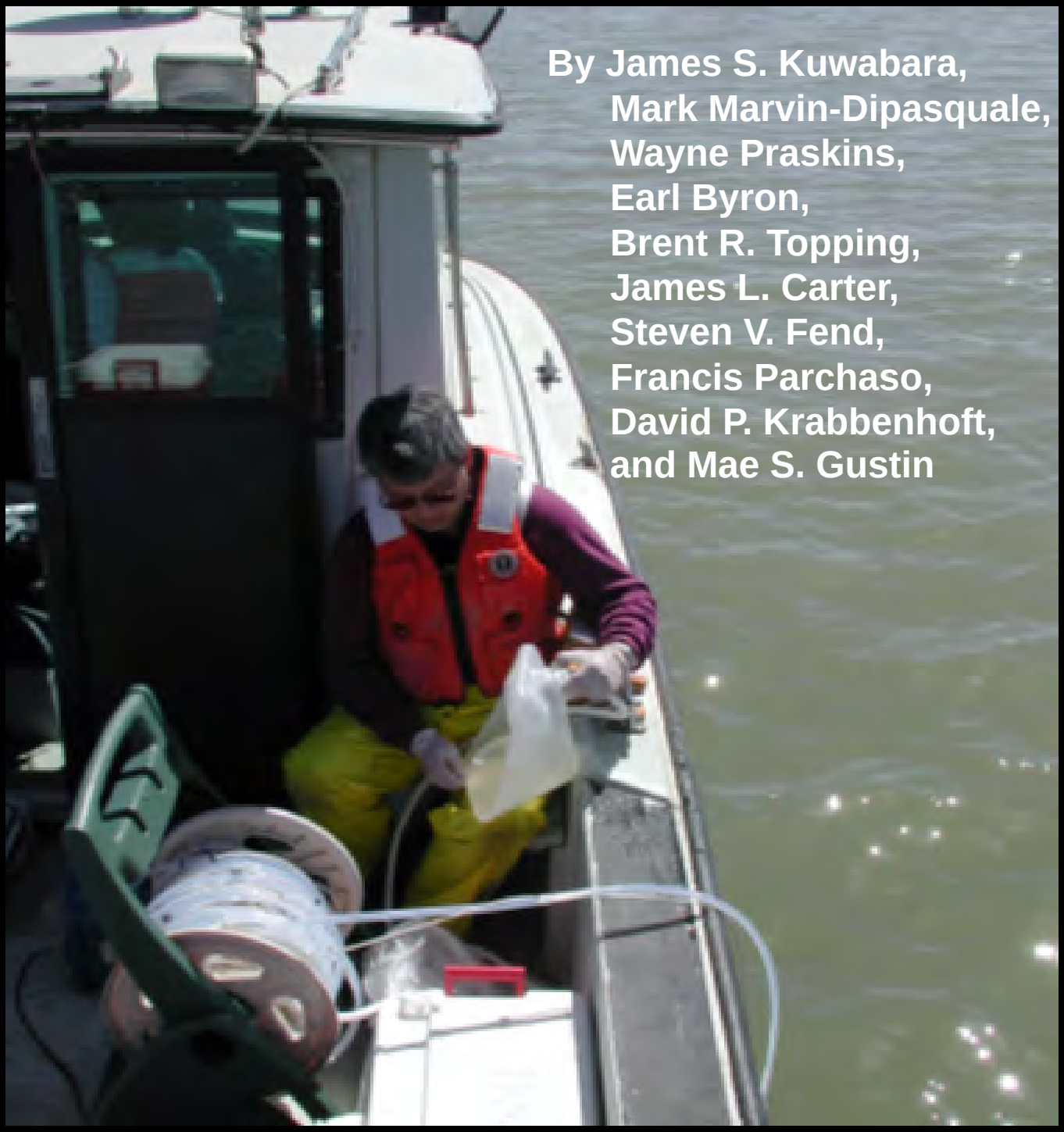

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U.S. Department of Interior
U.S. Geological Survey

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Executive Summary

Field and laboratory studies were conducted between April 30, 2001 and July 19, 2001 to provide the first direct measurements of the benthic flux of dissolved (0.2-micrometer filtered) mercury species (total and methylated forms) between the bottom sediment and water column at three sites: two in the southern lobe and one in the northern lobe of Lahontan Reservoir, Nevada (Background, Fig. 1). Dissolved-mercury species and predominant ligands (represented by dissolved organic carbon, and sulfides) were the solutes of primary interest. Benthic flux, sometimes referred to as internal recycling, represents the transport of dissolved chemical species between the water column and the underlying sediment.

Water-quality managers often assess and prioritize remediation strategies for aquatic systems, in particular Super Fund sites that have been adversely affected by anthropogenic activities. In the case of the Lahontan Reservoir along the Carson River, mercury associated with historic gold and silver extraction has been fluvially transported and accumulated in the bottom sediments. Frequent demands have been made by Super Fund site managers and the general public to quantify the connections between fluxes of contaminants and the health, abundance, and distribution of biological resources (Kuwabara and others, 1999). As part of a comprehensive examination of transport processes affecting mercury dynamics in Lahontan Reservoir, this study focuses on a poorly understood, yet potentially predominant, source of mercury to the reservoir water column, which is internal recycling, or benthic flux of mercury species and associated ligands. Mobilization, flux, and biological availability of mercury into the water column of the reservoir are affected by physical (e.g., advection and diffusion), chemical (i.e., oxidation-reduction reactions, complexation and repartitioning) and biological processes (Flegal and others, 1991; Kuwabara and others, 1996; Grenz and others, 2000, Topping and others, 2001).

The results described herein followed from the integration of current project studies with information needs identified by the U.S. Environmental Protection Agency, Region 9 (USEPA) to provide initial determinations of dissolved total and methyl-mercury fluxes from the sediments into the water column of Lahontan Reservoir. Recent mercury distribution and transformation studies in the Carson River system by Marvin-Dipasquale and others (2001) indicated the potential importance of sediment-water interactions in describing mercury speciation, its sources and sinks. Quantifying and understanding the magnitude and variability of these interactions are critical to the accurate assessment of contaminant sources and loads as well as to the development of process-integrated water-quality models for this mining-affected system.

With a variety of strategies under consideration to determine how to most efficiently improve the water quality in the Carson River system, the primary question posed in this study was, "What processes regulate the fate and transport of mercury species in Lahontan Reservoir? In particular, are sources and sinks of dissolved total and methyl mercury associated with the bottom sediment within Lahontan Reservoir significant relative to major surface-water inputs from the Carson River?" The question was motivated by a number of factors. First, extraction of precious metals up gradient of the reservoir provide a historic source of elemental mercury that continues to be fluvially transported in dissolved and particulate phases (Hoffman and Taylor, 1998; Carroll and others, 2000; Carroll and Warwick, 2001). Elevated total and methyl mercury concentrations into the reservoir have been well documented (Priessler and others, 1999; Jones and others, 1999; Allander and others, 2001, Marvin-Dipasquale and others, 2001). Therefore, determining whether some fraction of this sediment-associated mercury can remobilize for transport to the overlying water and subsequently to down-stream portions of the Carson River is necessary.

Second, elevated concentrations of mercury species in reservoir water, sediment, and fish have prompted a compilation, and comparison of dominant contaminant sources so that appropriate remedial strategies can be designed and implemented. Third, changes in oxidizing or reducing (redox) conditions and nutrient availability near the sediment-water interface (e.g., during phytoplankton blooms) can dramatically alter the mobility of metals and ligands associated with the bottom sediment as episodic sources of carbon settle out and accumulate (Thompson and others, 1981). Finally, there is a growing body of evidence from other aquatic systems that benthic flux or internal recycling of contaminants and nutrients is an important process to consider in developing appropriate ecosystem water-quality models (Wood and others, 1995; Kuwabara and others, 2000). The need for more refined numerical and conceptual models for mercury dynamics within the Carson River system thus clearly exists.

This report is formatted unconventionally in a pyramid-like structure to effectively serve the needs of diverse parties interested in reviewing or acquiring information at various levels of detail (Appendix 1). The report enables quick transitions between the initial summary information (figuratively at the top of the pyramid) and later details of methods or results (that is, figuratively towards the base) using hyperlinks to supporting figures and tables, and an electronically linked Table of Contents.

During two sampling events, two replicate sediment cores were collected from three reservoir locations (Fig. 1) for incubation experiments to provide flux estimates and benthic biological characterizations (Fig. 2). Ancillary data, including nutrient and ligand fluxes, were determined to provide a water-quality framework from which to compare the results for mercury. Because the first sampling event spanned the last day of April and the first days of May 2001, the results for that experimental series will be referred to as "April/May 2001". The following major observations from interdependent physical, biological, and chemical data were made:

Physical and Biological Characterizations

1. **Hydrologic Conditions**: The sampling associated with this study occurred during one of the driest years on record for the area as depicted by the discrepancy between the daily average streamflow into the reservoir and the historical average daily streamflow since the beginning of the construction of the Lahontan Dam in 1911 (Fig. 3; U.S. Geological Survey, 2001). These drought conditions caused a rapid draw down of the reservoir as indicated by reservoir-storage data between the two coring trips. The time of the second coring trip was greatly influenced by restricted boat-ramp access. For example, the water-column depths at the coring sites in the southern and northern lobes on April 30, 2001 were approximately 6.7 and 21.3 meters, respectively, while on July 18, 2001, during our second coring trip, those depths had decreased to 4.2 and 18.9 meters, respectively.

   Dissolved-oxygen and temperature profiles during the coring trips indicate that an increase in water-column temperatures between the trips coincides with a decrease in bottom-water dissolved-oxygen concentration (Fig. 4). However, the water column remained oxic through the summer, and in fact subsequent water-column sampling indicated that the bottom water did not become anoxic that year. This may partially be explained in the northern lobe by vertical mixing of water from the extraction depth of Lahontan Dam and the augmented input of oxygenated water from the Truckee Canal near the dam to meet agricultural demands down gradient along the Carson River (Fig. 1).

2. **Porosity**: The surficial sediments at the three coring sites were observably different during both sampling trips. Cores from main-channel, southern-lobe Site 1, consistently exhibited the lowest porosities (0.65 to 0.70). Less than 0.5 kilometers away from Site 1, the near-shore site in the southern lobe (Site 2) consistently indicated higher porosities (0.80 to 0.85). The highest porosities at the northern-lobe site 3 (0.89 to 0.93) were indicative of the fine-grained unconsolidated, viscous slurry at the sediment-water interface (Table 1).

3. **Macrobenthos**: The presence of certain macroinvertebrate taxa may increase, by orders of magnitude, the solute flux across the sediment-water interface due to biologically enhanced advection

(Caffrey and others, 1996; Kuwabara and others, 1999). Macroinvertebrate abundances from the Lahontan sites appear to be similar to published abundance values from around the world (Fig. 5). However, macroinvertebrate populations in the Lahontan samples were considered to be of low importance to benthic flux because they were dominated by very small immature tubificid worms and nematodes that typically pass through a conventional 500-micrometer mesh sieve (Tables 2 - 4; Macrobenthos discussion). Worm fragments, associated with particle aggregates, inflate the data depicted for Lahontan. The low biomass of the benthic community at the coring sites suggest that there were insufficient biota to generate biologically enhanced advection through bioirrigation or bioturbation. This is consistent with low benthic- chlorophyll concentrations (see section below), and with sulfide-flux determinations. However, other lines of evidence such as bromide injections, radon emanation or silica fluxes would be required to confirm that diffusion controls benthic flux.

4. **Benthic chlorophyll**: Chlorophyll-*a* concentrations in surficial sediments ranged from 0.8 to 2.1 micrograms per square centimeter with highest concentrations observed at the near-shore Site 2 during the spring sampling (April/May 2001), and the lowest concentrations at the northern-lobe site (Site 3) on both sampling dates (Table 5). A compilation of benthic biological data from our study and compilations from publications on 14 reservoirs and oligotrophic and mesotrophic lakes around the world (Fig. 5; Bass, 1992; Popp and Hoagland, 1995; Kuwabara and others, 2000) clearly indicated that the benthic algal biomass was sparse at all three coring sites relative to previously reported values.

Chemical Characterizations

Note: The dissolved-mercury concentrations discussed in this section refer to samples filtered with 0.7-micrometer quartz-fiber filters pre-combusted at 500 ^{o}C.

1. **Water-column concentrations**: Both dissolved total and methyl mercury concentrations in the bottom water over the two coring dates were lowest at the northern-lobe Site 3 (0.3 to 0.5 picomolar methyl mercury and 373 to 787 picomolar total mercury) and highest at the main-channel southern-lobe Site 1 (0.7 to 1.1 picomolar methyl mercury and 647 to 1952 picomolar total mercury). Total-mercury concentrations in reservoir bottom waters were higher on July 18, 2001 during low-flow conditions than during high-flow (April 30, 2001) by a factor of 2 to 3, but such temporal differences in methyl-mercury concentrations were not evident.

   Dissolved total and methyl mercury of riverine sources were 135 $\pm$ 21 picomolar (n=56) and 5.4 $\pm$ 1.2 picomolar (n=56), respectively (Priessler and others, 1999; Jones and others, 1999; Allander and others, 2001). Compared to the reservoir bottom waters (Table 7), the total-mercury concentrations of the riverine inputs were lower and the methyl-mercury riverine concentrations were higher.

2. **Benthic flux of dissolved forms of mercury**: Dissolved methyl-mercury fluxes were positive except from two cores at the main-channel southern-lobe Site 1. The highest fluxes were at the northern-lobe Site 3, and the lowest at the main-channel southern-lobe Site 1. The methyl-mercury fluxes were about three orders of magnitude less than the flux estimates for total mercury, that were roughly consistent with concentration differences between species.

   In contrast to methyl-mercury fluxes, dissolved total-mercury fluxes, based on end-point concentrations, were greatest at the main-channel southern-lobe Site 1. Total-mercury fluxes at the other two sites (near-shore southern-lobe Site 2 and northern-lobe Site 3) were lower in July than in April/May 2001.

Initial riverine, atmospheric and benthic flux estimates do not indicate a dominant transport process regulating the fate of mercury species in the reservoir water column (Fig. 7). But, at times, the release of dissolved mercury from the benthos is comparable to, or greater than, the magnitude of riverine and atmospheric flux (Fig. 6).

Remedial Implications

Because the benthic flux of mercury species may represent a dominant transport process, it also suggests an important management implication. Remediation efforts in the Carson River have an objective of decreasing concentrations and loads to downgradient systems. In the reservoir, a subsequent decrease in dissolved-mercury concentrations in the water column would shift and increase the vertical concentration gradient near the sediment-water interface. Consequently, solute flux into the water column from the sediments may increase as an unintentioned result of upstream remediation efforts (Fig. 7). Therefore, when water-quality models are developed to evaluate, justify and establish remediation strategies in the basin, the importance of reservoir sediment as mercury sources and sinks should be considered.

Background

What is benthic flux and why should it be considered?

Many fundamental processes affect the transport of dissolved chemical species (e.g., nutrients, metals, or ligands) through and within reservoirs and other aquatic systems. A conceptual model of these processes (Fig. 8) illustrates some physically based processes that have been examined and carefully quantified for several years (e.g., advective transport and point source inputs; Fisher and others, 1979).

Conversely, there are flux terms in the conceptual model that have received little attention. A prime example is the benthic flux term where no direct measurements for dissolved mercury species have heretofore been available for the study area. Benthic flux (sometimes referred to as internal recycling) represents the transport of dissolved chemical species between the water column and the underlying sediment. Flux of solutes can be either positive (into the water column from the sediment or atmosphere) or negative (out of the water column into the sediment or atmosphere) and can vary over multiple temporal and spatial scales (Kuwabara and others, 2000; Topping and others, 2001).

As a result of physical, chemical, and biological changes in the vertical cross section of the sediment-water interface, geochemical gradients take on a variety of forms that have been previously reported (Kuwabara and others, 2000; Fig. 9). When the solute concentration above and below the sediment-water interface are equivalent; there is no discernable concentration gradient, and, consequently, no net transfer of that substance across the interface results in a condition of **no benthic flux**. In contrast, when solute concentrations in the water column are higher than those in the bottom sediment pore waters, a **negative** benthic flux results whereby the substance moves **into the sediment**. Dissolved oxygen is a typical example of such a solute where microbial respiration can create a sediment demand for oxygen. When concentrations in the water column are lower than those in sediment pore waters, the vertical concentration gradient can physically drive the **release** from the sediment of chemical species to the overlying water yielding a **positive benthic flux**. The remobilization or chemical transformation of sediment-associated trace metals or ligands may exemplify this condition.

When interdependent factors regulate the benthic flux of biologically reactive substances, the vertical gradient for one dissolved species may be dependent on the gradient of another chemical species. For example, an **attenutated release** may occur when solute concentrations increase below the sediment-water interface only when another solute is depleted. Dissolved iron often behaves in this manner when suboxic conditions reduce it from ferric to ferrous forms, increasing its solubility. Additionally, macroinvertebrates can biologically enhance the benthic flux by irrigating, or perturbing surficial sediment layers (**bioirrigation, bioturbation,** or biologically enhanced **advection**). Certain productive benthic communities can enhance benthic flux beyond diffusive-control by orders of magnitude (Kuwabara and others, 1999). In summary, vertical chemical gradients generated by a variety of interdependent biogeochemical processes can induce the movement of dissolved-mercury species across the sediment-water interface.

Scientists and water-quality managers are only beginning to appreciate the importance of benthic flux in many aquatic environments. Within the past decade or two, researchers have gradually realized that there are non-hydrologic processes (for example, benthic flux) that must be incorporated into water-quality models in order to generate physically meaningful information. Benthic-flux studies have lagged behind studies of other transport processes (Flegal and others, 1991; Kuwabara and others, 1996) because quantifying benthic flux is instrument and manpower intensive. Each flux estimate requires a concentration time-series analysis or vertical-profile analysis.

Results and Discussion

Physical Data

There were visible differences in sediment texture among the three sampling sites that directly affected the collection of cores for incubation. The main-channel site in the southern lobe (Site 1) was consistently the lowest in porosity (Table 1) and the most difficult to core. This was because the finer surficial sediment of approximately 2 to 3 centimeter depth overlay sandy, unconsolidated material that was sometimes lost during core retrieval. The near-shore site in the southern lobe (Site 2), although less than 0.5 kilometers away from Site 1, was consistently higher in porosity and easier to core because of a thicker (4 to 5 centimeter) layer of fine-grained surficial material that allowed deeper core penetration. The deepest site in the northern lobe (Site 3) yielded sediment of the highest porosity.

Biological Data

1. **Benthic chlorophyll-*a*:** Benthic chlorophylls were lower in Lahontan Reservoir (Table 5) than reported for other oligotrophic/mesotrophic lakes and reservoirs (Fig. 5).

2. **Benthic macroinvertebrates**: The benthic macroinvertebrates were dominated by nematodes and tubificid worms both spatially and temporally (Fig. 10). Although the number of individuals in the benthic community was greater in July 2001 than in April/May 2001, the number of taxa remained similar. Typically, nematodes and immature tubificids are not quantitatively retained by 500-micrometer sieving. The effects of particle aggregation on sieving efficiency may therefore inflate the number of total individuals relative to other studies using a 500-micrometer mesh. Reible and others (1996) reported that the flux of hydrophobic organics was significantly increased when large, laboratory-cultured tubificids were present at densities between 6,000 to 27,000 per square meter. Not only were the densities of tubificids in Lahontan Reservoir cores typically much lower, but they were also dominated by smaller immature individuals (Tables 2 – 4).

   Taxonomic analyses of July 2001 cores were also performed using individuals retained on a 106–micrometer sieve as well as those retained on a 500-micrometer sieve to check for the potential importance of losing small individuals while using conventional-sized mesh. Data from these analyses indicate that the 500-micrometer mesh lost about 50% of the individuals that were retained by the smaller mesh, but those individuals were obviously smaller and insufficient in number to alter the conclusion that the benthic macroinvertebrate community at all reservoir sites contributed little to enhance benthic flux (Word, 1980; Miller, 1984; Tables 3 and 4; Fig. 11).

Chemical Data

For consistency with previous geochemical studies, mercury flux estimates were determined in mass-flux units, but were also tabulated in molar units.

1. **Dissolved oxygen (DO) consumption**: Dissolved-oxygen fluxes were consistently negative, indicating benthic consumption. DO flux ranged from -483 micromoles per square meter per hour at Site 3 in July 2001 (average of two measurements, n=8) to -1240 micromoles per square meter per hour at Site 2 on the same date (Table 6). A comparison of average dissolved-oxygen flux measurements for the three sites indicates that the southern-lobe Sites 1 and 2, in contrast to the northern-lobe Site 3, displayed a temporal increase in oxygen consumption in July 2001 relative to April/May 2001. This observation is consistent with the lower DO level and higher bottom-water temperature in July 2001 (Fig. 4). Although the reason for contrasting temporal trends for Site 3

are unclear, the augmented flow from the Truckee Canal and modified circulation patterns in the northern lobe during the drought conditions between the two coring dates could have affected the DO flux. It should be pointed out that these temporal differences, though notable, are not statistically significant because of non-linearities in the DO time series (Table 6).

2. **Bottom-water mercury concentrations**: Dissolved total-mercury concentrations in the reservoir bottom water over the two coring dates ranged from a minimum of 375 $\pm$ 3 picomolar (n = 2) at the northern-lob Site 3 on April 30, 2001, to a maximum of 1930 $\pm$ 32 picomolar (n = 2) at the main-channel southern-lobe Site 1 on July 18, 2001 (Table 7). Similarly, dissolved methyl-mercury concentrations in the bottom water were lowest at Site 3, on both coring dates (0.4 picomolar) and highest at Site 1 on April 30, 2001 (0.9 picomolar; Table 7). Furthermore in terms of spatial distributions, the lowest concentrations were consistently observed for both dissolved total and methyl-mercury concentrations at the northern-lobe Site 3 and highest at southern-lobe Sites 1 and 2. Temporally, the total-mercury concentrations in reservoir bottom waters were higher in July than April/May 2001 by a factor of 2 or 3, but similar differences in methyl-mercury concentrations were not evident.

   Although mercury analyses for filtered surface waters are not available, concentrations of filtered bottom-water samples may be compared to a time series of surface-water analyses taken at Weeks Bridge, near the riverine inflow to the reservoir (Priessler and others, 1999; Jones and others, 1999; Allander and others, 2001). The total-mercury concentration range for reservoir bottom waters determined in this study (373 to 2068 picomolar) were consistently higher than observed at Weeks Bridge in Fort Churchill, near the Carson River inflow to the reservoir (135 $\pm$ 21 picomolar, n=56) between April, 1997 and May, 2001. In contrast, the methyl-mercury concentration range in the bottom waters (0.3 to 1.1 picomolar) was lower than observed for the riverine source (5.4 $\pm$ 1.2 picomolar, n=56).

3. **Mercury benthic flux**: For both incubation experiments, dissolved methyl-mercury fluxes were positive except from two main-channel southern-lobe Site 1 cores (Table 8). The highest methyl-mercury fluxes were at the northern-lobe Site 3, and the lowest at the main-channel southern-lobe Site 1. For both sampling dates, the magnitude of the methyl-mercury fluxes were about three orders of magnitude less than the flux estimates for total mercury, roughly consistent with concentration differences between species.

   In contrast to methyl-mercury fluxes, dissolved total-mercury fluxes were greatest at the main-channel southern-lobe Site 1. Total-mercury fluxes at the other two sites (near-shore southern-lobe Site 2 and northern-lobe Site 3) were lower in July 2001 than in April/May 2001. In both series of incubation experiments, within-site variability in the flux estimates, typically obscured between-site differences. That is, differences between fluxes for dissolved total-mercury were not statistically significant (very low coefficients of determination for the regressions) when the intermediate sampling intervals at 2 and 6 hours of elapsed time were used for calculating flux. The total-mercury concentrations in all cores initially decreased before increasing between 6 and 12 hours. This type of non-linear temporal trend for the concentrations is atypical for incubation experiments of this kind, and may reflect a transient response to a perturbation at the beginning of the incubation. One possibility is that atmospheric exposure of the overlying water added at the beginning of the incubation caused initial mercury precipitation. This explanation might be tested by incubating over longer durations (for example, a 24-hour end point) and evaluating linearity. As one might expect, the bias of omitting the initial concentrations in each time series (that is, the use of only the 2, 6 and 12 hour data in the flux calculations) would yield statistically significant regressions and much larger flux values. Such values might represent maximum potential flux estimates. Because of volume constraints, methyl-mercury concentrations were measured only at the beginning and the end of the incubation period.

   A summary of mercury-flux data from a variety of processes (Fig. 7) illustrates that flux estimates not only vary over orders of magnitude, but also vary in direction where a positive flux represents

solute transport into the water column, and a negative flux indicates solute moving from the water column to the sediment or to the atmosphere. The core-incubations were used to estimate benthic-flux over the projected area of the reservoir for the two sampling dates (31.3 and 23.8 square kilometers on April 30 and July 18, 2001, respectively), and ranged between 0.8 to 5.7 and –78 to 7291 picomoles mercury per square meter per hour for methyl and total mercury, respectively, after using the mean of two replicate cores per site. This corresponds to a mass range of –4 to 27 and -374 to 35,100 nanograms mercury per square meter per day for methyl and total mercury, respectively. Secondly, a comparison of flux magnitudes from these sources and sinks indicate no obvious dominant process. Riverine fluxes into the reservoir for the two sampling dates were based on USGS stream-flow data available on the internet (USGS, 2001) for the Carson River at Weeks Bridge, just up gradient of the reservoir, and on storage data and surface-area estimates for Lahontan Reservoir (Katzer, 1971; USGS, 2001). In addition, dissolved total and methyl-mercury data from Weeks Bridge were used (Priessler and others, 1999; Jones and others, 1999; Allander and others, 2001). For April 30 and July 18, 2001, the riverine fluxes were estimated at 12.6 and 0.1 picomoles methyl mercury per square meter per hour, and 316 and 1.9 picomoles total dissolved mercury per square meter per hour, respectively. The estimate for daily Hg emission from the reservoir (21 picomoles mercury per square meter per hour or 100 nanograms mercury per square meter per day) is based on a diel atmospheric-flux study conducted in July 2001. Based on our current comparative information, it would seem unwise to neglect any of these processes when trying to quantitatively describe mercury-transport processes within and through this system.

4. **Dissolved organic carbon (DOC)**: DOC concentrations serve as a model for the presence of organic ligands that can complex and enhance the solubility of trace metals like mercury. The partitioning and speciation of certain metals in aquatic systems are significantly affected by organic complexation (Mantoura and others, 1978; McKnight and others, 1983) and, in particular, for mercury. Ligands represented by dissolved organic carbon can compete with sulfide to regulate solubility (see section 5 below, Ravichandran and others, 1998). This solute interdependency is consistent with concentration trends in bottom-water DOC (Table 9). Similar to total dissolved mercury in the bottom waters, DOC was: (1) significantly lowest on both dates at the northern-lobe site 3, and (2) consistently higher at all sites in July 2001 by approximately 40% relative to April/May 2001 concentrations. Application of a physical-transport model might clarify whether this temporal increase in the summer was primarily due to one or more of the following: snow-melt dilution (leading to lower DOC levels during April/May), a source of DOC available during the summer within the watershed, temporal changes in mass transport through the reservoir, and/or evaporation.

   The benthic flux of DOC ranged from –45 micromoles per square meter per hour at Site 3 in July 2001 (average of two measurements, n=8) to 116 micromoles per square meter per hour at Site 2 in April/May 2001 (Table 10). The fact that the highest measured DOC flux into the sediment at Site 3 was coincident with the lowest average consumption rate for dissolved oxygen (Table 6) suggests an electron-donor (organic carbon) limitation at Site 3. Similarly in July 2001, the highest DOC flux out of the sediment at Site 2 was coincident with the highest average rate of benthic consumption for dissolved-oxygen. This is not surprising with elevated DO concentrations in the shallower depths of the southern lobe relative to those of northern lobe bottom water (Fig. 4). In April/May 2001, the highest DOC flux out of the sediment was also at Site 2, but dissolved-oxygen saturation in the bottom waters of all three coring sites generated spatially comparable DO fluxes. This temporal shift in bottom-water DO concentrations may also have contributed to the reversal in direction of DOC flux for Site 3 between the April/May and July 2001 experiments.

5. **Dissolved Sulfides**: Sulfide flux was consistently positive (that is, from the sediment to the overlying water column) and ranged from 419 to 766 nanomoles per square meter per day (average of two

replicates per site per date; Table 11). A spatial comparison of dissolved-sulfide benthic fluxes indicates enhanced flux at the northern-lobe Site 3 in April/May 2001 relative to the southern-lobe Sites 1 and 2, but not in July 2001. For both coring dates, the sulfide benthic fluxes from the southern-lobe sites were statistically similar. Temporally, all sites exhibited an increase in dissolved-sulfide flux in July relative to April/May 2001, but this difference was most pronounced at the southern-lobe Sites 1 and 2 where flux nearly doubled between experiments. In contrast, the increase in average dissolved-sulfide flux for northern-lobe Site 3 was not statistically significant.

Using dissolved-sulfide concentrations from the bottom water and surficial porewater, a diffusive flux from the benthos was calculated with a diffusion coefficient of $1x\ 10^{-5}$ square centimeters per second (Li and Gregory, 1974). With one exception, the calculated diffusive-flux estimates were lower than directly measured sulfide-fluxes and were of the same order of magnitude (Table 11). In contrast, diffusive sulfide fluxes were three orders of magnitude lower than measured sulfide benthic flux in San Francisco Bay where bioturbation effects have been well documented (Caffrey and others, 1996; Kuwabara and others, 1999). This comparison supported the conclusion that the reservoir had insufficient biomass of critical macroinvertebrate species to impose bioturbation or bioirrigation effects observed in other environments.

Dissolved sulfides represent a metastable ligand with high affinity to complex most divalent metals, but particularly mercury that has a logarithmic mercuric-sulfide solubility product of approximately –50 (Hogfeldt, 1983). At thermodynamic equilibrium, the product of the molar concentrations of uncomplexed mercury and uncomplexed sulfide ions is expected to be very low (of the order of 10^{-50}) due to the formation of mercuric sulfide.

Study Design and Methods

The protocol described in this section focuses on method applications in this series of two core-incubation experiments. Details (e.g., quality control specifications) for each analysis have been previously documented (Woods and others, 1999; Praskins and others, 2001). Sampling was performed at a main-channel and near-shore site in the southern lobe of the reservoir (Fig. 1) (39 degrees 26.6328 minutes north latitude by 119 degrees 10.1403 minutes west longitude, and 39 degrees 22.9939 minutes N by 119 degrees 9.9412 minutes W, respectively), and one site up gradient of Gull Island in the northern lobe (39 degrees 26.3302 minutes N by 119 degrees 4.5980 minutes W). Horizontal coordinate information is referenced to the North American Datum of 1983 (NAD 83). Cores were taken during the morning to avoid the effects of wind on core retrieval and sediment resuspension. The site map (Fig. 1) also indicates flow of the Carson River into the southern delta region and the out-flow from the northern lobe, regulated at the Lahontan Dam. Also depicted is the Truckee Canal, a regulated supplemental input to the reservoir.

Coring Operation

On both sampling days (April 30 and July 18, 2001), three cores were collected at each of three sites using a coring device fabricated from non-metallic parts (Fig. 2, Savillex Corporation, Minnetonka, Minnesota). To avoid sample contamination, wetted surfaces of the coring device and core tubes were acid-washed polypropylene or fluoroethylene polymer. Because the bottom water was aerobic on both sampling dates, and throughout that summer, the nine cores were aerated overnight to condition wetted surfaces before the incubation period (Topping and others, 2001). Two of the three cores per site were selected for incubation based on visual inspection of the sediment-water interface for any disturbances. Water overlying the sediment in the selected cores was sampled at four intervals during a 12-hour incubation. Dissolved-mercury samples were processed in a Class-100 laminar-flow hood. The time series of solute concentrations was used to determine the benthic flux from each of the incubated cores.

Physical Data

Sediment Porosity: After core-incubations, approximately 10 milliliters of surficial sediment was collected from each core. Wet weight and dry weight after lyophilization were measured to calculate porosity (Table 1).

Biological Data

1. **Benthic Macrofauna**: After core incubations, each core was sieved (500-micrometer mesh). The sieved samples were fixed with 10% buffered formalin, transferred to 70% ethanol, then sorted and identified to the lowest practicable taxonomic level (Fig. 12). In July 2001, the sediments were also sieved at 106 micrometers to provide additional information about the scarcity of benthic invertebrates. Samples were stained with rose bengal to facilitate sorting.

2. **Benthic Chlorophyll-*a***: Each incubated core was sub-sampled in triplicate for benthic chlorophyll-*a*. Surficial sediment was collected on a glass-fiber filter and buffered with magnesium carbonate. Samples were then frozen in darkness for preservation until spectrophotometrically analyzed by methods described in Thompson and others (1981) and Franson (1985) (Fig. 13).

Chemical Parameters

1. **Bottom-water sampling**: Prior to coring at each sampling site, bottom-water samples (approximately 1 meter above the sediment-water interface) were collected for subsequent analysis of dissolved-mercury speciation by cold-vapor atomic fluorescence spectroscopy (CVAFS) using a high-displacement peristaltic pump and a tethered length of rigid fluoroethylene polymer tubing. These samples were used to replace overlying water in cores collected from each site for incubation.

2. **Dissolved Oxygen (DO)**: Consumption of dissolved oxygen was monitored for each incubated core using a micro-Winkler titration method (Fig. 14; Kuwabara and others, 2000).

3. **Dissolved Mercury**: Mercury samples from water-column sampling, and core incubations were filtered with 0.7-micrometer quartz membranes that had been baked for 6 hours at 500 degrees centigrade to remove residual organics. Filtered samples were acidified with quartz-distilled nitric acid (pH less than 2), and refrigerated in darkness until analyzed by CVAFS (Fig. 15, Krabbenhoft and others, 1998).

4. **Dissolved organic carbon (DOC)**: Dissolved organic carbon was determined by high-temperature, non-catalytic combustion (Qian and Mopper, 1997). Potassium phthalate was used as the standard. Low-DOC water (blanks <40 micrograms-organic C per liter) was generated from a double-deionization unit with additional ultraviolet treatment (Milli-Q Gradient, Millipore Corporation) (Fig. 16).

5. **Sulfides**: Dissolved sulfides in overlying-water samples were analyzed by square-wave voltammetry (Fig. 17; Kuwabara and Luther, 1993).

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Appendix 1: Comments on the Report Structure

A major objective of this electronic document is to provide a structure that is easily accessible to the wide range of interests in this work**.** Therefore, pathways within this document have been constructed to be both logical and intuitive. In addition to hyperlinks within the document to supporting figures and tables, links in Appendices 2 and 3 provide a quick way to directly review and examine all figures and tables.

Although hard copies of this report are available on request, the advantages of the electronic version relative to the hard copy are substantial in many respects, but particularly in the rapid access of information at multiple levels of detail.

Your comments about how to better serve readers with this evolving type of product are most welcome and may be directed to any of the email addresses provided in the title page, but we request that at least a copy of all comments about the report and its format be addressed to the major author (kuwabara@usgs.gov) so that they may be compiled for future revisions and reports.

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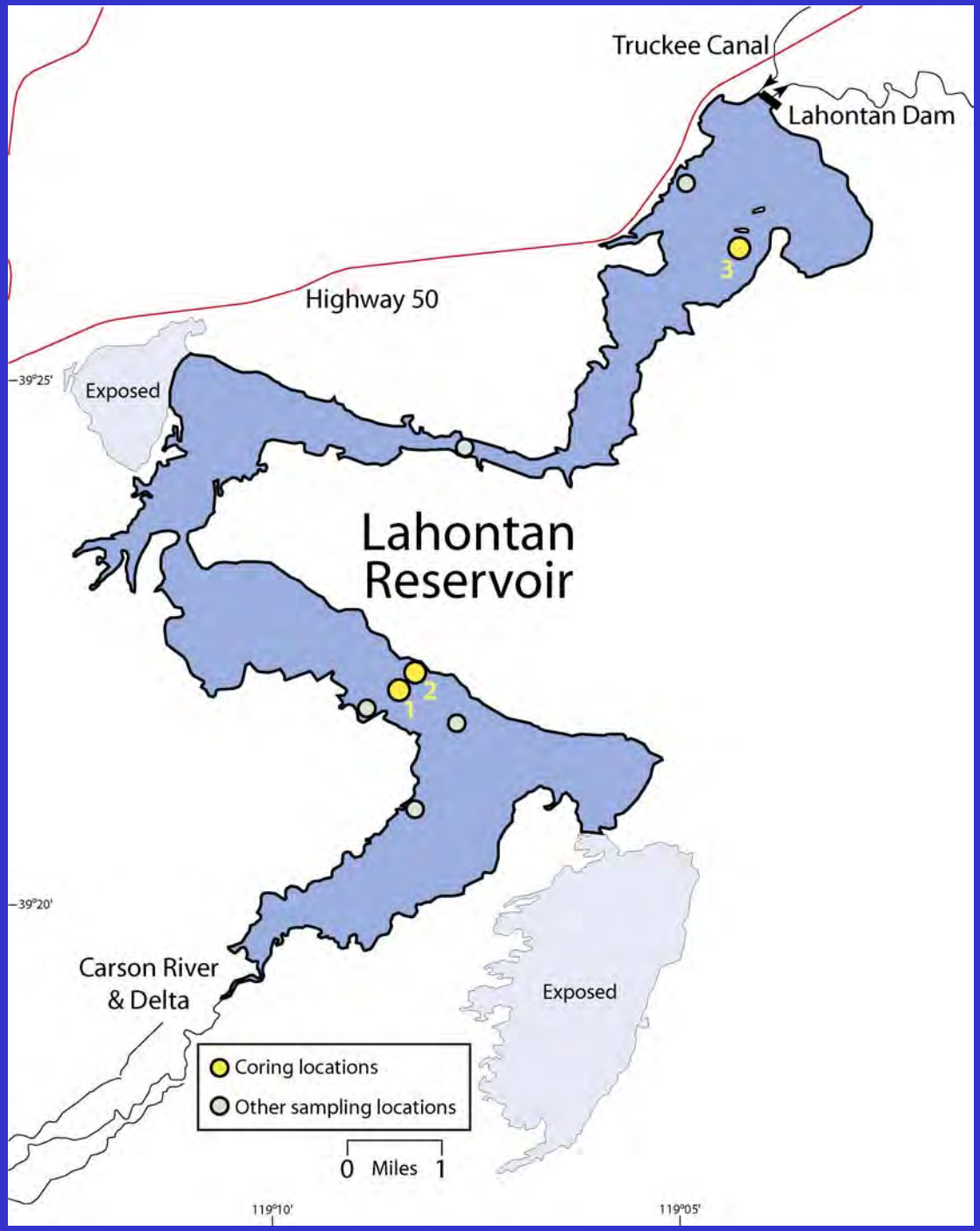
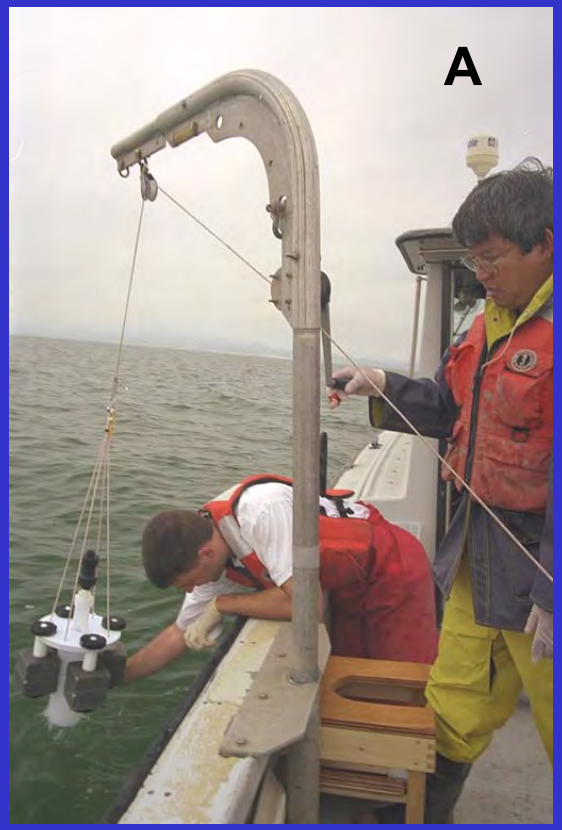
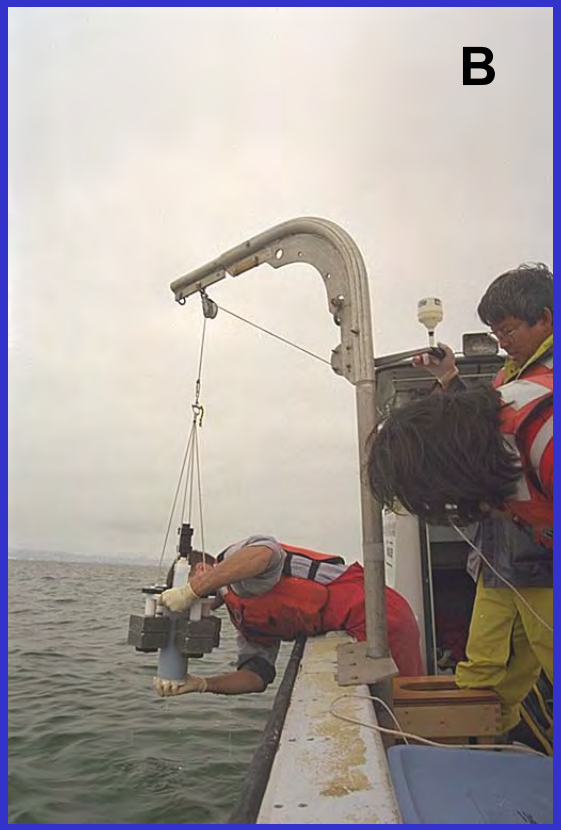
Fig. 1. Site map of the study area including locations of sampling and coring sites in Lahontan Reservoir.



USGS

Fig. 2. Photos of the coring operation showing the release of the non-metallic coring device (A), capping of the core upon retrieval (B) and removal of the core tube (C) in preparation for incubation.





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Modified from Kuwabara and others (2000)

Fig. 3. Hydrologic conditions during the study period, including 2001 inflow from the Carson River, averaged historical daily inflow since 1911, and reservoir storage (http://water.usgs.gov/nv/nwis).

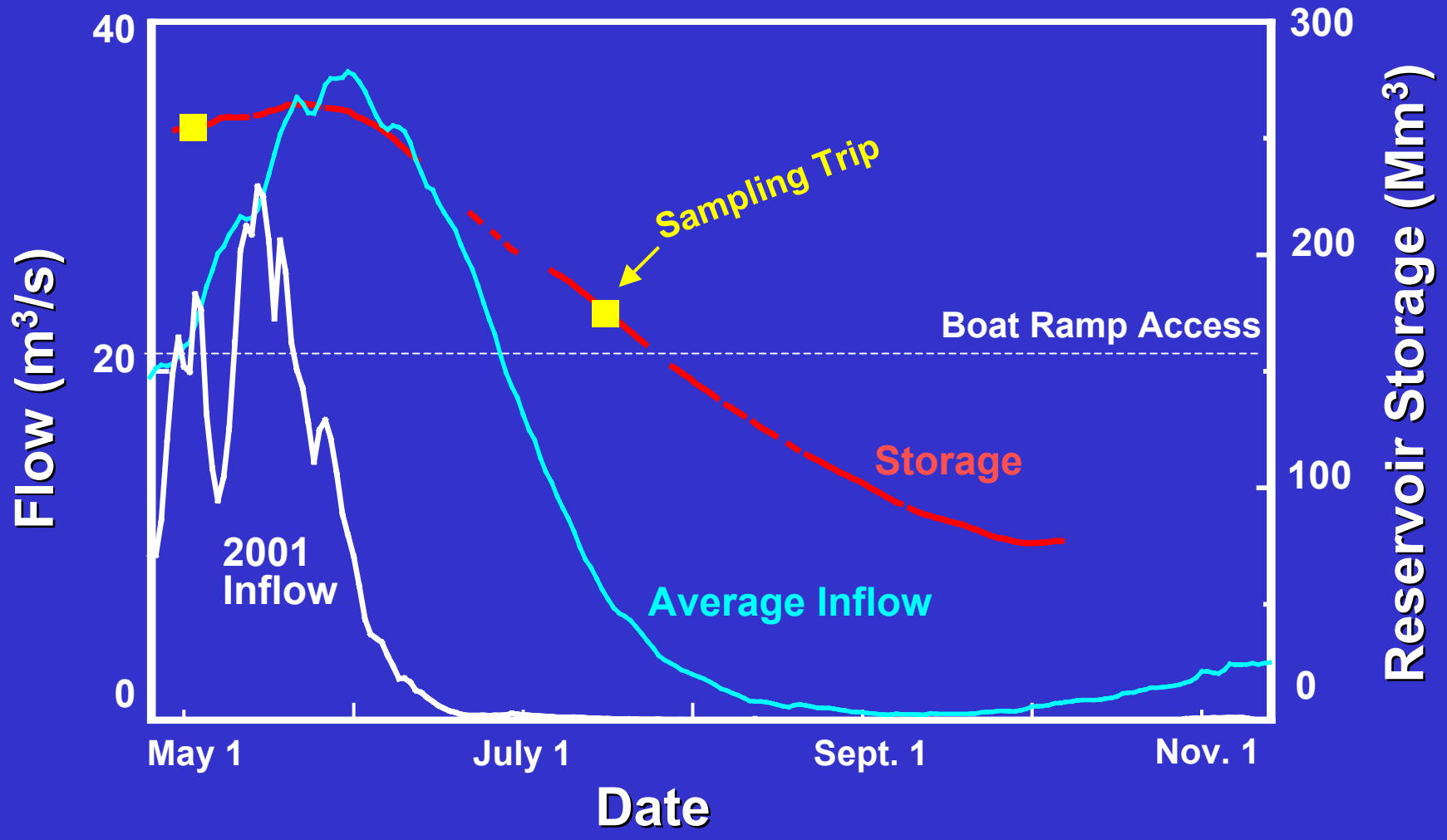

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Fig. 4. Temperature and dissolved oxygen (DO) concentration trends in water-column profiles between the May and July sampling periods, with arrows representing the direction of temporal change.

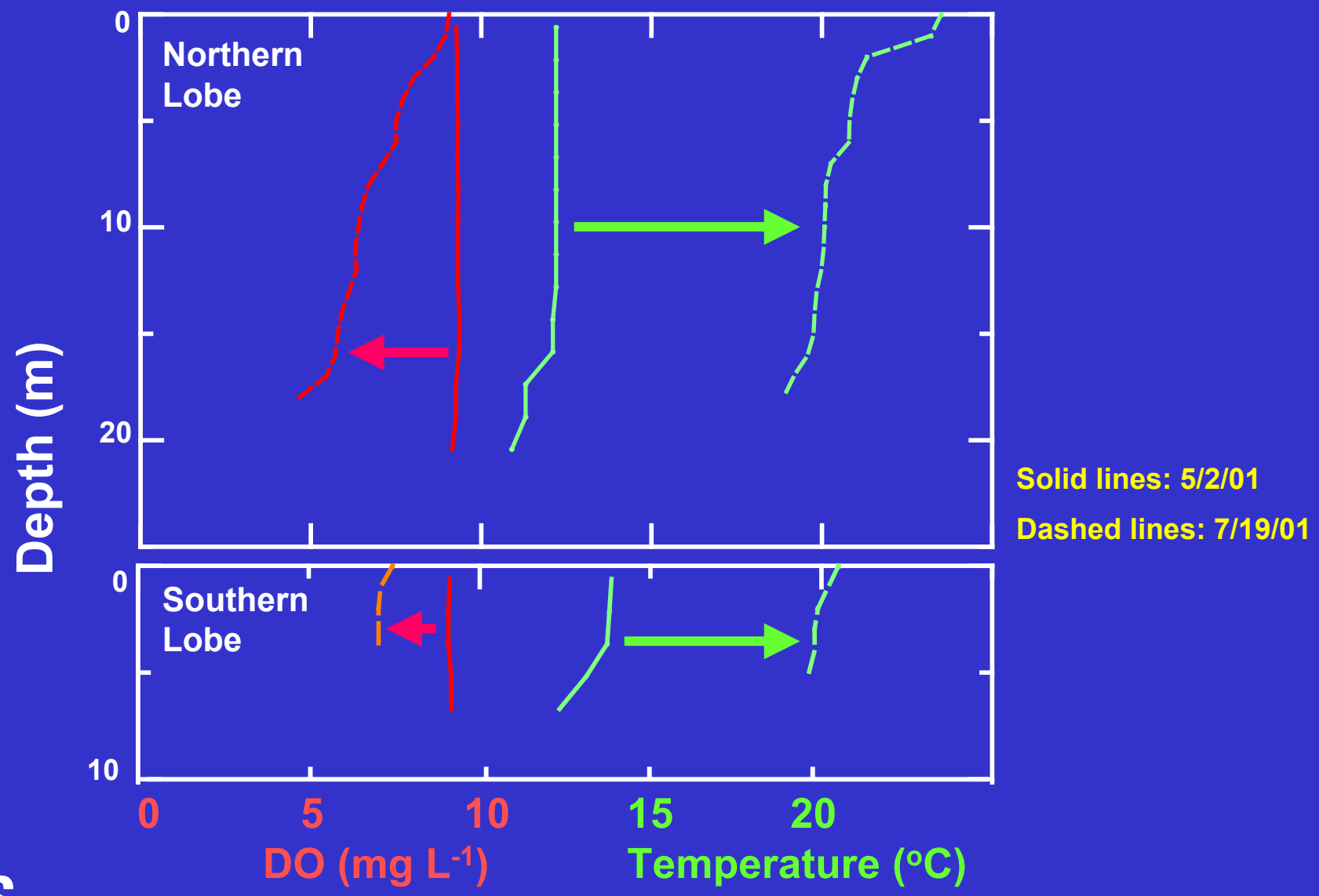

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Fig. 5 Comparison of the ranges for benthic indices (benthic chlorophyll-*a* and macroinvertebrate abundance) determined during this study in 2001, to those reported for other oligotrophic/mesotrophic lakes and reservoirs.

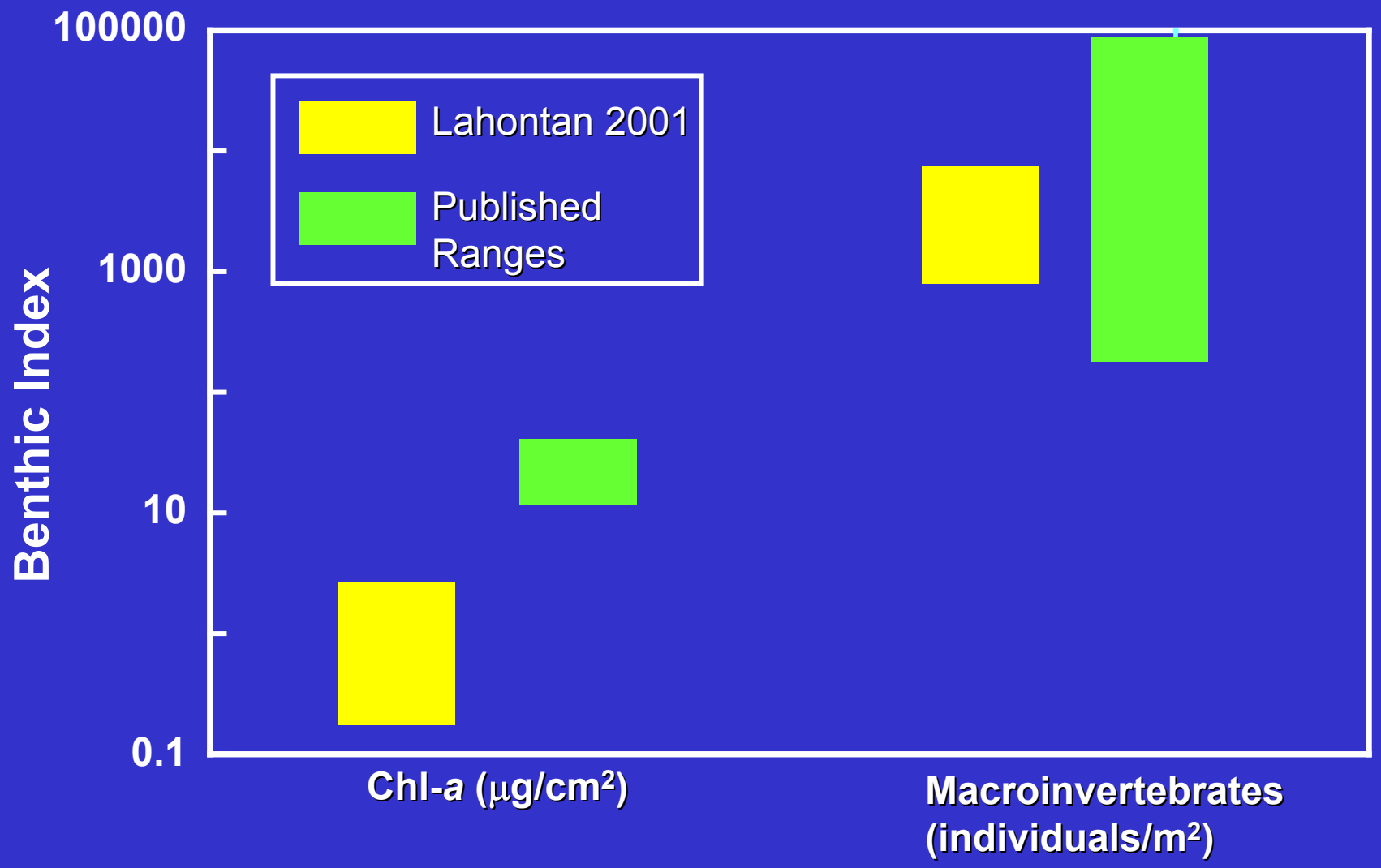

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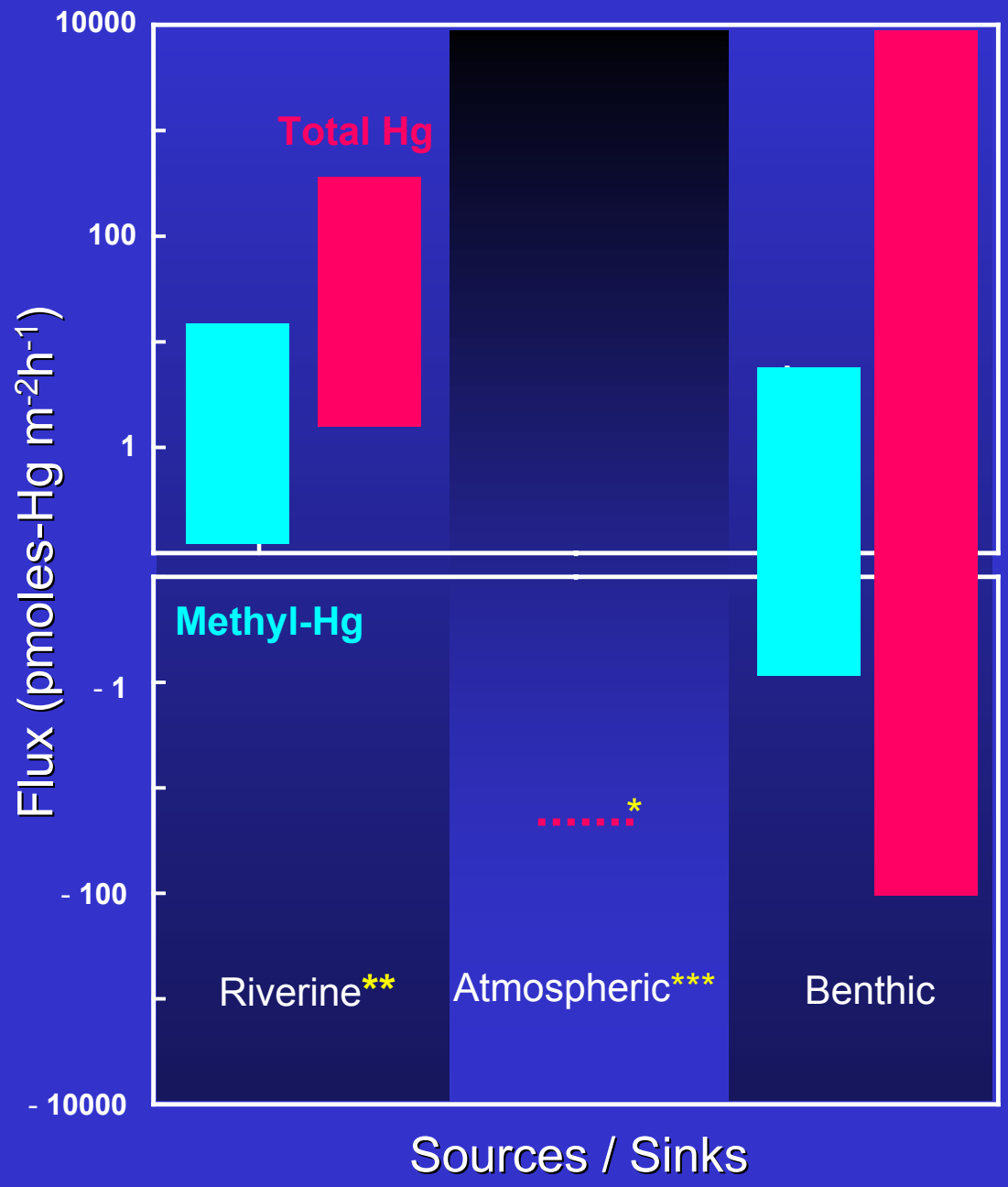
Fig. 6. Methyl-Hg and Total-Hg flux comparisons between riverine, atmospheric and benthic sources and/or sinks. Positive flux indicates flux into the water column.



* Daily emission estimate

** Source: http://water.usgs.gov/nv/nwis

*** Source: Estimate from diel-flux study in July, 2001

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Fig. 7. Illustration showing how remediation efforts in the river upstream can affect gradients in the lake. A regulated decrease in contaminant concentration in the water-column (green arrow) generates an increased concentration gradient across the sediment-water interface resulting in conditions favorable for enhanced benthic flux (yellow arrow).

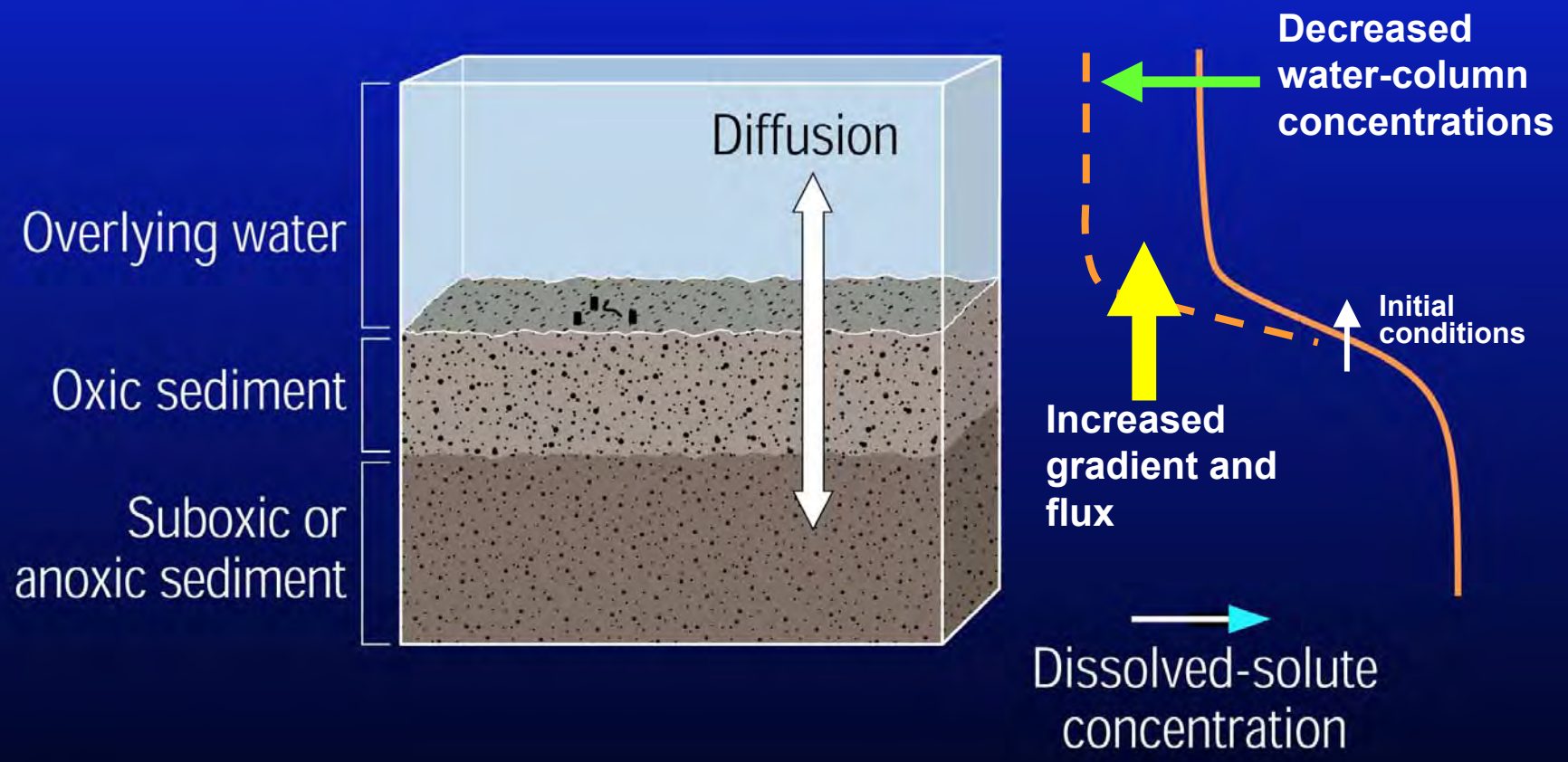

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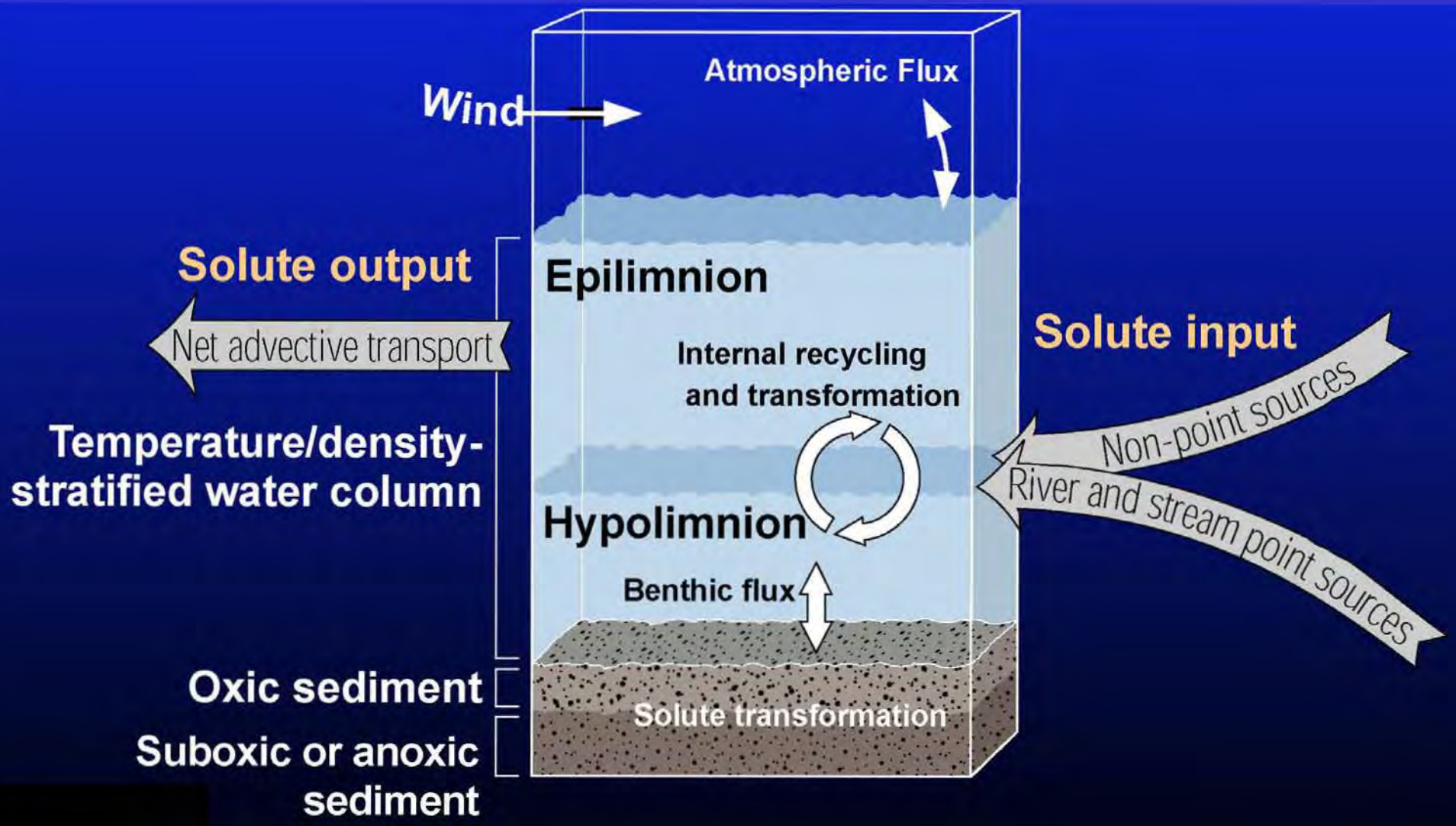
Fig. 8. Conceptual model of solute transport through a reservoir.



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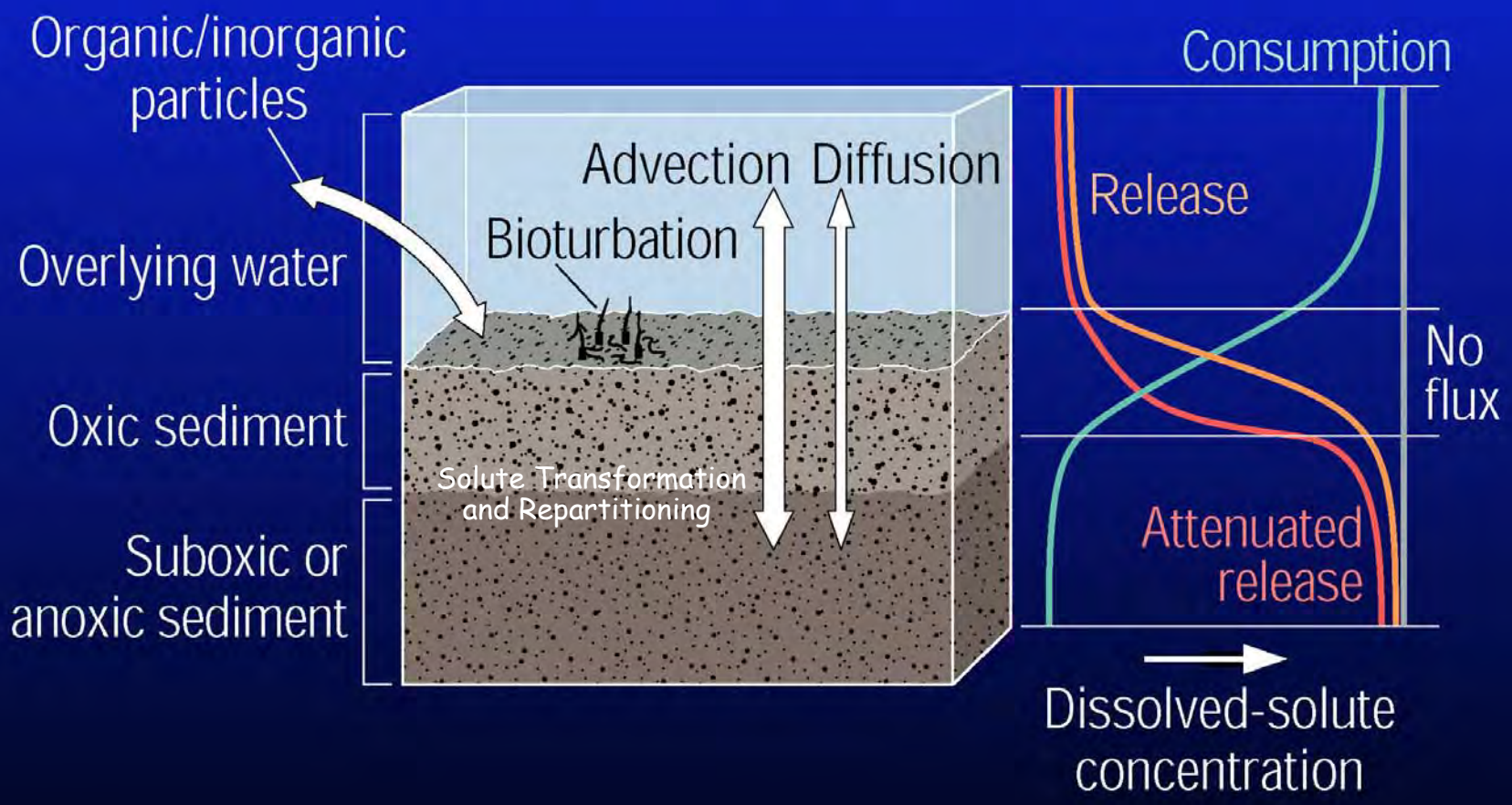
Modified from Kuwabara and others (2000)

Fig. 9. Processes regulating the benthic flux of solutes.



Modified from Kuwabara and others (2000)

Fig. 10. Abundance of benthic macroinvertebrates at each of the three sampling sites, during both sampling periods. The predominance of nematodes and tubificid worms is highlighted.

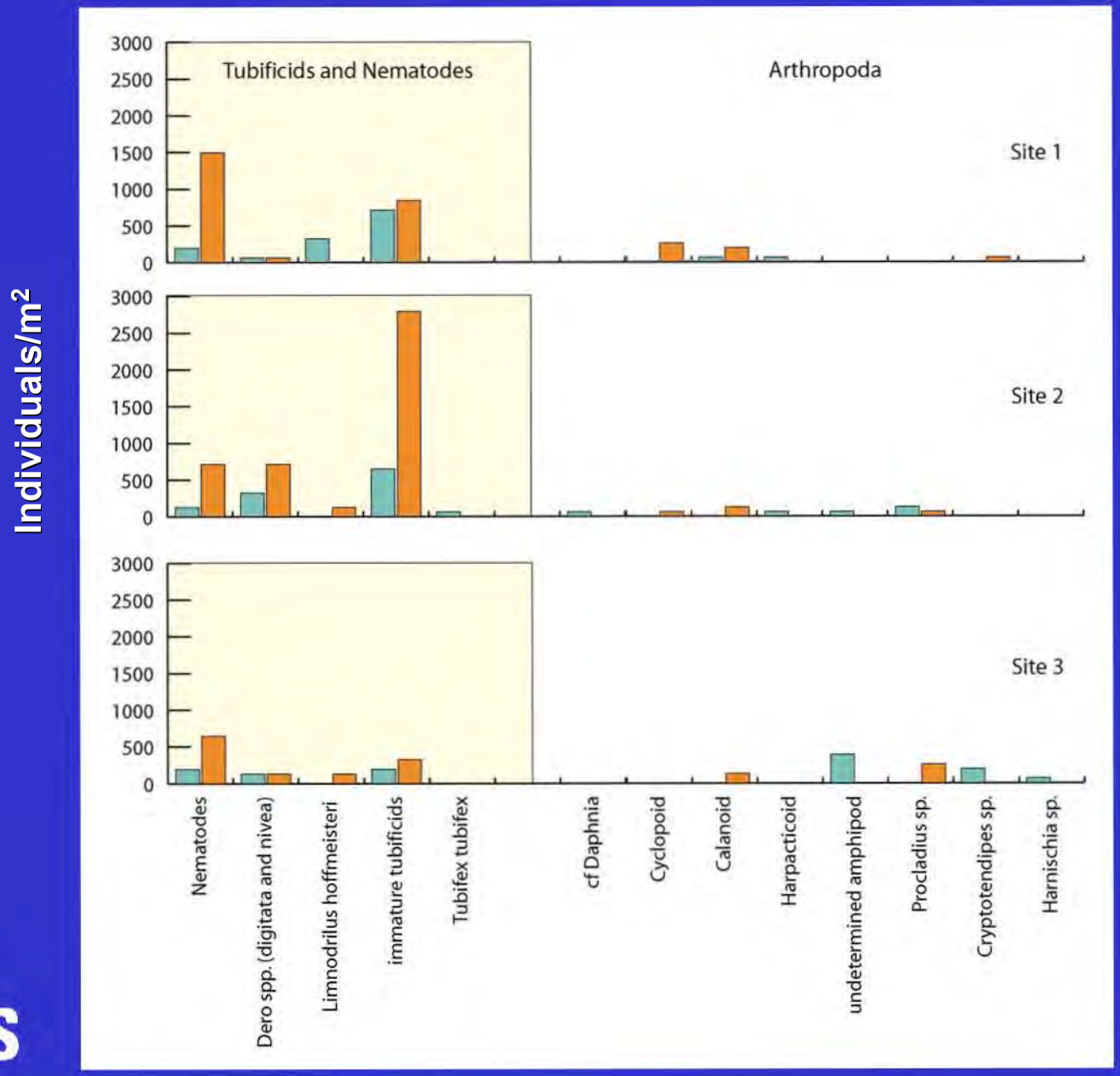


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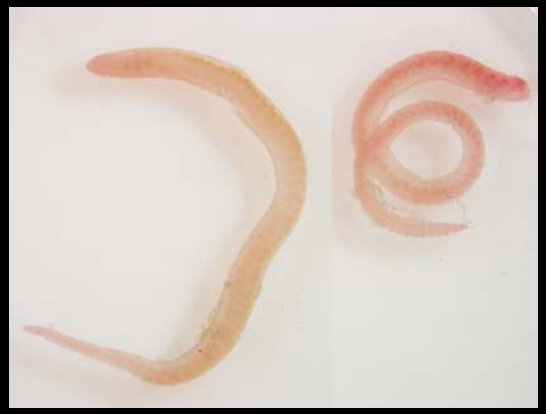
Fig. 11. Photomicrographs showing size variation of individuals retained by different sieve sizes.

Mature tubificid

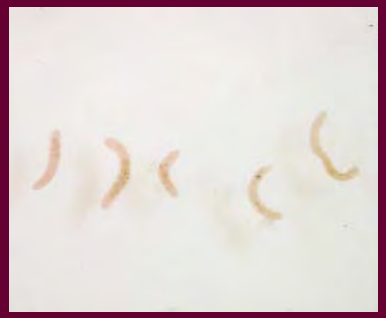


Immature tubificids

500 μm sieved

500 μm 1000 μm

Naidids



Immature tubificids

100 μm sieved

USGS

Photomicrographs by S.V. Fend

Fig. 12. Photograph showing the process of sorting and identification to the lowest practicable taxonomic level.

USGS

Modified from Kuwabara and others (2000)

Fig. 13. Photograph showing spectrophotometric equipment used for benthic-chlorophyll analyses.

USGS

Modified from Kuwabara and others (2000)

Fig. 14. Photograph showing Micro-Winkler Titration equipment used for dissolved-oxygen analyses during benthic-flux experiments.

USGS

Fig. 15. Photograph of Cold-vapor Atomic Fluorescence Spectroscopy (CVAFS) equipment used for analysis of dissolved-mercury speciation.

USGS

Modified from Kuwabara and others (2000)

Fig. 16. Photograph showing equipment used for Dissolved Organic Carbon (DOC) analyses by high-temperature non-catalytic oxidation.

USGS

Modified from Kuwabara and others (2000)

Fig. 17. Photograph showing polarographic equipment used for dissolved-sulfide analyses by square-wave voltammetry.

USGS

Modified from Topping and others (2001)

Table 1. Surficial-sediment Porosities for Lahontan Reservoir, 2001

5/1/2001		
Site#	Core#	Porosity
1	1	**0.65**
1	3	**0.65**
2	4	**0.81**
2	9	**0.80**
3	12	**0.89**
3	13	**0.91**

7/19/2001		
Site#	Core#	Porosity
1	1	**0.73**
1	3	**0.70**
2	6	**0.80**
2	7	**0.85**
3	10	**0.93**
3	11	**0.92**

Table 2. Macro-invertebrate taxonomy for Lahontan core samples 5/1/01, 500 uM sieve

Mean number of individuals per square meter based on core surface area of 77 square centimeters.

Note:Species are listed below in the same general order as on the plots	individuals/m^2					
	Site 1		Site 2		Site 3	
Core Number:	1	3	4	9	12	13
Phylum Platyhelminthes						
Class Turbellaria						
Order Rhabdocoela						
Gyratrix sp.						
Phylum Nematoda						
Nematodes		390		260	260	130
Phylum Annelida						
Class Oligochaeta						
Order Tubificida						
Family Naididae						
Amphichaeta leydigi						
Chaetogaster diastrophus						
Dero spp. (*digitata and nivea*)		130	390	260		260
cf. *Pristinells acuminata*						
Vejdovskyella comata						
Vejdovskyella intermedia						
Family Tubificidae						
Limnodrilus hoffmeisteri	390	260				
immature tubificids	390	1039	779	519	130	260
Tubifex tubifex				130		
Phylum Arthropoda						
Class Ostracoda						
Order Podocopida						
Ostracod						
Class Branchiopoda						
Order Anomopoda						
Family Daphnidae						
cf *Daphnia*			130			
Class Copepoda						
Order Cyclopoida						
Cyclopoid						
Order Calanoida						
Calanoid		130				
Order Harpacticoida						
Harpacticoid	130			130		
Class Malacostraca						
Order Amphipoda						
undetermined amphipod				130	130	649
Class Insecta						
Order Diptera						
Family Chironomidae						
Procladius sp.			130	130		
Cryptotendipes sp.					390	
Harnischia sp.						130
Macroinvertebrate Density (ind/m^2) =	909	1948	1429	1558	909	1429

Table 3. Macro-invertebrate taxonomy for Lahontan core samples 07/19/01, 500 uM sieve

Mean number of individuals per square meter based on core surface area of 77 square centimeters.

Note:Species are listed below in the same general order as on the plots	individuals/m^2					
	Site 1		Site 2		Site 3	
Core Number:	1	3	6	7	10	11
Phylum Platyhelminthes						
Class Turbellaria						
Order Rhabdocoela						
Gyratrix sp.						
Phylum Nematoda						
Nematodes	779	2208	260	1169	1039	260
Phylum Annelida						
Class Oligochaeta						
Order Tubificida						
Family Naididae						
Amphichaeta leydigi						
Chaetogaster diastrophus						
Dero spp. (*digitata and nivea*)	130		779	649		260
cf. *Pristinells acuminata*						
Vejdovskyella comata						
Vejdovskyella intermedia						
Family Tubificidae						
Limnodrilus hoffmeisteri			260		260	
immature tubificids	649	1039	909	4675	390	260
Tubifex tubifex						
Phylum Arthropoda						
Class Ostracoda						
Order Podocopida						
Ostracod						
Class Branchiopoda						
Order Anomopoda						
Family Daphnidae						
cf *Daphnia*						
Class Copepoda						
Order Cyclopoida						
Cyclopoid	390	130		130		
Order Calanoida						
Calanoid	130	260	260		130	130
Order Harpacticoida						
Harpacticoid						
Class Malacostraca						
Order Amphipoda						
undetermined amphipod						
Class Insecta						
Order Diptera						
Family Chironomidae						
Procladius sp.				130	519	
Cryptotendipes sp.		130				
Harnischia sp.						
Macroinvertebrate Density (ind/m^2) =	2078	3766	2468	6753	2338	909

Table 4. Macro-invertebrate taxonomy for Lahontan core samples 07/19/01, 106 uM sieve

Mean number of individuals per square meter based on core surface area of 77 square centimeters.

Note:Species are listed below in the same general order as on the plots	individuals/m^2					
	Site 1		Site 2		Site 3	
Core Number:	1	3	6	7	10	11
Phylum Platyhelminthes						
Class Turbellaria						
Order Rhabdocoela						
Gyratrix sp.	260					
Phylum Nematoda						
Nematodes	519	260	1429	130	6364	909
Phylum Annelida						
Class Oligochaeta						
Order Tubificida						
Family Naididae						
Amphichaeta leydigi					2208	1429
Chaetogaster diastrophus					130	
Dero spp. (*digitata and nivea*)				130		
cf. *Pristinells acuminata*					130	
Vejdovskyella comata			260	130	1169	130
Vejdovskyella intermedia		260	130			
Family Tubificidae						
Limnodrilus hoffmeisteri						
immature tubificids	260	519	909	2338	649	130
Tubifex tubifex						
Phylum Arthropoda						
Class Ostracoda						
Order Podocopida						
Ostracod	130					
Class Branchiopoda						
Order Anomopoda						
Family Daphnidae						
cf *Daphnia*	130					
Class Copepoda						
Order Cyclopoida						
Cyclopoid	649	1169	1558	649	1688	130
Order Calanoida						
Calanoid		130			130	130
Order Harpacticoida						
Harpacticoid						
Class Malacostraca						
Order Amphipoda						
undetermined amphipod						
Class Insecta						
Order Diptera						
Family Chironomidae						
Procladius sp.					260	
Cryptotendipes sp.						
Harnischia sp.	130					
Macroinvertebrate Density (ind/m^2) =	1818	2338	4286	3377	12727	2857

Table 5. Benthic Chlorophyll concentrations for Lahontan Reservoir, 2001

5/1/01		Chlorophyll-a	
		Mean	S.D.
Site#	Core#	ug/cm^2	ug/cm^2
1	1	**0.89**	0.38
1	3	**1.18**	0.51
2	4	**2.14**	0.33
2	9	**1.23**	0.92
3	12	**0.82**	0.74
3	13	**1.51**	0.88

7/19/01		Chlorophyll-a	
		Mean	S.D.
Site#	Core#	ug/cm^2	ug/cm^2
1	1	**0.97**	0.52
1	3	**1.08**	0.20
2	6	**1.77**	0.31
2	7	**1.70**	0.43
3	10	**1.40**	0.60
3	11	**0.78**	0.05

S.D.=Standard deviation

Table 6. Dissolved-oxygen flux for Lahontan Reservoir, 2001

May 1, 2001						
		DO Flux (umoles/m^2-h)				
Core#	Site#	Mean		CI	r^2	n
1	1	**-843**	+/-	2061	0.52	4
3	1	**-1088**	+/-	1995	0.66	4
4	2	**-1072**	+/-	416	0.98	4
9	2	**-585**	+/-	1676	0.44	4
12	3	**-756**	+/-	1470	0.63	4
13	3	**-1034**	+/-	2308	0.56	4

July 19, 2001						
		DO Flux (umoles/m^2-h)				
Core#	Site#	Mean		CI	r^2	n
1	1	**-860**	+/-	1928	0.56	4
3	1	**-1268**	+/-	2307	0.66	4
6	2	**-1331**	+/-	1942	0.75	4
7	2	**-1149**	+/-	1836	0.72	4
10	3	**-721**	+/-	491	0.93	4
11	3	**-245**	+/-	1185	0.21	4

CI=95% confidence interval

Table 7. Bottom-water concentrations for mercury species in Lahontan Reservoir, 2001

April 30, 2001						
	Methy-Hg (pM)			Total-Hg (pM)		
Site#	Ind. Rep	Site avg.	S.D.	Ind. Rep	Site avg.	S.D.
1	0.7	**0.9**	0.3	647	**671**	34
	1.1			695		
2	0.8	**0.8**	0.0	613	**632**	27
	0.8			651		
3	0.4	**0.4**	0.0	373	**375**	3
	0.5			377		

July 18, 2001						
	Methy-Hg (pM)			Total-Hg (pM)		
Site#	Ind. Rep	Site avg.	S.D.	Ind. Rep	Site avg.	S.D.
1	0.7	**0.8**	0.1	1907	**1930**	32
	0.9			1952		
2	0.6	**0.7**	0.1	1622	**1845**	315
	0.7			2068		
3	0.3	**0.4**	0.2	676	**732**	78
	0.5			787		

pM=picomolar

S.D.=Standard deviation

Ind. Rep. = Individual sample replicates

Table 8. Dissolved-mercury Fluxes for Lahontan Reservoir, 2001

May 1, 2001

Methyl-mercury flux

Core#	Site#	in pmoles/m^2-h	Site avg.	in ng/m^2-day	Site avg.
1	1	-1.9		-9.3	
3	1	1.7	**-0.1**	8.3	**-0.5**
4	2	3.3		15.8	
9	2	1.6	**2.4**	7.5	**11.7**
12	3	1.8		8.5	
13	3	2.9	**2.3**	13.9	**11.2**

Total Mercury flux (End points only)

Core#	Site#	in pmoles/m^2-h	Site avg.	in ng/m^2-day	Site avg.
1	1	2113		10174	
3	1	1362	**1738**	6554	**8364**
4	2	-1401		-6745	
9	2	2030	**314**	9771	**1513**
12	3	1358		6539	
13	3	1266	**1312**	6094	**6316**

Total Mercury (Entire time series)

Core#	Site#	in pmoles/m^2-h Mean		CI	in ng/m^2-day Mean		CI	r^2	n
1	1	**1770**	+/-	18345	**8521**	+/-	88312	0.06	4
3	1	**918**	+/-	20810	**4420**	+/-	100177	0.01	4
4	2	**-331**	+/-	8091	**-1594**	+/-	38947	0.01	4
9	2	**1560**	+/-	16940	**7512**	+/-	81548	0.05	4
12	3	**1310**	+/-	10515	**6304**	+/-	50620	0.09	4
13	3	**1633**	+/-	3740	**7863**	+/-	18005	0.06	4

July 19, 2001

Methyl-mercury flux

Core#	Site#	in pmoles/m^2-h	Site avg.	in ng/m^2-day	Site avg.
1	1	0.4		2.0	
3	1	-2.1	**-0.8**	-9.9	**-3.9**
6	2	1.6		7.6	
7	2	5.7	**3.6**	27.3	**17.4**
10	3	4.8		23.1	
11	3	6.6	**5.7**	31.7	**27.4**

Total Mercury flux (End points only)

Core#	Site#	in pmoles/m^2-h	Site avg.	in ng/m^2-day	Site avg.
1	1	5454		26253	
3	1	9129	**7291**	43948	**35101**
6	2	49		236	
7	2	1190	**620**	5729	**2983**
10	3	-793		-3819	
11	3	638	**-78**	3072	**-374**

Total Mercury (Entire time series)

Core#	Site#	in pmoles/m^2-h Mean		CI	in ng/m^2-day Mean		CI	r^2	n
1	1	**3661**	+/-	77454	**17622**	+/-	372858	0.01	4
3	1	**7813**	+/-	89964	**37614**	+/-	433081	0.05	4
6	2	**-940**	+/-	85015	**-4527**	+/-	409253	0.00	4
7	2	**-476**	+/-	70918	**-2293**	+/-	341392	0.00	4
10	3	**-734**	+/-	340	**-3532**	+/-	1637	0.97	4
11	3	**375**	+/-	4599	**1807**	+/-	22139	0.04	4

CI=95% confidence interval

Table 9. Bottom-water DOC Concentrations for Lahontan Reservoir, 2001

		Dissolved Organic Carbon (micromolar, uM)								Temporal
		30-Apr-01				**18-Jul-01**				Increase
	Rep	DOC (uM)		CI	n	DOC (uM)		CI	n	(%)
Site 1	A	**275.5**	+/-	2.0	5	**375.2**	+/-	1.6	4	**36.2**
	B	**275.6**	+/-	0.7	3	**381.5**	+/-	4.2	4	**38.4**
Site 2	A	**276.8**	+/-	0.8	3	**385.0**	+/-	1.6	4	**39.1**
	B	**282.0**	+/-	0.7	4	**404.4**	+/-	0.3	3	**43.4**
Site 3	A	**211.9**	+/-	1.8	5	**280.9**	+/-	1.2	4	**32.5**
	B	**207.0**	+/-	0.3	3	**297.1**	+/-	0.6	3	**43.5**

Rep= Individual sample replicate

CI= 95% Confidence Interval

Table 10. Dissolved Organic Carbon (DOC) Fluxes for Lahontan Reservoir, 2001

May 1, 2001							
		DOC Flux (umoles/m^2-h)					
Core#	Site#	Mean		CI	r^2	n	Site avg.
1	1	**-19**	+/-	17	0.05	14	**23**
3	1	**64**	+/-	11	0.49	16	
4	2	**96**	+/-	25	0.35	15	**116**
9	2	**136**	+/-	13	0.79	15	
12	3	**89**	+/-	15	0.47	17	**66**
13	3	**42**	+/-	13	0.24	16	

July 19, 2001							
		DOC Flux (umoles/m^2-h)					
Core#	Site#	Mean		CI	r^2	n	Site avg.
1	1	**19**	+/-	2	0.79	14	**26**
3	1	**32**	+/-	2	0.90	17	
6	2	**25**	+/-	12	0.11	16	**49**
7	2	**72**	+/-	14	0.53	14	
10	3	**-9**	+/-	10	0.04	14	**-45**
11	3	**-80**	+/-	18	0.45	14	

CI= 95% Confidence Interval

Table 11. Dissolved Sulfide Flux and Diffusive Estimates for Lahontan Reservoir, 2001

May 1, 2001										
		Sulfide Flux (nmoles/m2-h)								
Core#	Site#	Mean		CI	r^2	n	Porewater S^{-2} (uM)	Bottom-water S^{-2} (nM)	Conc. Grad. Gradient (nM)	Calculated **Diffusive** S^{-2} flux (nmoles/m2-h)	Site averaged **Measured** S^{-2} flux (nmoles/m2-h)
1	1	**341**	+/-	33	0.96	8					
3	1	**497**	+/-	40	0.96	9	0.70	1.15	701	**126**	**419**
4	2	**500**	+/-	29	0.98	8					
9	2	**349**	+/-	63	0.87	8	2.46	5.14	2456	**442**	**424**
12	3	**705**	+/-	128	0.81	9					
13	3	**527**	+/-	69	0.89	9	2.55	5.26	2547	**458**	**616**

July 19, 2001										
		Sulfide Flux (nmoles/m2-h)								
Core#	Site#	Mean		CI	r^2	n	Porewater S^{-2} (uM)	Bottom-water S^{-2} (nM)	Conc. Grad. Gradient (nM)	Calculated **Diffusive** S^{-2} flux (nmoles/m2-h)	Site averaged **Measured** S^{-2} flux (nmoles/m2-h)
1	1	**778**	+/-	133	0.88	8					
3	1	**754**	+/-	48	0.98	8	1.16	2.65	1154	**208**	**419**
6	2	**838**	+/-	72	0.97	8					
7	2	**630**	+/-	36	0.98	8	1.53	3.25	1530	**275**	**424**
10	3	**594**	+/-	41	0.97	9					
11	3	**870**	+/-	32	0.99	9	0.51	1.75	510	**92**	**616**

* Diffusive flux was based on 2cm diffusion layer thickness and a diffusion coefficient='0.00001 cm^2 s^{-1} (Li and Gregory, 1974)

CI= 95% Confidence Interval