

Paleolimnological Analysis of the History of Metals Contamination in the Great Salt Lake, Utah

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*Dr. Katrina Moser Preparing the Glew Corer to
Collect Sediment in the Great Salt Lake (Gunnison Bay)*

Abstract

Three sediment cores from the Great Salt Lake were analyzed to determine the magnitude and timing of the deposition of 21 metal contaminants. In the main lake (Gilbert Bay) concentrations of copper, lead, zinc, cadmium, silver, molybdenum, tin, mercury and others began increasing in the sediments in the late 1800s or early 1900s and peaked in the 1950s. These increases were coincident with increases in mining and smelting activities for these metals in Utah. Contamination indices in the 1950s were 20-60 fold above background concentrations for silver, copper, lead and molybdenum, and <15-fold for most other metals. Since the 1950s, concentrations of most metals in the sediments have decreased 2-5 fold, coincident with decreases in mining and improved smelting technologies. Nevertheless concentrations for many metals in surficial sediments are still above acceptable criteria established for freshwater ecosystems. In contrast to most metals, concentrations of selenium and arsenic were stable or increasing slightly in the Gilbert Bay sediments. In a coring site located in Farmington Bay near an EPA Superfund Site discharge canal, concentrations of metals were high and showed no indication of decreasing in more recent sediments, but this core only provided a 50-year record of contamination. Surficial sediments from additional sites in the Great Salt Lake indicated that metals were more concentrated towards the southern end of the lake where the primary sources of contamination were located.

Introduction

Mining and industrial activities frequently release toxic metals directly to waterways (Moore et al. 1991) or indirectly via atmospheric contamination (Nriagu 1979; Nriagu et al. 1982). The Great Salt Lake of Utah is located close to both major mining and smelting activities, and is also bordered by a large metropolitan area with extensive industrialization with wastewater releases into river tributaries. Beginning in the late 1800s a variety of anthropogenic activities increased air and water pollution in the region (Lamborn and Peterson 1985). Mining for silver, lead and gold began in the region ca. 1863 (Varley et al. 1921; Mcphee 1977; Ege 2005) and the Bingham Canyon Mine, located in the Oquirrh Mountains (Fig. 1), opened in 1905. This mine was once the largest copper producer in the world, and over 16 million metric tons of copper have been produced from the open pit mine (Brix et al. 2006; Utah Geological Survey 2011b). Gold, silver and molybdenum have also been produced there in significant quantities as well as in other mines located in the Oquirrh Mountains, and in the Wasatch Range areas of Little Cottonwood Canyon and Park City. The smelter for the Bingham Canyon mine is currently located

on the shore of the Great Salt Lake (Fig. 2), but other smelters were located in the Salt Lake Valley, where ores from not only Utah, but much of the intermountain west were processed (Varley et al. 1921). The site of the former Murray smelter, located on the Jordan River 30 miles south of the Great Salt Lake, is now an EPA Superfund Site because of contamination of lead and arsenic that lasted from 1872 until 1949 (Keck and Lepori 2012). In addition to mining in the Oquirrh and Wasatch mountains, large strip mines to extract phosphate are located in the Bear River watershed in Idaho and Utah (Mars and Crowley 2003), and this river is the largest source of water for the Great Salt Lake. In addition to phosphorus, these mines release significant quantities of selenium (Hamilton and Buhl 2005). Industrial activities in the greater metropolitan Salt Lake City have included metallurgical plants, railroad maintenance, petroleum refining and other

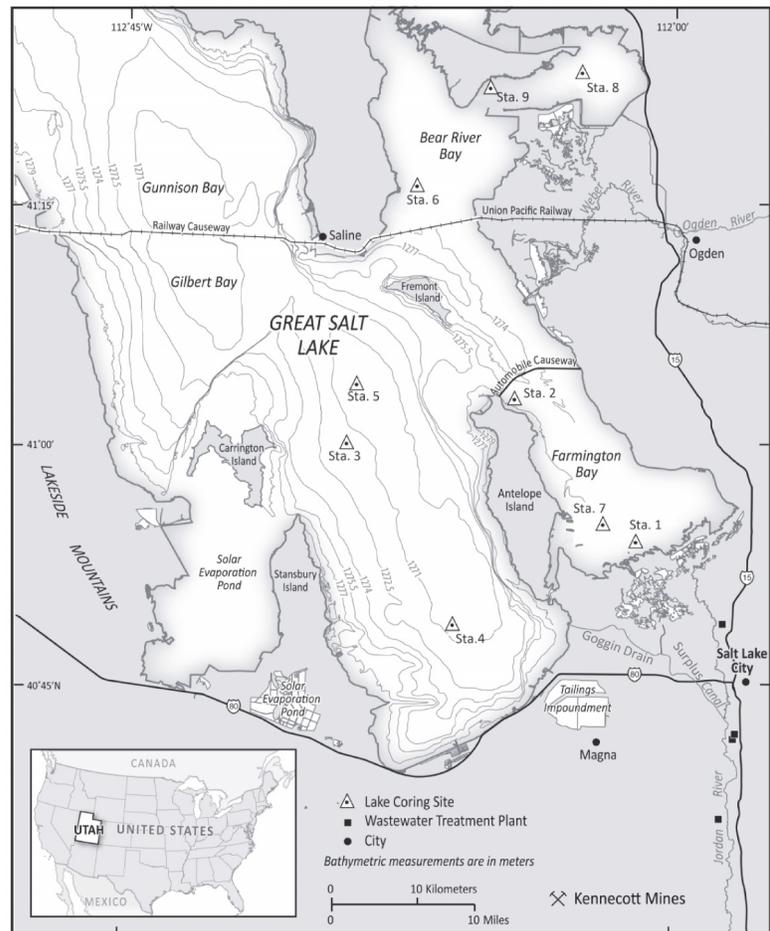


Figure 1. Map of the Great Salt Lake showing the coring sites, causeways, and small (\bullet , $<50 \text{ L sec}^{-1}$) and large (\bullet , $>100 \text{ L}^{-1} \text{ sec}^{-1}$) wastewater treatment plant locations. The location of the Bingham Canyon porphyry copper mine (Kennecott Mines) is also shown.

manufacturing that have also released metals into the Great Salt Lake watershed and air shed (The Forrester Group 2001).

The accumulation of metals near the Salt Lake area has been measured previously utilizing sediment cores retrieved from lakes that record the chronology of deposition. Kada et al. (1994) indicated that metal contaminants from river inputs likely contaminated Echo and Deer Creek Reservoirs located just east of the Salt Lake Valley. More recently, a network of sediment records showed metal contamination in several Salt Lake Valley sites (Naftz et al., 2000). More remote sites have also been



Figure 2. Aerial view of the Kennecott Corporation (Rio Tinto) smelter (lower right) and its tailings pond (upper left), both located on the south shore of Gilbert Bay, Great Salt Lake. Salt Lake City is in the background. Photo 9 March, 2012.

affected by metal pollution from the Salt Lake Valley. For example, remote Mirror Lake, located in the Uinta Mountains 100 km from the Salt Lake Valley, received atmospheric inputs of metals beginning in the early 1900s (Kada et al., 1994). Additional paleolimnological research in these mountains demonstrated unequivocally that the source of the metals to these alpine lakes was from atmospheric dust originating from smelters processing the ores of the Oquirrh and Wasatch Mountains (Moser et al. 2010; Reynolds et al. 2010). Although smelting activities are frequently associated not only with metals deposition, but also the acidification of lakes and streams, this has not been a major problem for these lakes. For lakes closest to the Salt Lake Valley this is probably related to the underlying bedrock, which is comprised of limestone and dolomites that buffer acidification (Ellis 1986). In the Uinta Mountains the underlying quartzite bedrock offers little buffering capacity, but it is believed that aeolian alkaline dust from the west desert of Utah reaches these watersheds and provides the needed buffering (Ellis 1986).

Despite this previous research, analyses of metal contamination in the Great Salt Lake have been limited because bioaccumulation in fish has not been a concern in this hypersaline system that only has abundant fish in its estuaries. However, in recent years concern has increased that bioaccumulation of metals through the food chain could affect the large populations of migratory birds that utilize the high productivity of the lake to fuel their migrations (Aldrich and Paul 2002; Cavitt 2007; Vest et al. 2009; Cline et al. In prep.). In a preliminary analysis, Waddell et al. (2002) found relatively high levels of selenium and other metals in the brine shrimp and brine flies that are the dominant food sources for migratory birds (Aldrich and Paul 2002; Wurtsbaugh 2009; Roberts 2013). The finding of high selenium levels, combined with a permit application by the Kennecott Copper Corporation to discharge additional selenium into the lake led to an extensive examination of this contaminant in the ecosystem (UDWQ 2008). Naftz et al. (2008) documented excessively high levels of total and methyl mercury in parts of the

Great Salt Lake, and this was followed by a coordinated study to assess sources and bioaccumulation of this toxin in the system (Naftz et al. 2009a; UDEQ 2011; Wurtsbaugh et al. 2011). Finally, an extensive analysis of many metals and other toxins in the Great Salt Lake and its fringing wetlands was recently completed that documented metals concentrations that were frequently above Threshold Effects Concentrations (Waddell et al. 2009), which could have detrimental effects to freshwater organisms (Macdonald et al. 2000).

To help understand the degree of metals contamination in the Great Salt Lake, and to evaluate the chronology of this contamination relative to mining history and industrialization, we analyzed two sediment cores from the south arm of the lake (Gilbert Bay), and one from the semi-enclosed Farmington Bay, which also receives significant loading of metals and nutrients from greater metropolitan Salt Lake City. The sediment cores allowed us to evaluate 150-200 years of metals deposition in the ecosystem. Additionally, surface sediments were analyzed from five other sites in the lake to assess the spatial extent of contamination. The results for Gilbert Bay show that the deposition of most metals began in the late 1800s and peaked at high levels in the mid-1900s, and have since declined. Selenium and arsenic, however, continue to increase in the sediment record, as have most metals in Farmington Bay.

Study Site and Methods

Study Site—The Great Salt Lake (Fig. 1) is a 5200 km² closed-basin system in Utah, USA (41.04 N, 112.28 W) bordered on its eastern and southeastern shores by the Salt Lake City metropolitan area with over a million residents. The lake has been impacted by industrial and municipal discharges, as well as by transportation causeways that divide the system into four large bays (Fig. 1). The main body of the lake was divided in two by a railway causeway constructed in 1959. Culverts and a breach allow exchange of water and salts. However, salinity in the northern arm of the lake, Gunnison Bay (2520 km²) is normally at saturation (ca. 330 g L⁻¹). The southern arm of the lake, Gilbert Bay, covers approximately 2400 km², but lake area, depth, volume and salinity vary greatly with precipitation cycles. Since 1850, surface salinities in Gilbert Bay have varied from 60 to ~330 g L⁻¹. At the mean lake elevation of 1280 m, the mean and maximum depths of Gilbert Bay are 4.4 and 10.1 m (Baskin 2005). Gilbert Bay, however, receives an underflow of saturated brine from Gunnison Bay that passes through the causeway's culverts and fill material, so that ~50% of the southern bay is underlain by a deep brine layer (monimolimnion) that is anoxic and has high concentrations of hydrogen sulfide (Gwynn 2002a; Jones and Wurtsbaugh 2014). Stephens (1990) provides a historical review of the effects of salinity changes in Gilbert Bay to the lake's biota.

The two bays on the eastern side of the lake receive freshwater inflows from the Wasatch and Uinta mountains, and can be characterized as estuaries with salinities that vary from freshwater to hypersaline conditions, both spatially, and on seasonal and decadal time scales. Farmington Bay (~310 km²) in the SE is shallow, with a mean depth of only 1.1 m at mean lake elevation (EPA 2010). It receives flows from the Jordan River that passes through Salt Lake City and wetlands before reaching the lake and by smaller creeks on its eastern side of the bay. At lake elevations above approximately 4201 m Farmington Bay was joined on the north and south to Gilbert Bay, but at lower lake levels Antelope Island was contiguous with the mainland, thus partially isolating the bay (Fig. 1). In 1952 an automobile causeway was constructed to the southern tip of Antelope Island, thus reducing interchange with Gilbert Bay even when lake levels were above 4201 m. In 1969 another automobile causeway was constructed to

the northern tip of Antelope Island with only two small culverts and a bridge that allows water to exchange between Farmington and Gilbert Bays. Both automobile causeways were flooded in the mid-1980's when the lake reached an historic high level of 4211 m. The overall influence of the two automobile causeways, however, has been to make Farmington Bay fresher, with salinities ranging spatially and temporally from $<1 \text{ g L}^{-1}$ up to 90 g L^{-1} (Wurtsbaugh and Marcarelli 2006). A salt wedge intrudes from Gilbert Bay so that the northern half of Farmington Bay is underlain with an anoxic, hydrogen-sulfide rich deep brine layer below a depth of $\sim 1 \text{ m}$ (Bioeconomics 2012). Bear River Bay in the NE has an area of 212 km^2 and a mean depth of only 0.6 m at mean lake elevation (EPA 2010). In the late 1960s interchange between Bear River Bay and Gilbert Bay was constricted by construction of salt ponds and a bridge by the Great Salt Lake Minerals Corporation (Behrens 1980). Bear River Bay receives flows from the lake's largest tributary, the Bear River, and thus has fresh water during spring runoff, but salinities can climb to over 250 g L^{-1} when the bay dries and concentrates salts that have entered the bay from the adjoining salt ponds and/or Gilbert Bay (Tiffany et al. 2007a).

In addition to the causeways that have modified the lake's circulation and salinities, other changes have occurred since the settlement of Salt Lake City by Mormon pioneers in 1847 (Table 1). The human population and accompanying industrial development on the shores of the lake have increased rapidly: 8,000 in 1850; 143,000 in 1900; 389,000 in 1950, and; 1,570,000 in 2010 (Counties of Salt Lake, Davis and Weber; Tiffany et al. 2007b)). To reduce flooding during spring runoff, the Surplus Canal was constructed in 1885 to divert water from the Jordan River from a point south of Salt Lake City and into the wetlands bordering the southern edge of Farmington Bay (Fig. 1). In 1952 the Surplus Canal was joined with the Goggin Drain so that flood waters could be diverted into Gilbert Bay (Deseret News 1952). An analysis of flows from 1975 to 1984 suggests that an average of 36% of the Jordan River's flow is diverted into Gilbert Bay via the Goggin Drain (CH2MHill 2012). The majority of these flows to Gilbert Bay are during spring runoff when wastewaters are diluted. In 1889, the first sewer system in Salt Lake City began discharging raw sewage into the Jordan River, but by 1911 conditions in the river were so bad that a Sewer Canal was constructed to bypass the Jordan River and discharge sewage directly to Farmington Bay (Hooton no date). In 1922, the Northwest Oil Drain was connected to the Sewer Canal to discharge industrial wastes from refineries and other industries in the NE part of Salt Lake City (Fig. 3). Because of the metals and other contaminants that have accumulated in the NW



Figure 3. Northwest Oil Drain and refineries in northern Salt Lake City. The Oil Drain joins the Salt Lake City Sewer Canal and enters the lake at the south end of Farmington Bay. The State Capitol and high-rise buildings of Salt Lake City are visible in the upper right. The Wasatch Mountains are in the background. Photo 9 March 2012.

Oil Drain, it is now an EPA Superfund Cleanup Site (The Forrester Group 2001). In addition to the industrial activities of the growing city, the mining activities in the immediate region have contributed to high metals loading to the lake. Water quality conditions in

| Year | Event |
|---------------------|---|
| 1847 | Mormon pioneers settle Salt Lake Valley. |
| 1863 | Copper mining begins at Bingham Mine; intensifies in 1873. |
| 1873 | Lake reaches high level (4211' ; 1283.5 m), but salinity only decreases to $\sim 136 \text{ g L}^{-1}$. |
| 1885 | Surplus Canal constructed that diverts flood flows of Jordan River to wetland at the south end of Farmington Bay |
| 1889 | First sewer line in SLC to the Jordan River; by 1908 flow was $0.60 \text{ m}^3 \text{ sec}^{-1}$ ($21 \text{ ft}^3 \text{ sec}^{-1}$). |
| 1892 | First smelter for gold, silver and lead. |
| 1906 | Agriculturists win lawsuit against four Salt Lake Valley smelters, forcing them to reduce emissions of sulfuric acid and arsenic. |
| 1911 | Outlet Sewage Canal completed to the Great Salt Lake (Farmington Bay). Flow discontinued into the Jordan River. |
| 1922 | The current Oil Drain - Sewage Canal to the Great Salt Lake completed. |
| 1930 | Farmington Bay Waterfowl Management Area constructed. Diking likely helps retain Jordan River nutrients and sediments in wetlands. |
| 1952 | Goggin Drain connected to Surplus Canal so that spring floods of Jordan River can be diverted to Gilbert Bay. Record flood in Jordan River. |
| 1952 | South causeway to Antelope Island constructed to prevent raw sewage from reaching swimming beaches at south end of Gilbert Bay when high lake levels flooded the sill to the Island. Sewage would have been "treated" longer in Farmington before reaching Gilbert. |
| 1959 | Railroad Causeway completed that separates Gilbert and Gunnison Bay. Causes salinity to decrease in Gilbert, but deep brine layer begins to form. |
| 1959 to 1962 | Three sewer districts formed in Davis Co. to discharge wastewater into Farmington Bay. Combined flow ca. $265 \times 10^3 \text{ m}^3 \text{ day}^{-1}$ (70 million gallons per day). |
| 1963 | Lake reaches lowest recorded level (1277.8 m; 4192.2'). Deep brine layer in Gilbert likely disappeared. South end of Farmington Bay would have been just river/wastewater flow (or dry, depending on where river channel went.) |
| 1965 | Secondary treatment facility completed in Salt Lake City; discharge of raw sewage stopped. |
| 1969 | Automobile causeway to Antelope Island completed, partially isolating Farmington Bay. Top elevation 1282.1 m (4206.5 ft). |
| 1970 | Clean Air Act of US; 1977 significant increases in regulations. |
| 1984 | Automobile causeway to Antelope Island flooded by high water until mid-1989. |
| 1985 | Gilbert Bay reaches record high level (4208.85'; 1282.86 m); Salinity declines to 58 g L^{-1} . |
| 1992 | Automobile causeway rebuilt. |

Table 1. Dates of important events that have influenced contaminants in the Great Salt Lake, Utah.

Farmington Bay, which received the majority of the wastewater discharges from Salt Lake City, presumably degraded rapidly, although documentation of this is poor. Gwynn (2002b) provides a history of raw and treated sewage discharge into Farmington Bay and reviews the earlier studies. He notes that the sewage discharge resulted in what Van der Meide and Nicholes (1972) called a “sewage delta” near the outfall of the Sewer Canal. Raw sewage was reaching swimming beaches at the south end of Gilbert Bay (Saltair) when the lake was high enough to permit water exchange at the south end of Antelope Island. In 1952 an elevated roadway was constructed to the southern tip of the island, in part to keep raw sewage from reaching the swimming beaches at SaltAir on the south end of Gilbert Bay (Gwynn 2002b).

Recent studies have shown that water quality conditions in the bays of the Great Salt Lake differ markedly (Wurtsbaugh and Marcarelli 2006; Wurtsbaugh et al. 2008). Farmington Bay is hypereutrophic with phosphorus loading ca. $3 \text{ g m}^{-2} \text{ y}^{-1}$ (Marcarelli and Wurtsbaugh 2005), large blooms of a nitrogen-fixing cyanobacteria, *Nodularia spumigena*, and mean chlorophyll *a* concentrations of $141 \mu\text{g L}^{-1}$ (Wurtsbaugh et al. 2012). Oxygen concentrations range from supersaturated during the day to anoxia throughout the water column at night. pH varies with the diel swings in photosynthesis with values ranging from ca. 8.6-9.5 (Tiffany et al. 2007a; Wurtsbaugh et al. 2012). Temperatures in the shallow bay reach 30°C during the daytime in summer, and the bay is ice covered in the winter. Strong temporal and spatial salinity gradients ($1\text{-}90 \text{ g L}^{-1}$), however, cause blooms of *Nodularia* to vary greatly. Depending on salinities, phytoplankton in Farmington Bay can be limited by either nitrogen or phosphorus (Marcarelli et al. 2006). Although all sewage now entering the bay has had secondary treatment, some evidence of human pathogens is present in the bay’s sediments (Gast et al. 2011). Sorensen et al. (1988) also found hypereutrophic conditions and a dominance by *Nodularia* in 1986 when the lake was at a high level and the Antelope Island Causeway was flooded. Naftz et al. (In preparation) found total and methyl mercury concentration in Farmington Bay of 8 and 2 ng L^{-1} , respectively. Total selenium concentrations were near $0.5 \mu\text{g L}^{-1}$ (Naftz et al. 2009b).

Gilbert Bay, with recent salinities between 100 and 160 g L^{-1} is much less productive than Farmington Bay. The brine is composed of 55% Cl, 28% Na, 8% SO_4 , 4% Mg and 5% other salts (Sturm 1980). Oxygen concentrations are usually near saturation in the mixed layer, but are zero in the deep brine layer below $\sim 6.5 \text{ m}$. Temperatures reach 24°C in summer and can decline to 0°C or below in winter. The pH is relatively stable around 8.1 in the mixed layer (Wurtsbaugh and Marcarelli 2006). This bay is dominated by green algae and abundant populations of *Artemia* and *Ephydra* (Felix and Rushforth 1979; Wurtsbaugh and Marcarelli 2006; Wurtsbaugh et al. 2011). Chlorophyll *a* concentrations vary seasonally, with highs near $70 \mu\text{g L}^{-1}$ in winter, to ca. $1 \mu\text{g L}^{-1}$ in summer when grazing by *Artemia* depresses the phytoplankton populations. Phytoplankton growth is limited by nitrogen when salinities are $>70 \text{ g L}^{-1}$ because cyanobacteria cannot fix atmospheric nitrogen at those levels (Stephens and Gillespie 1976; Wurtsbaugh 1988; Marcarelli et al. 2006). Wurtsbaugh et al. (2008) estimated that nitrogen loading to Gilbert Bay was high ($>2.4 \text{ g m}^{-2} \text{ yr}^{-1}$), with approximately 45% contributed by the outflow from Farmington Bay. Total mercury and selenium concentrations in the mixed layer of Gilbert Bay are near 5 ng L^{-1} and $0.5 \mu\text{g L}^{-1}$, respectively, but much higher mercury concentrations occur in the deep brine layer (Naftz et al. 2008; Jones et al. 2009; Wurtsbaugh et al. 2011; Jones and Wurtsbaugh 2014). Less information is available concerning water quality in Bear River Bay, but mean chlorophyll levels are near $22 \mu\text{g L}^{-1}$ (Wurtsbaugh et al. 2012). Total selenium concentrations are ca. $0.5 \mu\text{g L}^{-1}$ in the bay and total mercury concentrations in an adjacent diked wetland are ca. 2 ng L^{-1} (Naftz et al. 2009b; Naftz et al. In preparation). Water quality conditions in Gunnison Bay, with salinities

that are usually near saturation and a near absence of metazoan organisms, has been studied less than the other bays, with a focus on microbial diversity and function (Post 1981; Parnell et al. 2011).

Table 2. Coring site locations and characteristics.

| Site Characteristics | Bay and Site Number | | | | | | | |
|--------------------------------------|----------------------------|----------------------------|----------------------------|----------------------------|----------------------------|----------------------------|----------------------------|----------------------------|
| | Farmington Bay | | | Gilbert Bay | | | Bear River | |
| | 1 | 2 | 7 | 3 | 4 | 5 | 6 | 8 |
| Coordinates | 40.91425 N -112.04996 W | 41.05844 N -112.21719 W | 40.93177 N -112.09557 W | 41.00905 N -112.43955 W | 40.82507 N -112.29711 W | 41.07202 N -112.42842 W | 41.27657 N -112.35191 W | 41.39331 N -112.12797 W |
| Collection Date | 17-Aug-09 | 17-Aug-09 | 25-Jun-10 | 18-Aug-09 | 19-Aug-09 | 20-Aug-09 | 21-Aug-09 | 8-Jul-10 |
| Station Depth (m) | 0.15 | 1.01 | 0.68 | 7.35 | 7.8 | 7.94 | 0.12 | 0.7 |
| Bottom Salinity (g L ⁻¹) | 1 | 80 | 4 | 147 | 147 | 147 | 145 | 2 |
| Secchi Depth (m) | >>0.15 | 0.16 | 0.38 | 0.7 | 3.8 | 2.15 | 0.5 (deeper site) | >0.7 |

Coring, Sectioning & Dating—Coring and sectioning were described in detail by Leavitt et al. (1980). Briefly, cores were collected with a Kajak-Brinkman gravity corer (Glew et al. 2001) with a 63-mm diameter tube at 10 sites in three of the bays of the Great Salt Lake on 17-21 August 2009 and 8-9 July 2010. Site coordinates and depths at the time of collection are given in Table 2. The cores were either processed in the field (2009) or transported upright to the laboratory (2010) and sectioned in 0.5-cm intervals with a Glew push rod extruder (Glew et al., 2001).

The locations of the nine coring sites are shown in Fig.1. Two of the successfully-dated cores were in Gilbert Bay (Site 3, 4) at depths > 7 m where an anoxic deep brine layer (monimolimnion) was present. The other dated core was at the south end of Farmington Bay. At low water levels this site is in a broad channel running E-W that is partially disconnected from the main bay (Fig. 4). At the time of coring Site 1 was only 0.15 m deep and ca. 1 km from the discharge point of the Sewer Canal/NW Oil Drain, but at mean lake elevations it would have been ca. 6 km from the outfall. At low lake levels, Site 1 might be better characterized as being part of the former Jordan River delta, whereas at mean lake elevation (1280 m; 4200 ft.) water depth would have been ca. 2 m and it would have mixed well with the main Farmington Bay. Site 1 is near



Figure 4. Aerial photograph showing the SE end of Farmington Bay and the location of the Site 1 core. At higher lake levels the sand sill to the north of this site would be inundated and the water would mix more with the rest of the bay. The Site 1 core was taken when Gilbert Bay was at a lake elevation of 1,278.6 m (4,195 ft.). The image was taken 21 April 2008 when Gilbert Bay was at an elevation of 1279 m. Photo by W. Wurtsbaugh.

the fresh water inflows from the Central Davis Sewer District, the Salt Lake City Sewer Canal, and the Jordan River after it has passed through the wetlands at the south end of the bay (Figs. 1, 4). These inflows create strong salinity gradients along the N-S axis of the bay with nearly fresh water near Site 1, and salinities of 20-90 g L⁻¹ at the northern end after the construction of the Automobile Causeway (Hayes 1971; Wurtsbaugh and Marcarelli 2006). The strong salinity gradient results in very different plankton communities from north to south (Hayes 1971; Wurtsbaugh and Marcarelli 2006). Consequently, the results for Site 1 should not be interpreted as being characteristic of the entire bay.

²¹⁰Pb dating of the cores is described in detail by Leavitt et al. (1980). Briefly, 12-14 core sections were freeze-dried and then counted either at the University of Regina Environmental Quality Analysis Laboratory (Sites 1-3, 6) and University of Waterloo (Sites 4, 5) using gamma spectrometry and an Ortec High-Purity Germanium (HPGe) Coaxial Well Photon Detector System. To validate the ²¹⁰Pb dates ¹³⁷Cs activity was measured to identify the period of maximum fallout from atmospheric nuclear weapons testing (1963-65). Sediment age-depth relations were calculated using the CRS (constant rate of supply) model (Appleby and Oldfield 1983). Bulk sediment accumulation rates (g cm⁻² yr⁻¹) were computed from output of the CRS model (Appleby and Oldfield 1983) and represent the mass of dried material (sediments + salt) in each 0.5 cm interval (g cm⁻²) divided by the time represented in the interval (yr.). Dates earlier than ~1875 CE (Common Era, formerly AD) were approximated by extrapolation of mass accumulation rates.

The dating chronologies for the three cores are shown in Fig. 5 and a detailed description of these results is available in Leavitt et al. (2012). The Site 4 core in Gilbert Bay yielded a confirmatory cesium peak in the mid-1960s. In the Site 3 core the expected mid-1960s ¹³⁷Cs peak occurred between 1928-1942 (based on the ²¹⁰Pb depth estimate), making the chronology at this site less dependable. Consequently, we have relied more on the Site 4 core for interpretation of the chronology in Gilbert Bay. Nevertheless, the patterns seen at Sites 3 and 4 were relatively similar (see Results). The Site 1 core also had a cesium peak in the mid-1960s, but the peak was distorted in the surface sediments due to either sediment mixing or post-depositional migration of ¹³⁷Cs (see Leavitt et al. 2012). Of the ten cores collected, seven did not yield useful ²¹⁰Pb chronologies, likely because in many of the shallow sites wind mixing has homogenized the sediments so that useful temporal resolution was lost. Nevertheless, we analyzed the upper 0-1 cm of sediments from five of these sites to provide an estimate of the spatial variation across Great Salt Lake.

Analytical Methods for Metals—Sediment subsamples from each core slice were oven-dried at 70°C until constant weight was obtained. Metal concentrations in the sediment samples were measured at the Utah Veterinary Diagnostic Laboratory (<http://www.usu.edu/uvdl/>) in Logan, Utah. The nitric acid leachable mineral concentrations in each slice were quantified using inductively coupled plasma mass spectroscopy (ICP-MS). Test materials were digested in screw-cap Teflon tubes on a heat block at 90°C for 4 hours. Weighed sediment samples (~0.5 g) were digested in 10 ml trace mineral grade nitric acid. The digests were diluted 1:20 with 18.2 MOhm ultrapure water, to provide a 5% nitric acid matrix prior to analysis. This resulted in a matrix match to the standards and quality control samples. Samples that had higher mineral content than the high standard were diluted 1:10 in 5% nitric acid and reanalyzed. Standard curves for all elements, except mercury, consisted of five concentrations between 10 and 2500 µg l⁻¹. Standard curves for mercury consisted of three concentrations from 2.5 to 10 µg l⁻¹. A quality control (QC) test sample was analyzed with every five samples to validate analytical accuracy. The QC

sample had to be +/- 5% of the known mineral specifications to pass. Any group of samples that had a failed QC test was re-analyzed. A small number of samples for silver yielded values below detection. For these we assumed that concentrations were 80% of the detection limit because adjoining samples in the cores had values just slightly over the detection limit.

The metals concentration in each core slice were salt-corrected, to account for the dissolved salts in the interstitial water. Consequently, most of the data are presented on a per-gram of dry sediment basis. The salt correction was particularly important in the top 3-6 cm of the Gilbert Bay cores which had unconsolidated sediment and consequently contained considerable amounts of water and salt. The water content was measured by weight loss during freeze-drying of the samples used for ^{210}Pb analyses. Salt content was then estimated by multiplying this amount of water by the salt content of the overlying water (Table 2) using the assumption that diffusion of salts would bring the pore water into equilibrium with the overlying water. In the surface 3-4 cm of the Gilbert Bay cores, the salt in the interstitial water accounted for about 50% of the dry weight of material, so that the salt correction approximately doubled the estimate of metals in these samples. Mean metal concentrations prior to 1860 were assumed to represent background levels, and were used to calculate contamination indices for metals in more recent strata.

Historical Production of Metals in Utah—The annual production of many commercial metals in the State of Utah were obtained from the Utah Geological Survey (2011a). Most of these minerals were processed in Salt Lake Valley or the Tintic/Mercur mining districts that are within 40 km of the Great Salt Lake (Ege 2005). Although small amounts of mercury have been mined in Utah, mercury residues in the lake may have originated initially from imported mercury used to extract gold and silver with the mercury amalgamation process, and subsequently from mercury released in the smelting polymetallic replacement deposits containing lead and zinc (Plumlee et al. 1996), as well as from other sources of contamination. Consequently, as a metric for mercury use and production, we have used the annual extraction of lead in the State. This is not ideal, however, because of the multiple potential sources of mercury contamination.

Indices and Criteria—Following the approach of (Hokanson 1980), Belzile et al. (2004) and others, we have calculated indices that allow chronologies of multiple metal concentrations to be easily compared. A *Contamination Index* throughout a core was calculated as the ratio of the concentration of a metal relative to the mean concentration in pre-industrial sediments (before 1860). A *Maximum Contamination Index* was calculated at the maximum concentration encountered in a core relative to the pre-industrial mean concentration. A *Current Contamination Index* was calculated as the ratio of the mean concentration in recent sediments (top 1-cm) relative to the mean background concentration observed in the core prior to 1860. Because of the unusual chemistry and biota in many parts of the Great Salt Lake, site-specific criteria have not been established for assessing the degree of impact a particular metal might have. Nevertheless, to provide some context for understanding the concentrations of metals in the Great Salt Lake sediments we have used freshwater sediment Threshold Effects Concentrations (TEC) where some impacts on biota may be possible, and Probable Effects Concentrations (PEC) where impacts are likely (MacDonald et al. 2000).

Although these metrics are useful, they do not necessarily mean those criteria can be directly applied to the Great Salt Lake ecosystem, especially in Gilbert Bay where salinities are very high and tolerant salt-adapted biota are present.

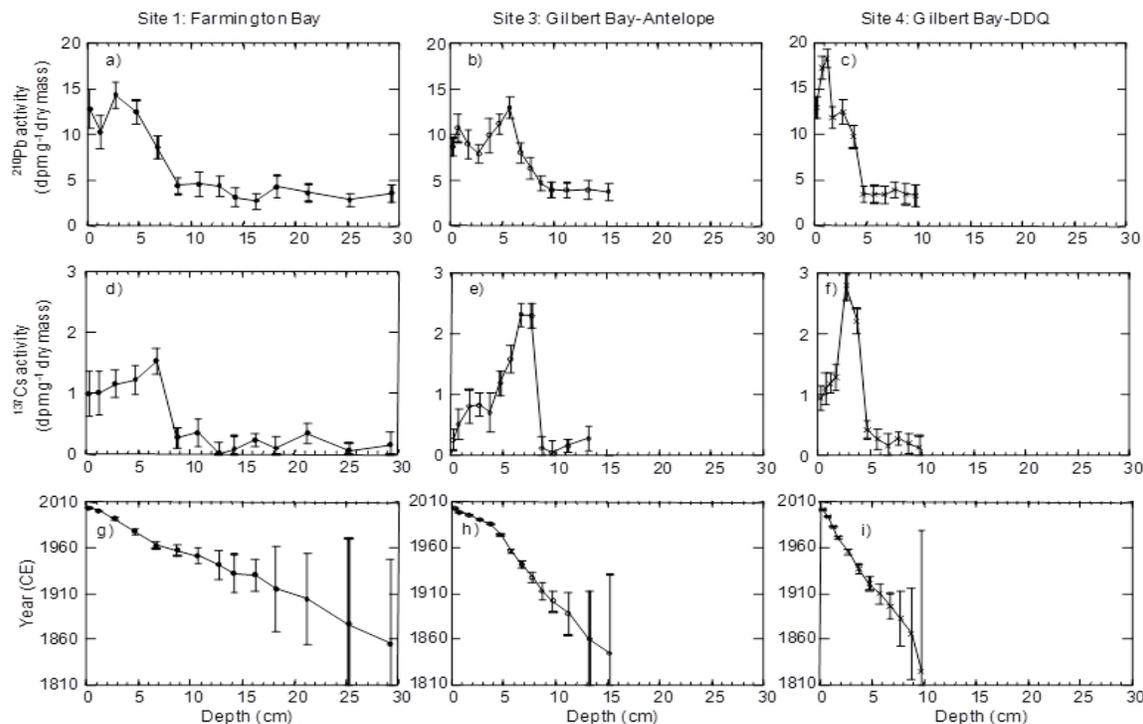


Figure 5. Dating chronologies for Site 1 in Farmington Bay and Sites 3 and 4 in Gilbert Bay. The top row shows ^{210}Pb lead activities (disintegrations per minute, dpm) uncorrected for background (supported) activity. The middle row shows ^{137}Cs cesium activity and the last row shows estimated dates and s.d. error bars based on variances in ^{210}Pb counts. From Leavitt et al. 2012.

Results

Historical changes in metals concentrations

Gilbert Bay—Analyses of both cores in Gilbert Bay indicated that many metals began increasing in the sediment record in the late 1800s and early 1900s coincident with mining and smelting activities, peaked in the mid-1950s and have since declined (Fig. 6, Appendix 1a, 1b). For example, at both sites copper concentrations rose steadily from background levels of $11 \mu\text{g g}^{-1}$ before the onset of commercial mining to concentrations of $500 \mu\text{g g}^{-1}$ when copper extraction reached 250,000 metric tons per year in the 1950s. Since the 1950-1960s, concentrations of copper in the sediment record have decreased markedly: however, in surficial sediments they are still 10-20 times higher than background levels. These declines have occurred despite an overall increase in copper production in the past two decades.

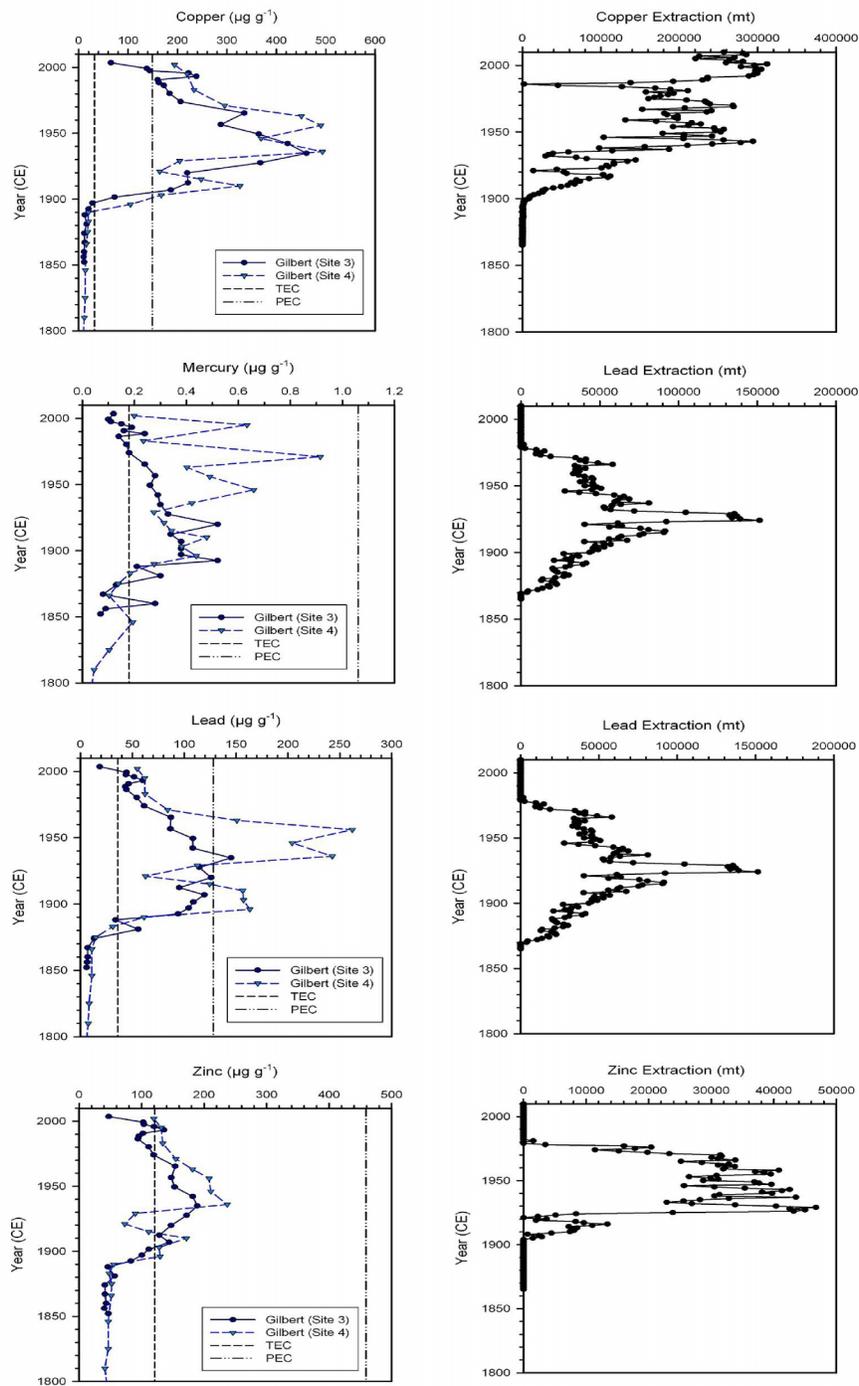


Figure 6. Left frames: Concentrations of Cu, Hg, Pb and Zn in sediments of different ages from the two coring sites in Gilbert Bay. Concentrations are salt-corrected. Dotted vertical lines show Threshold Effects Concentrations (TEC; ---) and Probable Effects Concentrations (PEC; - - -) of sediments in fresh water ecosystems. Right frames: Metal production in Utah derived from Utah Geological Survey (2011b). Note that although zinc was mined along with lead, it was not valued as a commodity until the early 1900s (K. Krahelec personal communication).

was more variable. Background mercury concentrations were near $0.1 \mu\text{g g}^{-1}$. When lead production increased sharply in the 1870s and 1880s, mercury concentrations began to increase (Fig. 6). Mercury concentrations in the sediments from Site 3 peaked at $0.52 \mu\text{g g}^{-1}$ in about 1920, and have declined significantly to levels near $0.12 \mu\text{g g}^{-1}$ in the surface sediments. At Site 4 the overall trend is similar, but with a much more irregular decline over the last half century.

Lead and zinc concentrations had similar chronologies in the two Gilbert Bay cores (Fig. 6). Both had low background concentrations of 7 (Pb) and 45 (Zn) $\mu\text{g g}^{-1}$, but concentrations increased in the late 1800s and early 1900s, coinciding with the onset of lead and zinc production. Concentrations of lead and zinc in the sediments peaked in the early 1930s (Site 3) or mid-1940s (Site 4) at levels far above background and subsequently declined. As with mercury, the sediment record for lead and zinc concentrations were more variable at Site 4 than at Site 3. Although lead and zinc production in Utah has declined to zero in Utah, and although sediment concentrations have declined substantially, sediment concentrations remain well above background at both sites.

In contrast to most metals, concentrations of selenium and arsenic have been stable or are increasing in the sediments of Gilbert Bay (Fig. 7). Background concentrations of selenium were near $1 \mu\text{g g}^{-1}$ at both sites. Concentrations did not increase significantly until the 1930s-1950s, and then have remained relatively constant or increased slightly, reaching concentrations of 4-6 $\mu\text{g g}^{-1}$ in recent sediments. Background levels of arsenic were near $13 \mu\text{g g}^{-1}$ and began increasing during the same period when selenium was increasing. Concentrations of arsenic in recent sediments reached 30-50 $\mu\text{g g}^{-1}$ at both sites.

An indicator of fluvial transport of metals, aluminum (Schindler 2012), did not change markedly in the cores (Appendix 2). This suggests that much of the deposition of other metals in the Great Salt Lake has been by atmospheric deposition rather than by riverine transport. Changes in iron and

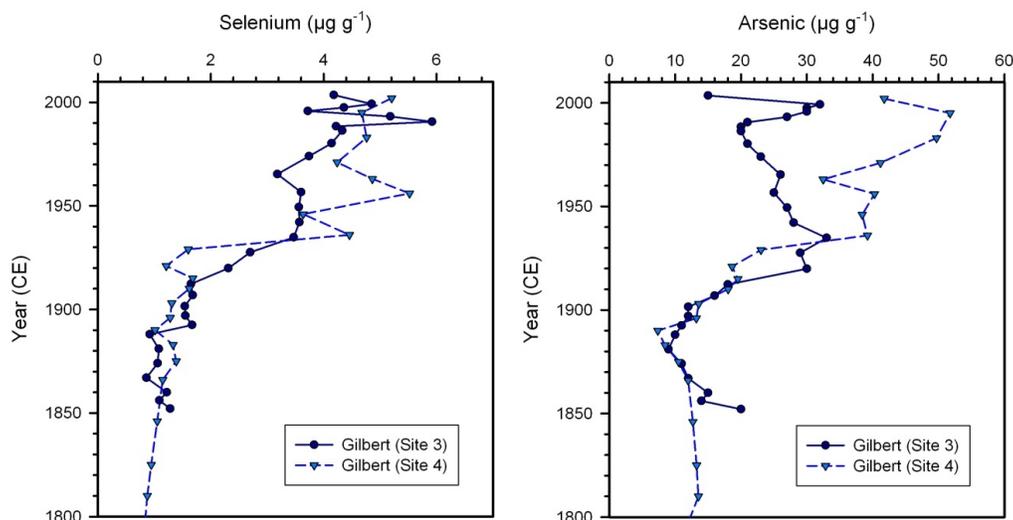


Figure 7. Concentrations of selenium and arsenic from two sediment cores in Gilbert Bay of the Great Salt Lake. Concentrations are expressed μg per gram of salt-corrected dry sediment.

aluminum concentrations at Site 4, however, did show some parallel changes with other metals after approximately 1910, but this pattern was not evident at the other two coring sites.

Farmington Bay (Site 1) - Our sedimentary record of metals in the Farmington Bay core (Fig. 8; Appendix 1c) differs considerably from the patterns seen in the two Gilbert Bay cores (cf. Fig. 6 & 8). The estimated chronology for the core samples from Site 1 in Farmington Bay suggests that metal concentrations were near baseline up until the early 1950s when there was a simultaneous and marked increase in most metals. For example, background copper concentrations in the core material were near $11 \mu\text{g g}^{-1}$ until the early 1950s, peaked in ~ 1960 at concentrations near $200 \mu\text{g g}^{-1}$ and then stabilized at concentrations near $150 \mu\text{g g}^{-1}$. Most metals (Ag, Cd, Cr, Cu, Hg, Mo, Pb, Sn, and Zn) showed this same pattern (Appendix 1c). Copper, mercury, lead, zinc and some other metals are now above threshold effects concentrations in the upper sediments at Site 1 (Fig. 8). The contrast between the abrupt pattern in Farmington Bay and that in Gilbert Bay is shown for copper in Fig. 8a. It is likely that the sediment record in Farmington Bay was not preserved prior to the 1950s, thus yielding an erroneous perception of pristine conditions in the bay in the first half of the century (see Discussion).

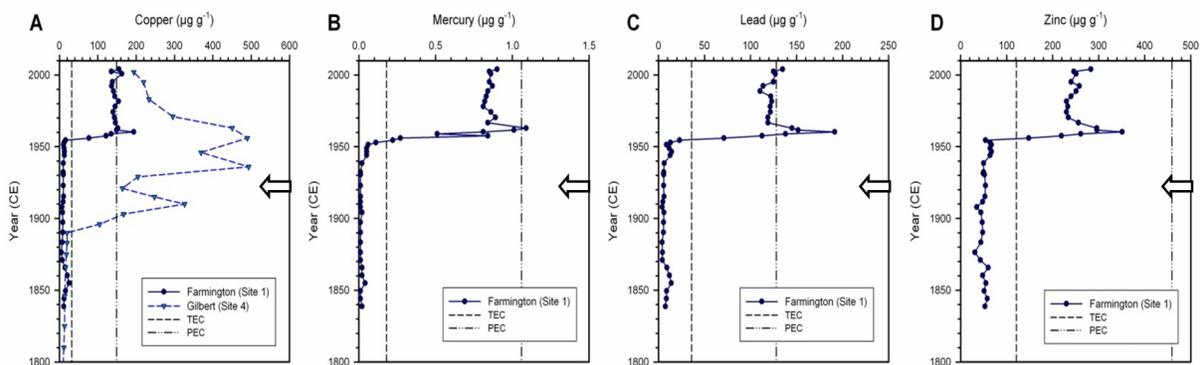


Figure 8. Concentrations of copper (A), mercury (B), lead (C) and zinc (D) in the sediments from a core at the south end of Farmington Bay. Frame A also shows the copper concentrations from Site 4 in Gilbert Bay. The arrow on the left shows the date of completion of the Northwest Oil Drain that began discharging industrial effluents into the bay via the Sewer Canal. Vertical dotted lines show Threshold Effects Concentrations (TEC) and Probable Effects Concentrations (PEC) for fresh water ecosystems. The timing of the onset of metal pollution in Farmington Bay is problematic (see Discussion).

Metal Accumulation Rate—Concentrations of metals in the lake sediments are partly a function of differing sedimentation rates that dilute the metals. Although there were variations through time, Site 4 in Gilbert Bay had the lowest sedimentation rate, Farmington Bay had an intermediate rate, and Site 3 in Gilbert had a somewhat higher rate (Fig. 9a). These rates, when combined with the measured concentrations in the sediments, show the micrograms of each metal that was accumulating per year under a square centimeter of lake bottom. Although sedimentation rates varied spatially and temporally in Great Salt Lake, the general patterns of metal accumulation reflect the trends in metal concentrations (Fig. 6). One notable exception is mercury. Mercury accumulation rates at the two sites in Gilbert Bay were relatively similar, with broad peaks rates around 1900. In contrast, mercury accumulated much faster in

Farmington Bay than in Gilbert Bay, at least after the mid-1950s (Fig. 9b). Copper accumulation rates in the two Gilbert sites were relatively similar throughout the data record, and after mid-1950s, the Farmington Bay sediments also accumulated this metal at about the same rate (Fig. 9c). Selenium accumulation rates were relatively similar in all three, and all three sites showed increasing rates, particularly in recent sediments (Fig. 9d).

Contamination and Current Indices—The Contamination Indices of the metals that show their concentrations relative to pre-industrial concentrations are shown in Fig. 10. These plots show that many of the trends are dependent on the mining districts, and presumably mining activity, where specific minerals were found. Gilbert Bay’s sedimentary concentrations of metals associated with the Bingham Canyon porphyry copper, molybdenum, gold and silver deposits and Mercur gold mining districts are shown in Fig. 10a. Copper, silver, and molybdenum all show similar concentration peaks in the 1950s that were 20 to 40-fold higher than background concentrations. Silver and copper also show a secondary peak in the relative abundances in the early 1900s.

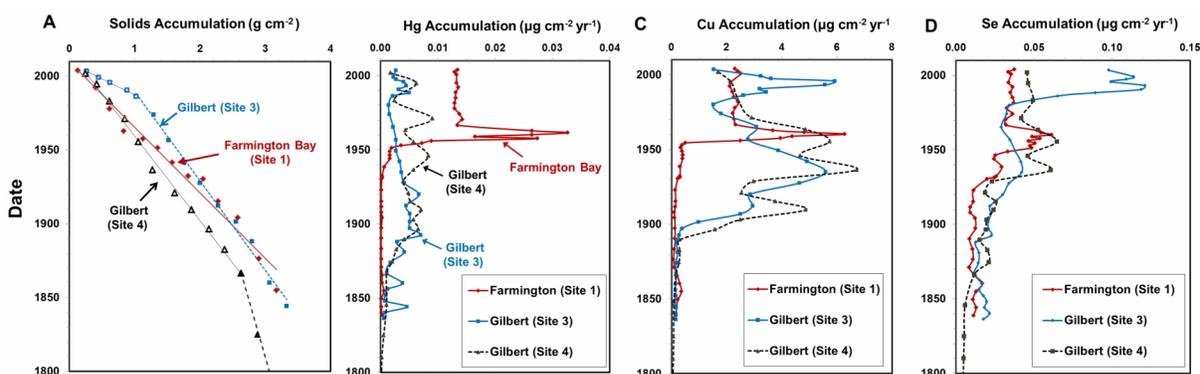


Figure 9. A. Estimated solids (sediment + salt) accumulation rates from two cores collected in Gilbert Bay and one in Farmington Bay. These were used to estimate rates of mercury (frame B), copper (C), and selenium (D) deposition in the sediments of Farmington Bay and Gilbert Bay (2 sites). For the Gilbert Bay sites solids accumulation rates were broken into two phases (Site 3: 1844-1986 and 1986-2003; Site 4: 1745-1866 and 1866-2002). For Farmington Bay, three phases were recognized (1855-1930, 1930-1963 and 1963-2004). The accumulation rates are dependent on accurate core dating.

The sediment concentrations of metals from the polymetallic (Pb-Zn-Cu-Ag-Au plus associated Cd, As, Sb and Hg) replacement deposits of the Tintic, Park City and Bingham mining districts also showed a similar temporal congruence in their relative abundances in the sediment cores (Fig. 10b). Lead and cadmium had the highest relative increases with peaks about 30 times background levels in the late 1940s. The other metals also peaked at the same time, with tin showing a 14-fold increase whereas zinc increased only 5X above background levels. Although the temporal pattern for mercury closely followed those for the other from this district, it only increased 9-fold above background levels. As with the porphyry and sedimentary ores from Bingham Canyon and Mercur, the concentrations of metals from the polymetallic replacement deposits have also decreased considerably in the lake sediments since the peak levels in the 1940s and 1950s.

In contrast to the metals shown in Figs. 10a and 10b, the semi-metals, selenium and arsenic, showed a different concentration pattern in the sediments (Fig. 10c). The origin of these metals is more difficult to characterize. Selenium and arsenic began to increase in the early 1900s and reached moderate levels in the early 1950s. Although the concentration peaks for selenium and arsenic were not particularly high relative to some of the other metals, they are stable or increasing slightly in the recent sediments.

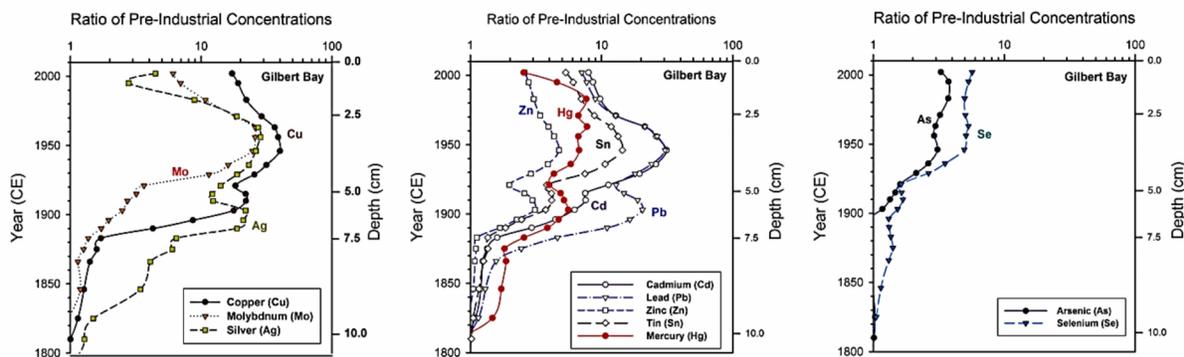


Figure 10. Contamination indices of metal concentrations at each depth in a Gilbert Bay core (Site 4) relative to mean concentrations prior to 1860. A ratio of 1 indicates background, uncontaminated conditions. A. Metals derived primary from porphyry copper and sedimentary rock-hosted gold and silver deposits in the Bingham Canyon and Mercur mining districts of Utah. B. Ratio of metals derived from polymetallic replacement deposits in the Tintic, Park City and Bingham mining districts. C. Ratios for arsenic and selenium. Three-point running means are plotted to smooth the curves. Note log scales.

The maximum degree of pollution in the sediments can be characterized by calculating a Maximum Contamination Index which shows the maximum concentration observed throughout a core, relative to the mean pre-industrial background concentration (Fig. 11a). The two core sites in Gilbert Bay showed relatively similar patterns. Silver and copper in these cores showed the highest Maximum Contamination Indices, with values >40-fold above background. The indices for silver are somewhat problematic, as pre-industrial concentrations in some strata were below levels of detection, so we assumed that these were 80% of the detection level, and this influenced the resulting Contamination Indices. Lead, molybdenum and cadmium had intermediate Maximum Contamination Indices (16-34 fold above background). Mercury had a Maximum Contamination Index of 4 and 12 at Sites 3 and 4, respectively. Selenium, zinc and arsenic all had contamination indices <6. The Maximum Contamination Indices at Site 1 in Farmington Bay did not show the same trends as those in Gilbert Bay. In Farmington Bay mercury and silver were much more enriched relative to background concentrations with Maximum Contamination Indices above 50. Lead, tin, copper and cadmium in the Farmington Bay core had Maximum Contamination Indices of 12-23 which were generally lower than those in Gilbert Bay.

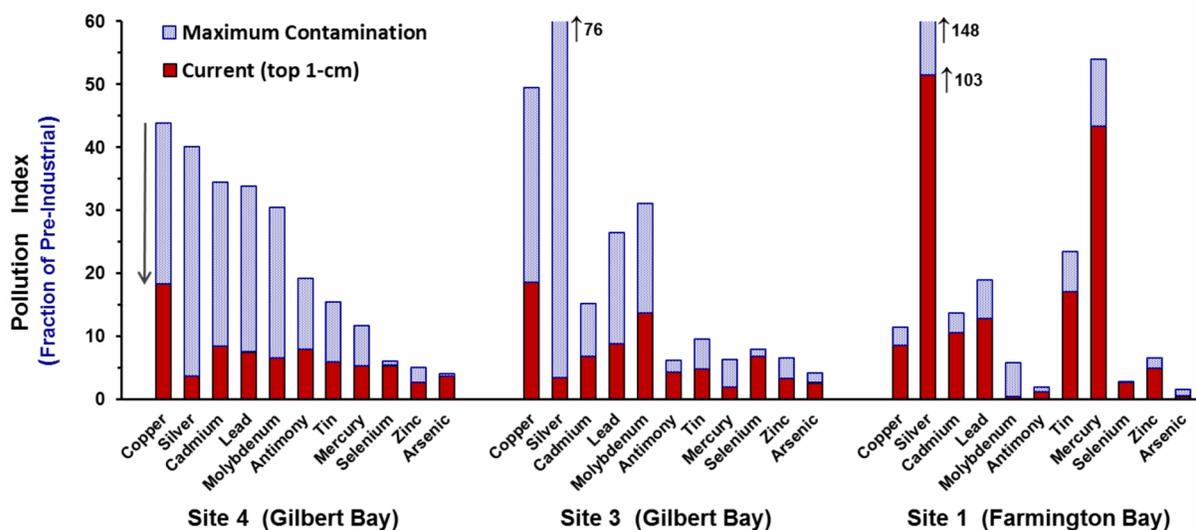


Figure 11. Maximum Contamination Index (maximum/background) in the sediment cores (blue) and the Current Contamination Index (surface concentration/background; red) for 11 metals and metalloids at three coring sites in the Great Salt Lake. These values are based on the actual maxima (not the 3-pt running means) that are shown in Figure 10. The down-arrow depicts how concentrations of many metals in Gilbert Bay are now considerably lower than during the highest contamination period in the 1950s and 1960s.

The present condition of surficial sediments can be summarized with the Current Contamination Index, or the amount of a metal in the top 1-cm of sediments expressed relative to background conditions (Fig. 11). When viewed in relation to the Maximum Contamination Index, the degree of improvement can be assessed. For Gilbert Bay the Current Index indicates that both selenium and arsenic are still near the highest levels observed in the cores, either because inputs are continuing, or because these two metalloids are more mobile in sediments than others. Copper and lead, which had some of the highest Contamination Indices, have recovered significantly, and concentrations in surface sediments are only 22-42% of the highest concentrations observed in the 1950s and 60s. Nevertheless, most metal concentrations in the surface sediments of Gilbert Bay are still 3-18 fold higher than background concentrations.

For the Farmington Bay core site, the Current Contamination Indices show that there has been relatively little decrease in the concentration of most metals (Fig. 11). The surface concentrations of arsenic and molybdenum are the only ones that have declined markedly since peak levels of contamination. In fact, the main reason that the other metals have Current Contamination Indices much below The Maximum Contamination Index is because of the short-lived spike in concentrations that occurred in the middle of the 1950s (Fig. 8). Following the recovery from that spike, most metal concentrations have maintained relatively similar or slightly increasing concentrations for the last 40 years, suggesting that metal pollution is continuing in this area of Farmington Bay (Fig. 8, Appendix 1).

Table 3. Average concentrations ($\mu\text{g g}^{-1}$) of metals in the top 1-cm of cores taken in three bays of the Great Salt Lake. Threshold effects concentrations (TEC) and the Probable Effects Concentrations (PEC) (McDonald et al. 2000) are also shown. For silver, the Washington State Standard (2002) for marine waters is shown as a TEC. For selenium the US Department of Interior (1998) values for "no effects" and "toxicity threshold" are used. Lake sediment concentrations exceeding the TECs are shown in bold numerals, and those exceeding the PECs are underlined. Note that these threshold and toxicity guidelines are shown only to provide comparison with criteria developed for fresh waters, and are not necessarily indicative of hazardous concentrations in Great Salt Lake sediments.

| Metal | TEC | PEC | Gilbert Bay | | | Farmington Bay | | | Bear River Bay | |
|---|------|-------|-------------|-------------|-------------|----------------|-------------|-------------|----------------|------|
| | | | 3 | 4 | 5 | 1 | 7 | 2 | 6 | 9 |
| Metals exceeding freshwater threshold effect concentrations at one or more sites | | | | | | | | | | |
| Selenium | 1.0 | 4.0 | 4.5 | 4.9 | 4.3 | 2.4 | 2.1 | 3.6 | 1.2 | 0.8 |
| Copper | 32 | 149 | 102 | 207 | 91 | 145 | 132 | 47 | 16.4 | 14.8 |
| Arsenic | 10 | 33 | 23 | 47 | 18 | 19 | 21 | 14 | 7 | 5 |
| Mercury | 0.18 | 1.06 | 0.11 | 0.42 | 0.30 | 0.88 | 0.66 | 0.20 | 0.05 | 0.05 |
| Cadmium | 1.0 | 5.0 | 1.1 | 2.4 | 0.7 | 2.8 | 2.5 | 0.9 | 0.4 | 0.4 |
| Lead | 36 | 128 | 31 | 58 | 21 | 130 | 83 | 29 | 19 | 15 |
| Zinc | 121 | 459 | 75 | 125 | 47 | 264 | 205 | 76 | 46 | 35 |
| Other Metals | | | | | | | | | | |
| Cobalt | | | 3 | 4 | 2 | 5 | 4 | 2 | 4 | 4 |
| Chromium | 43.4 | 111.0 | 8.5 | 11.5 | 5.9 | 35.3 | 23.1 | 7.5 | 9.3 | 7.3 |
| Molybdenum | | | 32.6 | 16.9 | 21.6 | 1.2 | 1.1 | 20.9 | 0.4 | 0.1 |
| Nickel | 22.7 | 48.6 | 9.8 | 14.3 | 6.5 | 12.7 | 13.8 | 8.1 | 13.3 | 9.7 |
| Antimony | | | 0.40 | 0.64 | 0.62 | 0.04 | 0.06 | 0.48 | 0.03 | 0.02 |
| Silver | | | 0.02 | 0.1 | <0.03 | 1.9 | 2.0 | 0.1 | 0.1 | 0.1 |
| Tin | | | 0.77 | 1.65 | 0.62 | 2.76 | 0.06 | 0.48 | 0.03 | 0.02 |
| Titanium | | | 0.39 | 0.84 | 1.40 | 0.43 | 0.35 | 1.54 | 0.49 | 0.32 |
| Vanadium | | | 13.7 | 19.7 | 9.7 | 21.8 | 20.6 | 11.4 | 9.6 | 8.3 |

Metal concentrations in surface sediments

The surface sediment data indicate that both Gilbert and Farmington Bays have concentrations of several metals that are above Threshold Effects Criteria established for fresh waters, but that Bear River Bay is relatively uncontaminated (Table 3; Appendix 1). From a current ecosystem health perspective, the most relevant metal concentrations are those at the sediment-water interface, as they are the ones that most influence diffusion into the water column, and uptake by the benthic biota. Selenium exceeded the freshwater threshold effects criteria most frequently (Table 3). Concentrations of selenium were highest at all three sites in Gilbert Bay, and exceeded the freshwater Probable Effects Concentration at all of these. Selenium also exceeded the Threshold Effects Concentrations at all three sites in Farmington Bay, but never exceeded the PEC concentration. Selenium in the sediments in Bear River Bay only exceeded

the TEC at one site. Copper and arsenic concentrations exceeded the TEC at all of the sites in Gilbert and Farmington Bays but never in Bear River Bay. Mercury was also relatively high in the surface sediments of Gilbert and Farmington Bay, but concentrations did not exceed the PEC level in any of them. Again, Bear River Bay had relatively low levels of this contaminant. Cadmium, lead and zinc had concentrations above TEC levels at one or more sites in Gilbert and Farmington Bays.

Coring sites closest to many pollutant sources at the south end of the lake usually had higher surface metal concentrations than those further from these putative sources (Fig. 12). In Gilbert Bay, Site 4 which is closest to smelters and industrial centers in the Salt Lake Valley had higher concentrations of nearly all the metals than did Sites 3 and 5. This was particularly true for copper. In Farmington Bay there was a consistent gradient for all of the metals that were above TEC concentrations, with Site 1 near the NW Oil Drain discharge and closer to smelters having the highest concentrations, followed by Site 7 which is also at the south end of the bay, and with the lowest concentrations at Site 2 at the northern end of the bay. Bear River Bay at the northern end of the lake had the lowest concentrations of all of the metals measured.

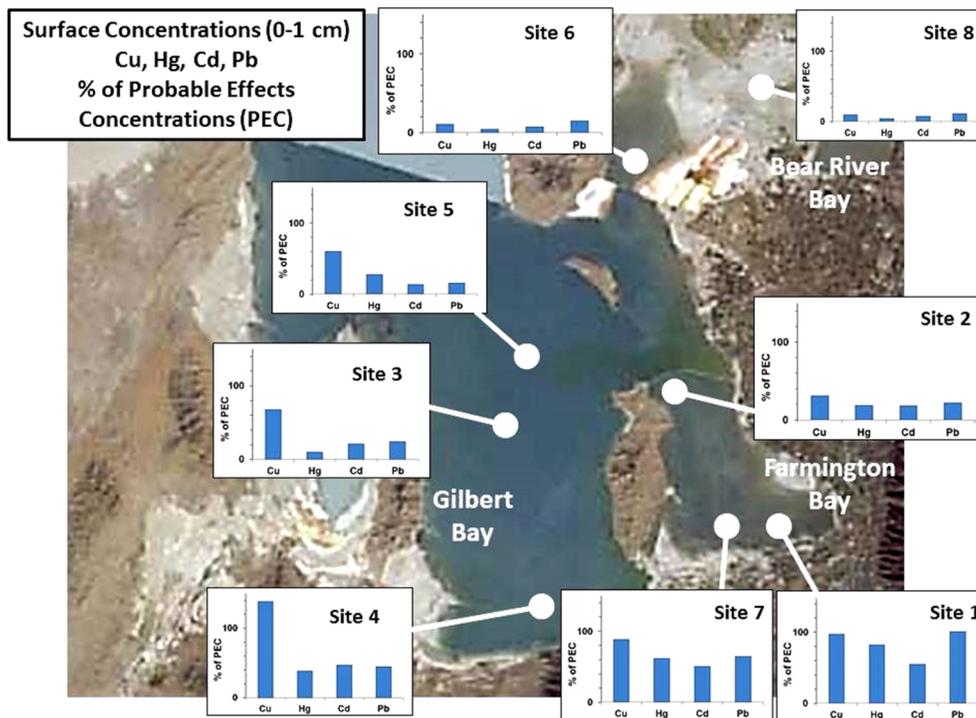


Figure 12. Mean surface (0-1 cm) concentrations of copper (Cu), mercury (Hg), cadmium (Cd) and lead (Pb) at eight sites in the Great Salt Lake expressed as a percentage of the probable effects concentrations (PEC) developed for freshwater organisms (Table 3; Macdonald et al. 2000).

Discussion

Gilbert Bay chronologies—In Gilbert Bay the sediment chronologies for most metals showed 10-60 fold increases in concentrations in the early to mid-1900s, followed by significant declines in the latter half of the century. The increases of many of these metals in the lake paralleled the increases in production of these metals from smelters in the Salt Lake Valley and nearby mining districts. These processed metals originated not only from mines near the Great Salt Lake, but also from more distant areas; as the Salt Lake Valley was a smelting center for the intermountain west during the early and mid-20th century (Varley et al. 1921). The increases in metal contaminants in the sediments are consistent with those described in Uinta Mountain lakes, 90-155 km east of the Great Salt Lakes (Kada et al. 1994; Moser et al. 2010; Reynolds et al. 2010), thus suggesting that atmospheric deposition has been an important mechanism for the contamination of the Great Salt Lake sediments. The magnitude of increase in metals in the Great Salt Lake was much greater, however, than in the distant Uinta Lakes. For example, in Marshall and Mirror Lakes that are 90-110 km from the Great Salt Lake, increases in lead were 11-12X background, compared to increases of 21-34X at Sites 3 and 4 in the Great Salt Lake (Kada et al. 1994; Reynolds et al. 2010). The difference in copper deposition was even greater between the sites, with contamination factors of ~2X in the two Uinta Lakes and 34X in the Great Salt Lake. Maximum contamination indices were even less at a Uinta lake 155 km east of the Salt Lake Valley (Reynolds et al. 2010). Consequently, the very high metal concentrations found in older Great Salt Lake sediments is not surprising given the lake's proximity to the smelting activities and other industrial activities in the Salt Lake Valley.

The decline in metal contamination in younger sediments in the Great Salt Lake is likely due to improvements in smelting techniques, and in some cases, to lower production of specific metals from the mines. In the case of lead and zinc, production in the State has dropped to zero, and this is reflected in the sediments with current concentrations of only 22-33% and ca. 45-53% of the peaks in the 1940s and 50s, respectively. Additionally, lead use in gasoline peaked in the early 1970s and has also declined to near zero (Laws 1993) and this may have contributed to lower concentrations in recent sediments. In contrast to lead, copper production in Utah has continued to increase to record levels (Fig. 6), but copper levels in the sediments have nevertheless declined to 33-42% of the maximum levels found in the mid-1950s. The decline in copper and other metals in the sediments is likely due to improvements in technology and virtual halt of lead-zinc mining and smelting in the Great Basin in the early 1970s. Some of the improvements began as early as 1906 when farmers sued smelters in Midvale, Utah and forced them to reduce emissions of sulfuric acid, lead and zinc (<http://utahrails.net/mining/smelters-midvale.php>). The smelters responded by processing ores with less sulfur, and by building a higher smoke stack to help disperse the pollutants from the valley. Other improvements were initiated after the Clean Air Act of 1963 and subsequent amendments in the 1970s. Economic factors also helped reduce emissions, because smelters began to recover the sulfuric acid that carries the metals, and sell it for a profit. As a result of emission controls and economic factors, one of the largest copper producers in the world (Kennecott Copper, now Rio Tinto) located on the south shore of the Great Salt Lake, reduced emissions of sulfuric acid from ~22 MT h⁻¹ in 1974 to ~0.03 MT h⁻¹ (Newman et al. No Date).

Another mechanism that could have contributed to the decline of metals in the sediments involves the construction of the Southern Pacific railroad causeway that divided the lake in half and led to the formation of a deep brine layer (monimolimnion) in Gilbert Bay (Gwynn 2002a; Jones and Wurtsbaugh

2014). This dense brine layer is frequently present below a depth of ~6.5 m, and the density of the deeper brine reaches 1.2 g cm^{-3} , which is higher than that of most algal and detrital particles (Crosman and Horel 2009). Since most metals normally reach lake sediments as part of the algal and detrital sedimentation process, the presence of the deep brine layer may retain some of this material above the sediments, where mineralization would release it to the dissolved form. The deep brine layer contains extremely high concentrations of particulate organic carbon, as well as very high concentrations of dissolved mercury (Tank and Dodds 2003; Jones and Wurtsbaugh 2014), yielding some support for this hypothesis. Wurtsbaugh and Jones (Jones and Wurtsbaugh 2014) estimated that about 40% of the deep brine water may be entrained into the upper mixed layer each year, which could also keep metals from accumulating in the sediments. Cores taken by Reynolds and Moser (2010) in the Uinta Mountain lakes east of smelting activities did not record decreases in copper comparable to what we found in the Great Salt Lake. Additionally, most metals began to decline in the sediments of the Great Salt Lake at estimated dates of approximately 1950 (Site 4; Fig. 10), before the advent of much cleaner smelting technologies, and close to the time of construction of the railway causeway in 1959. These observations provide some support for the hypothesis that the deep brine layer may play a role in reducing metal accumulation in the sediments. However, more research will be necessary to determine the validity and importance of this hypothesized mechanism.

Like many of the other metals, mercury concentrations in the sediments of Gilbert Bay are also declining from the highest levels observed in the first part of the 20th century, although the record of decline at Site 4 has been erratic. Atmospheric and riverine deposition of mercury in the Great Salt Lake is continuing and may be enhanced by gold mining activities in Nevada and Utah, in addition to emissions from coal-burning power plants in central Utah and worldwide (Naftz et al. 2009a). However, the atmospheric and riverine input to the lake is not abnormally high compared to other western areas of the United States (Smith 2003; Peterson and Gustin 2008), nor has it been suggested that deposition rates are decreasing. The overall decline of mercury concentrations in the sediments suggests that the high mercury levels found in the lake (Naftz et al. 2008) may be due at least partially to legacy effects of early mining activities. Naftz et al. (2008) also found decreasing mercury concentrations in the surface strata of a core collected in Gilbert Bay that were comparable to the declines measured at Sites 3 and 4, suggesting that this may be a general phenomenon, at least for the deep areas of the lake.

The reason for the slightly higher and more irregular pattern of mercury and other metal concentrations in the Gilbert Bay Site 4 core than in the Site 3 core is not clear. One possible explanation is that periodic releases of sediments from the Kennecott/Rio Tinto south tailings pond near the south shore of the lake have contributed irregularly to the metals loading, and thus caused peaks in concentration. Site 4 is located at the south end of the lake ~13 km from the tailings pond whereas Site 3 is located 37 km to the north. The tailings pond dikes failed in 1941-42, 1944, 1951, 1964 and in 1969, although it is not clear how much material from the pond made it to the Great Salt Lake (EPA 2002). Additionally, in 1965 Kennecott experimented with waste disposal by building a south-shore dike that contained $56,000 \text{ m}^3$ (2 million ft^3) of tailings material. The dike was destroyed by wave action within two years, and the materials dispersed into the lake (EPA 2002). Typical tailings materials at the site contain $30 \text{ } \mu\text{g g}^{-1}$ arsenic, $240 \text{ } \mu\text{g g}^{-1}$ lead and $6 \text{ } \mu\text{g g}^{-1}$ selenium. If this material was spread over just 25% of the southern area of Gilbert Bay, it would have only deposited a layer about 0.8 mm thick and thus could have had some impact on concentrations. Dust blowing from the tailings ponds (EPA 2002) could

also contribute some metals to the southern end of the lake (particularly As, Se, and Pb), but the impact of this is unknown.

In contrast to most metals, selenium and arsenic concentrations in more recent sediments are either stable or showing slight increases. Arsenic occurs in both the Mercur gold ores and lead-zinc ores of western Utah. Selenium occurs at low, but geochemically anomalous levels in the Bingham Canyon porphyry copper deposits and is released from smelting activity (Newman et al. No Date). Selenium is also released in the Bear River watershed that drains phosphate mining districts in Idaho and the Uinta Mountains (Mars and Crowley 2003, Hamilton and Buhl 2005). Strip mining of phosphate has increased in recent years (Hughes and Thackray 1999) and thus contributions from this source may be increasing. However, Naftz et al. (2009b) found that only 25% of the riverine selenium loading was from the Bear River, and 75% were from the Salt Lake Valley watershed, particularly sub-drainages and outfalls associated with the Kennecott Mine tailings ponds. Arsenic and selenium are also emitted from smelting activities, as demonstrated locally by accumulations in Uinta Mountain lakes (Moser et al. 2010; Reynolds et al. 2010), and from tailings wastewater (EPA 2002). Consequently, the mining-related activities in the Great Salt Lake region may be the predominant source of this contamination.

Farmington Bay metal chronologies—The sediment core taken from the south end of Farmington Bay also showed significant levels of metal contamination, but the chronology is very different from that in Gilbert Bay (Fig. 13A), and not consistent with known periods when contaminants were entering the lake. The chronologies shown in Fig. 8 imply that there was little or no contamination with toxic metals in Farmington Bay from the late 1800s until the late 1950s or early 1960s, and then concentrations for most metals rose suddenly to high levels. However, the Gilbert Bay cores and the studies in the Uinta Mountains (Kada et al. 1994; Moser et al. 2010; Reynolds et al. 2010) indicate that atmospheric deposition of metals was widespread from the Salt Lake Valley smelting activities in the early 1900s, so it seems improbable that Farmington Bay escaped this contamination. Additionally, the Jordan River also

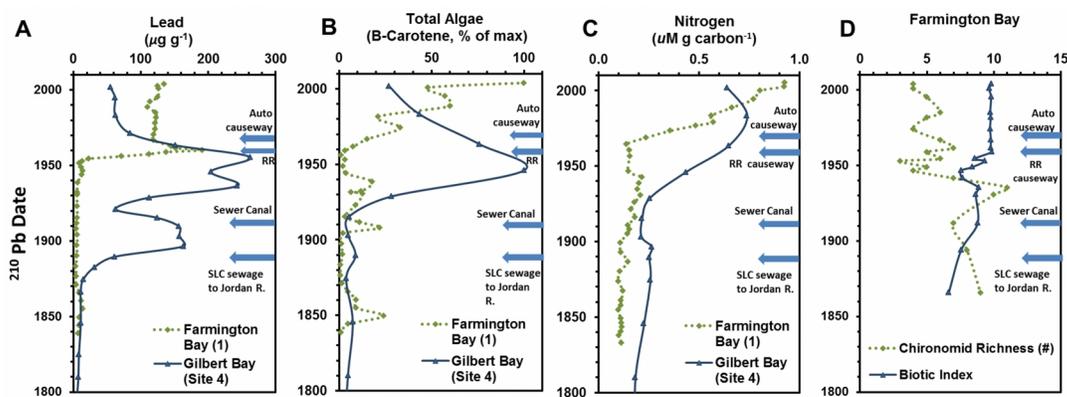


Figure 13. A-C. Comparisons of lead, a total algal pigment indicator and nitrogen content of the sediments in the Farmington Bay core, and in Site 4 in Gilbert Bay. D. Changes in chironomid species richness and a biotic index in Farmington Bay (from Moser et al. 2012). For the biotic index (D), the earliest estimate (1841) was not plotted because only 1 chironomid head capsule was found. Arrows show the timing of important events increasing pollution to Farmington Bay, and in changing the hydrological connectivity between different parts of the lake.

discharges near Site 1, and was contaminated with lead and arsenic by sludge from the Murray Smelter that operated from 1872 until 1949 (and is now a Superfund Site; (Keck and Lepori 2012). Furthermore, the Northwest Oil Drain that joins the Sewer Canal and discharges 1-2 km from the coring site has been “used to convey urban stormwater flows, industrial stormwater discharges and wastewater effluent discharges” since its construction in 1922. The canal is currently an EPA Superfund Cleanup Site and lead, copper and arsenic are metals of concern (The Forrester Group 2001). The Sewer Canal also connects to the City Canal which drained industrial areas of much of central Salt Lake City. Discharges into the Oil Drain did likely increase in the late 1940s and 50s when refinery expansion occurred (Strack 2009), but this cannot explain the lack of metals in the core that should have been deposited from atmospheric contamination occurring from the late 1800s through the entire 20th century. Redox, pH and organic matter accumulation can influence the mobility and thus the accumulation of metals in sediments (Boudreau 1999). However, Farmington Bay was reportedly highly organic since the diversion of raw sewage into the south end of the bay starting in 1911 (Van Der Meide and Nicholes 1972), so metals likely had binding sites. The automobile causeways to Antelope Island constructed in 1952 and 1969 may have contributed to metal increases in the sediments by reducing advection to Gilbert Bay, and increasing algal growth (Healey and Hendzel 1980). Additionally, the diversion of flood flows to Gilbert Bay via the Goggin Drain that began in 1952 may have reduced flushing of Farmington Bay with relatively high quality snow-melt water. Measurements of a core from Site 1 indicated that total carbon in the sediments rose gradually from 5-6% from 1840-1950, and then more rapidly from 6-9% during the 1950-2000 period (P. Leavitt, unpublished data). This suggests that perhaps there was more organic carbon after 1950 that could have increased metal retention, but the differences over the length of the core are not large. Additionally, metals are relatively immobile in alkaline environments (Dillon et al. 1988; Belzile et al. 2004) such as those in the Great Salt Lake, especially when sulfates and H₂S production are high and metal sulfides are precipitated (Carignan and Nriagu 2003). Naftz et al. (2000) analyzed a sediment core from the south end of Farmington Bay and found that lead and other pollutants increased significantly between 1916 and 1950, but the depth resolution of their analysis was insufficient to help resolve the discrepancy in my core between the ²¹⁰Pb dating and known pollution activities. An additional core they took in Decker Lake, located near Farmington Bay and in the floodplain of the Jordan River yielded no interpretable ²¹⁰Pb dating (Naftz et al. 2000; Naftz, personal communication). The reason that the Farmington Bay core showed no indication of metal contamination up until the 1950’s is thus unclear, but it is not likely that the sediments accurately recorded the metals in the bay during this period.

The sudden increase of metals in Farmington Bay in the 1950s and early 1960s parallels similar increases in most eutrophication metrics (e.g. pigments, nitrogen, isotope signatures, diatoms, etc.; (Healey and Hendzel 1980) and macroinvertebrate bioindicators (Dodds et al. 2002). For example, the ²¹⁰Pb chronology indicates that the metric of total algal pigments (B-carotene) and nitrogen content in Farmington Bay did not increase significantly above the 1850s-1950s baseline until after 1960, whereas the increases began much earlier in Gilbert Bay (Figs. 13b, c). There were some changes in Farmington Bay prior to 1960 in purple sulfur bacteria, the cyanobacteria *Gleotrichia* sp. and chlorophyte fossils (Healey and Hendzel 1980), but the majority of eutrophication parameters did not respond until after this date. Similarly, the benthic invertebrates in Farmington Bay had only minor responses until after ~1950 (Fig. 13d). These delayed responses are not consistent with the changes in Gilbert Bay that started in the earlier part of the century because: (1) most nutrients and organic pollution first enter Farmington Bay a few kilometers from Site 1 before being advected into Gilbert Bay, and; (2) dilution of nutrients in Gilbert

Bay would have been 100-fold greater than in Farmington Bay prior to the Railway Causeway construction, and 50-fold after the construction. Nutrient and organic pollution of Farmington Bay (and the rest of lake) would have begun with the early sewage discharges into the Jordan River in 1889, but would likely have increased markedly in 1911 when the Sewer Canal was constructed to the south end of Farmington Bay (Hooton no date). Vander Meide and Nicholes (1972) described the pollution situation there as: “...until recently, the entire liquid sewage refuse of the Wasatch Front towns ... ended up untreated, except for dilution, in the estuary. This practice has resulted in a sewage delta of rather large proportions in a very localized area.” Consequently, it seems that for many of the eutrophication indicators, like the metals, the sediments did not retain an accurate record of eutrophication in the early part of the 20th century. Reduced hydrologic flushing due to the construction of the Antelope Island causeways likely exacerbated an already bad situation by retaining nutrients and by freshening Farmington Bay and consequently allowing the growth of nitrogen fixing cyanobacteria. This, in turn, may have increased nitrogen export to Gilbert Bay.

Regardless of difficulties in interpreting the timing of the older sediment record of metals in Farmington Bay, the recent pattern is clear: Concentrations of all metals are high and showing no indication of decreasing (Fig. 8). This contrasts with the substantial declines in the concentrations of most metals in the more recent sediments Gilbert Bay. The stability of the metal concentrations in the Farmington Bay core suggests that loading via the Northwest Oil Drain/Sewer Canal or some other source may still be contributing to the high levels of many metals that are now higher than those found in the Gilbert Bay cores. Additionally, the smearing of the ¹³⁷Cesium (Fig. 5) suggests that some sediment mixing, or mobility of metals within the sediments could be occurring at this site, and this would also minimize decreases of metals in the surficial sediments.

Spatial distribution and toxicity of metals—Concentrations of metals in the surface sediments varied widely in the three bays, with those at the south end of Farmington Bay being the highest, and those in Bear River Bay the lowest. These results can be compared with studies done on sediments in 1986 (Sorensen et al. 1988) when the lake was near its all-time high elevation and a recent study (Waddell et al. 2009) when the lake was low. Sorensen et al. (1988) sampled sediments in a grid pattern in the vicinity of the Sewer Canal outfall that included our Site 1 location (Fig. 14), and found mean levels in surficial sediments that were within 20% of the values we found (Table 3). However, 5 km NW of Site 1 they found concentrations of most metals that were <20% of the highest concentrations found near the outfall (Fig. 14), thus indicating that our Site 1 may represent a worst-case scenario for metals contamination in Farmington Bay. Waddell et al. (2009) found concentrations near the Sewer Canal outfall that were approximately 50% of what we measured, but Sorensen et al.’s study demonstrated that there are fine-scale pockets of contamination associated with oil and grease deposits, so the difference between the studies is not surprising.

Our samples in Bear River Bay generally had relatively low concentrations of metals (Table 3), with only selenium exceeding threshold effects concentrations. Given the heavy mining activity and selenium contamination in the Bear River drainage, this is surprising. However, selenium only exceeded the Threshold Effects Concentration at one of the two sites, and only by a small amount, so the situation does not appear to be severe. The metal concentrations in Bear River Bay may also be low because of the high sediment load carried by the river (Allred and Judd 2002) that should “dilute” metals entering the system. Sorensen et al. (1988) also sampled sediments where Bear River Bay connects with Gilbert Bay

(Lucin Cutoff) , and generally found metal concentrations 2-4 times higher than we encountered at the two sites within Bear River Bay itself. The reason for the difference is not clear, but their sampling site was near the railway line, and thus might have had local contamination from the locomotives.

In Gilbert Bay’s surface sediments we found that most metals exceeded the Threshold Effects Concentrations, and copper, arsenic and mercury exceeded Probable Effects Concentrations. However, these concentrations were found in deep area of the lake under the deep brine layer (monimolimnion) where there are no macroinvertebrates due to the anoxic, reducing conditions there (Collins 1980; Bioeconomics 2012). The movement of the metals beneath the deep brine layer into strata that might influence the food web in the mixed layer may be dependent on internal waves (seiches) that can suspend sediments at the periphery where this layer contacts the bottom sediments (Beisner et al. 2009). Additionally, when the lake reaches very low elevations, the deep brine layer can disappear and this might allow colonization by brine fly larvae, thus exposing them to the metals in the deep sediments. An additional factor that we cannot assess with our limited coring is the vertical distribution of metals in the sediments in shallower water. These sediments might be more vulnerable to turbulence that could homogenize the upper layers, thus providing higher concentrations of most of the metals to benthic invertebrates and brine shrimp that sometimes graze at the sediment-water interface. Consequently, although we noted marked decreases in the concentrations of most metals in the surficial sediments, these decreasing profiles are not necessarily representative of sediments in shallow areas of the lake where macroinvertebrates feed. However, Waddell et al. (2009) did not find marked differences in metals concentrations between samples collected at deep offshore sites, and those in shallower water. Their sampling, however, was with a Ponar dredge that penetrated ca. 3-7 cm into the sediments (N. Darnall, personal communication), and thus would have provided a sample that averaged metal concentrations that may have varied several fold. In some cases a sample that averaged several decades of deposition would be desirable, but if the more active surface layer is the target of an investigation, care should be taken to standardize the thickness of the sediments that are sampled. Nevertheless, the metal concentrations measured by Waddell et al. (2009) were generally similar to our surface samples, thus suggesting that their concentrations were representative of the metal concentrations that the biota would be exposed to.

The primary invertebrates in Gilbert Bay that would be exposed to the metals in the sediment are brine flies which are relatively resistant to toxins (Brix et al. 2004). Consequently, the fact that many

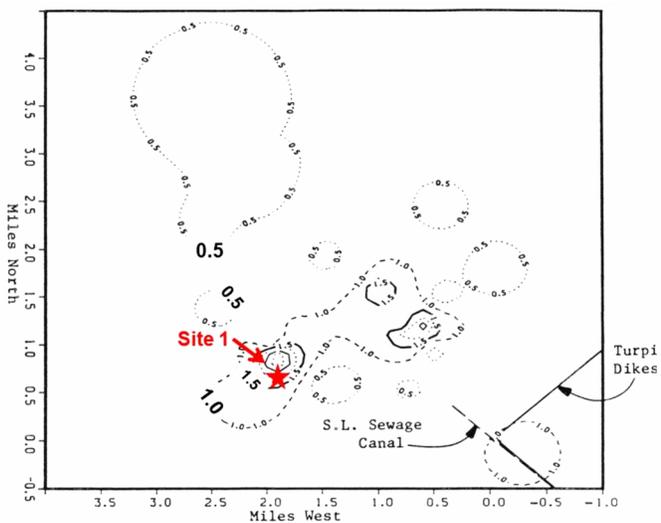


Figure 14. Isopleths of total mercury concentration ($\mu\text{g g}^{-1}$) near the outlet of the Salt Lake City Sewage Canal (NW Oil Drain) described by Sorensen et al. (1988). The location of our Site 1 is shown with a red star, and demonstrates that our core was located in a hot-spot of mercury contamination (and other metals).

metals in Gilbert Bay sediments exceed Threshold- and Probable Effects Concentrations for fresh water organisms is not necessarily a direct concern for this species. A bigger issue is whether brine flies bioaccumulate metals and transfer them into the birds that rely on them to fuel their migrations. Analyses of selenium and mercury bioaccumulation in the brine flies living on stromatolites in the lake have not shown significant biomagnification in the lower food chain (Wurtsbaugh 2009; Wurtsbaugh et al. 2011). However, stromatolite and benthic sediment communities may function quite differently, so care must be applied to extending this earlier work to all of the lake community. The south end of Farmington Bay has a much wider diversity of invertebrates (Dodds et al. 2002) that are adapted to fresh water, and the high concentrations of metals in those sediments could potentially impact them if the freshwater Threshold Effects Concentrations and Probable Effects Concentrations are exceeded. However, since the spatial extent of sediments in Farmington Bay with high metal concentrations is limited (Fig. 12; Sorensen et al. 1988) the effects may be localized.

Conclusions

The Great Salt Lake, located next to a major mining, smelting and industrial area, has been very contaminated with a variety of metals. Despite the very high metal concentrations in sediments deposited in mid-1950s, most metal concentrations in more recent deposits have declined substantially, likely as a result of: (1) decreased mining for some minerals such as lead and zinc; (2) cleaner smelting technologies; (3) removal of lead from automotive gasoline, and (4) other environmental controls. The formation of the deep brine layer starting in 1969 also likely has reduced toxic metal delivery to the sediments. Although there have been marked decreases in metal concentrations, levels of copper, mercury, cadmium, lead, zinc, selenium and arsenic remain above freshwater Threshold Effects Concentrations at one or more stations in Gilbert and Farmington Bays. Bear River Bay, although not sampled extensively, does not appear to have a metals contamination problem. The invertebrates in the high-salinity areas of the Great Salt Lake may be relatively resistant to a variety of pollutants and the freshwater criteria consequently may not be applicable for them. Nevertheless, these invertebrates can be vectors that transfer metals to birds that depend on this ecosystem. Additional research will be needed to determine if some of the poorly-studied metals in the lake (Cu, As, Cd, Pb and Zn) might be having adverse effects on the bird community. Notwithstanding the declining levels of many metals in the lake, concentrations of arsenic and selenium are either stable or increasing in the sediments. Managers should carefully monitor these metals to insure that they do not increase to even higher levels.

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Appendix 1a. Gilbert Bay (Site 3) metal concentrations in the sediment core from the Great Salt Lake. These concentrations are corrected for the amount of salt in the interstitial water, and thus represent concentrations per gram of dry sediment. To calculate the actual values measured in the dried sediment+salt mixture, multiply concentrations by the value "Sediment (fraction of dry wt)".

| 210Pb Year (estimate) | Core Mid-Depth (fraction of dry wt) | Sediment (fraction of dry wt) | Salt-Corrected Concentration ($\mu\text{g g}^{-1}$ dry sediment; ppm) | | | | | | | | | | | | | | | | | | | | | | | | | | | |
|-----------------------|-------------------------------------|-------------------------------|--|-------|------|------|------|------|------|-------|-------|------|-------|--------|------|------|-------|-------|------|------|------|------|------|-----|--|--|--|--|--|--|
| | | | Ag | Al | As | Be | Cd | Co | Cr | Cu | Fe | Hg | Mo | Na | Ni | P | Pb | Sb | Se | Sn | Sr | Tl | V | Zn | | | | | | |
| 2003.5 | 0.25 | 0.52 | 0.02 | 2550 | 15.0 | 0.15 | 0.70 | 1.78 | 5.7 | 65.4 | 3627 | 0.12 | 14.81 | 334838 | 5.8 | 540 | 18.6 | 0.387 | 4.18 | 0.46 | 187 | 0.31 | 8.5 | 48 | | | | | | |
| 1999.2 | 0.75 | 0.52 | 0.02 | 6071 | 31.8 | 0.41 | 1.45 | 4.41 | 11.3 | 137.9 | 8966 | 0.10 | 50.42 | 320453 | 13.7 | 1510 | 44.4 | 0.406 | 4.85 | 1.08 | 451 | 0.46 | 19.0 | 103 | | | | | | |
| 1997.5 | 1.25 | 0.55 | 0.01 | 6108 | 29.6 | 0.45 | 1.36 | 4.51 | 11.7 | 144.1 | 9358 | 0.11 | 58.23 | 280874 | 14.6 | 1419 | 43.9 | 0.217 | 4.36 | 0.92 | 511 | 0.43 | 19.0 | 104 | | | | | | |
| 1995.8 | 1.75 | 0.59 | 0.07 | 6556 | 30.3 | 0.49 | 1.75 | 4.85 | 12.7 | 222.4 | 10010 | 0.15 | 45.44 | 186598 | 15.7 | 1418 | 51.7 | 0.187 | 3.72 | 1.19 | 543 | 0.54 | 21.7 | 120 | | | | | | |
| 1993.2 | 2.25 | 0.52 | 0.02 | 6144 | 27.0 | 0.45 | 2.08 | 4.72 | 12.7 | 238.2 | 9625 | 0.19 | 62.85 | 235051 | 16.2 | 1345 | 60.0 | 0.427 | 5.18 | 1.42 | 644 | 0.58 | 23.4 | 136 | | | | | | |
| 1990.6 | 2.75 | 0.44 | 0.02 | 4696 | 20.6 | 0.41 | 1.51 | 3.73 | 9.7 | 159.8 | 7256 | 0.16 | 79.71 | 372837 | 13.0 | 1058 | 46.4 | 0.475 | 5.92 | 0.93 | 356 | 0.47 | 19.2 | 102 | | | | | | |
| 1988.4 | 3.25 | 0.47 | 0.21 | 4419 | 20.0 | 0.32 | 1.43 | 3.75 | 11.3 | 162.3 | 7042 | 0.24 | 75.80 | 418502 | 12.9 | 1251 | 43.2 | 0.728 | 4.22 | 0.94 | 330 | 0.41 | 20.1 | 96 | | | | | | |
| 1986.3 | 3.75 | 0.49 | 0.24 | 4690 | 19.8 | 0.30 | 1.44 | 3.76 | 10.6 | 172.5 | 7461 | 0.14 | 65.17 | 398801 | 12.9 | 1251 | 44.3 | 0.325 | 4.33 | 0.98 | 346 | 0.41 | 18.9 | 94 | | | | | | |
| 1980.2 | 4.25 | 0.52 | 0.33 | 5259 | 20.8 | 0.37 | 1.72 | 4.31 | 12.4 | 183.9 | 8540 | 0.17 | 66.78 | 307993 | 14.5 | 1256 | 54.3 | 0.638 | 4.14 | 1.24 | 395 | 0.54 | 21.7 | 111 | | | | | | |
| 1974.0 | 4.75 | 0.54 | 0.35 | 5774 | 22.8 | 0.41 | 1.99 | 4.59 | 12.6 | 206.3 | 9402 | 0.18 | 46.45 | 300300 | 15.5 | 1358 | 61.4 | 0.350 | 3.74 | 1.42 | 454 | 0.55 | 22.5 | 119 | | | | | | |
| 1965.3 | 5.25 | 0.57 | 0.52 | 6528 | 26.5 | 0.45 | 3.18 | 5.29 | 13.9 | 335.2 | 10608 | 0.24 | 37.35 | 229877 | 17.2 | 1234 | 87.2 | 0.157 | 3.18 | 1.82 | 749 | 0.77 | 27.5 | 154 | | | | | | |
| 1966.6 | 5.75 | 0.60 | 0.46 | 6688 | 24.8 | 0.46 | 3.17 | 5.29 | 15.4 | 287.7 | 10252 | 0.28 | 41.49 | 220027 | 17.3 | 1273 | 86.7 | 0.531 | 3.60 | 1.83 | 645 | 0.68 | 27.2 | 148 | | | | | | |
| 1969.4 | 6.25 | 0.67 | 0.57 | 7031 | 27.2 | 0.48 | 4.17 | 5.25 | 16.4 | 364.4 | 9847 | 0.26 | 60.63 | 204455 | 17.7 | 1142 | 108.4 | 0.677 | 3.56 | 2.08 | 619 | 0.90 | 24.6 | 153 | | | | | | |
| 1942.1 | 6.75 | 0.73 | 0.55 | 11106 | 27.9 | 0.47 | 3.78 | 5.09 | 17.2 | 422.9 | 10994 | 0.29 | 72.28 | 145529 | 18.0 | 1059 | 108.3 | 1.196 | 3.57 | 2.35 | 811 | 1.09 | 23.7 | 182 | | | | | | |
| 1934.8 | 7.25 | 0.76 | 0.49 | 8807 | 32.8 | 0.49 | 4.79 | 5.43 | 17.0 | 461.0 | 11041 | 0.30 | 73.31 | 110537 | 18.3 | 1026 | 145.2 | 0.844 | 3.47 | 2.61 | 1168 | 1.23 | 24.1 | 189 | | | | | | |
| 1927.6 | 7.75 | 0.79 | 0.42 | 5794 | 32.8 | 0.42 | 4.67 | 4.53 | 13.3 | 367.5 | 9880 | 0.33 | 58.65 | 112479 | 16.9 | 905 | 114.5 | 0.633 | 2.70 | 2.25 | 1662 | 0.98 | 21.4 | 172 | | | | | | |
| 1919.9 | 8.25 | 0.81 | 0.41 | 4982 | 29.9 | 0.42 | 3.68 | 4.02 | 9.2 | 219.8 | 8044 | 0.52 | 28.45 | 88246 | 15.5 | 761 | 125.9 | 0.555 | 2.31 | 1.44 | 2335 | 0.55 | 18.7 | 147 | | | | | | |
| 1912.3 | 8.75 | 0.83 | 0.37 | 5003 | 17.9 | 0.38 | 1.59 | 3.58 | 7.3 | 221.1 | 8323 | 0.34 | 11.44 | 88246 | 16.1 | 693 | 95.2 | 0.084 | 1.65 | 0.96 | 2125 | 0.36 | 15.2 | 128 | | | | | | |
| 1906.9 | 9.25 | 0.83 | 0.49 | 5426 | 16.2 | 0.43 | 1.53 | 4.00 | 7.9 | 186.5 | 8712 | 0.38 | 11.58 | 78054 | 16.7 | 675 | 119.4 | 0.371 | 1.68 | 1.08 | 1940 | 0.34 | 16.3 | 144 | | | | | | |
| 1901.5 | 9.75 | 0.84 | 0.35 | 5878 | 12.3 | 0.48 | 1.01 | 3.86 | 7.6 | 72.8 | 9289 | 0.38 | 6.20 | 80235 | 15.6 | 619 | 108.9 | 0.119 | 1.54 | 0.62 | 1819 | 0.24 | 15.0 | 112 | | | | | | |
| 1897.0 | 10.25 | 0.84 | 0.34 | 6281 | 11.6 | 0.57 | 0.92 | 4.04 | 8.2 | 28.0 | 9766 | 0.38 | 5.25 | 77513 | 15.9 | 660 | 104.4 | 0.095 | 1.55 | 0.51 | 1715 | 0.23 | 15.5 | 100 | | | | | | |
| 1892.5 | 10.75 | 0.85 | 0.15 | 5981 | 10.7 | 0.45 | 0.76 | 3.84 | 7.9 | 20.0 | 9029 | 0.52 | 3.11 | 77232 | 15.5 | 596 | 94.1 | 0.106 | 1.67 | 0.44 | 1590 | 0.19 | 14.6 | 83 | | | | | | |
| 1888.0 | 11.25 | 0.86 | 0.02 | 5307 | 9.8 | 0.37 | 0.43 | 3.44 | 7.3 | 12.5 | 8166 | 0.21 | 2.44 | 65614 | 10.0 | 562 | 33.9 | 0.058 | 0.92 | 0.33 | 1731 | 0.23 | 14.3 | 46 | | | | | | |
| 1881.0 | 11.75 | 0.85 | 0.01 | 5329 | 9.3 | 0.39 | 0.48 | 3.35 | 7.1 | 16.2 | 8487 | 0.30 | 2.65 | 75776 | 15.3 | 563 | 55.6 | 0.129 | 1.08 | 0.34 | 1739 | 0.19 | 13.6 | 57 | | | | | | |
| 1874.0 | 12.25 | 0.85 | 0.01 | 5703 | 11.0 | 0.45 | 0.27 | 3.61 | 7.6 | 11.4 | 8657 | 0.13 | 2.68 | 74745 | 14.7 | 522 | 13.1 | 0.059 | 1.06 | 0.29 | 1578 | 0.20 | 14.2 | 41 | | | | | | |
| 1867.0 | 12.75 | 0.85 | 0.01 | 5794 | 12.0 | 0.45 | 0.23 | 3.58 | 7.8 | 12.6 | 8520 | 0.08 | 2.78 | 71609 | 14.3 | 513 | 7.0 | 0.059 | 0.86 | 0.28 | 1532 | 0.20 | 14.0 | 42 | | | | | | |
| 1860.0 | 13.25 | 0.85 | 0.01 | 5759 | 14.9 | 0.49 | 0.26 | 3.73 | 8.0 | 10.9 | 9013 | 0.28 | 3.91 | 76238 | 11.1 | 548 | 7.0 | 0.129 | 1.22 | 0.31 | 1636 | 0.24 | 14.8 | 44 | | | | | | |
| 1856.1 | 13.75 | 0.85 | 0.01 | 5576 | 14.5 | 0.43 | 0.23 | 3.50 | 7.7 | 10.2 | 8706 | 0.09 | 4.19 | 70966 | 11.8 | 524 | 6.5 | 0.176 | 1.09 | 0.29 | 1646 | 0.26 | 14.0 | 40 | | | | | | |
| 1852.1 | 14.25 | 0.86 | 0.01 | 5633 | 20.1 | 0.54 | 0.26 | 4.09 | 8.7 | 11.3 | 9222 | 0.07 | 4.75 | 70319 | 14.8 | 615 | 6.0 | 0.105 | 1.28 | 0.31 | 1693 | 0.26 | 15.8 | 47 | | | | | | |
| 1848.1 | 14.75 | 0.86 | 0.01 | 5221 | 18.0 | 0.45 | 0.45 | 3.87 | 8.8 | 7.7 | 8774 | 0.06 | 5.15 | 69500 | 15.2 | 575 | 6.2 | 0.290 | 1.44 | 0.28 | 1738 | 0.23 | 15.0 | 43 | | | | | | |
| 1844.2 | 15.25 | 0.87 | 0.01 | 5383 | 18.6 | 0.52 | 0.34 | 3.20 | 8.4 | 11.8 | 8665 | 0.34 | 5.32 | 67822 | 16.0 | 651 | 7.2 | 0.289 | 1.32 | 0.34 | 1739 | 0.28 | 15.8 | 46 | | | | | | |
| 1840.2 | 15.75 | 0.87 | 0.01 | 5696 | 17.4 | 0.45 | 0.25 | 3.59 | 7.9 | 11.6 | 8921 | 0.06 | 4.17 | 58074 | 15.5 | 589 | 7.1 | 0.162 | 1.56 | 0.29 | 1714 | 0.29 | 15.0 | 43 | | | | | | |
| 1836.3 | 16.25 | 0.87 | 0.01 | 5402 | 14.7 | 0.39 | 0.30 | 3.41 | 7.8 | 11.1 | 8536 | 0.05 | 3.33 | 61648 | 10.0 | 534 | 7.9 | 0.092 | 1.27 | 0.32 | 1736 | 0.43 | 15.5 | 42 | | | | | | |

Appendix 1b. Gilbert Bay (Site 4) metal concentrations in the sediment core from the Great Salt Lake. These concentrations are corrected for the amount of salt in the interstitial water, and thus represent concentrations per gram of dry sediment. To calculate the actual values measured in the dried sediment-salt mixture, multiply concentrations by the value "Sediment (fraction of dry wt)".

| 210Pb Year (estimate) | Core Mid-Depth | Sediment (fraction of dry wt) | Salt-Corrected Concentration ($\mu\text{g g}^{-1}$ dry sediment; ppm) | | | | | | | | | | | | | | | | | | | | | | | | | | | |
|-----------------------|----------------|-------------------------------|--|------|------|------|------|------|------|-------|-------|------|-------|--------|------|------|-------|-------|------|------|------|------|------|-----|--|--|--|--|--|--|
| | | | Ag | Al | As | Be | Cd | Co | Cr | Cu | Fe | Hg | Mo | Na | Ni | P | Pb | Sb | Se | Sn | Sr | Tl | V | Zn | | | | | | |
| 2002.0 | 0.25 | 0.50 | 0.08 | 5260 | 41.8 | 0.42 | 2.24 | 4.03 | 11.1 | 193.8 | 8008 | 0.20 | 15.63 | 320824 | 14.0 | 1429 | 54.7 | 0.775 | 5.20 | 1.49 | 458 | 0.87 | 18.7 | 119 | | | | | | |
| 1994.8 | 0.75 | 0.57 | 0.05 | 5942 | 51.7 | 0.53 | 2.51 | 4.56 | 11.8 | 219.3 | 9171 | 0.63 | 18.08 | 247103 | 14.5 | 1458 | 61.8 | 0.510 | 4.68 | 1.81 | 471 | 0.81 | 20.6 | 131 | | | | | | |
| 1983.2 | 1.25 | 0.60 | 0.02 | 5867 | 49.7 | 0.44 | 2.53 | 4.51 | 12.0 | 233.4 | 8835 | 0.23 | 19.87 | 223932 | 14.7 | 1414 | 62.3 | 0.503 | 4.76 | 1.79 | 501 | 0.79 | 20.9 | 134 | | | | | | |
| 1971.3 | 1.75 | 0.57 | 0.40 | 6024 | 41.2 | 0.53 | 3.22 | 5.15 | 15.0 | 295.5 | 9253 | 0.91 | 44.88 | 245307 | 18.6 | 1391 | 84.1 | 0.897 | 4.24 | 2.27 | 564 | 1.13 | 25.2 | 155 | | | | | | |
| 1963.4 | 2.25 | 0.62 | 0.58 | 7078 | 32.5 | 0.55 | 5.12 | 6.15 | 19.6 | 450.6 | 11280 | 0.40 | 76.11 | 217078 | 20.9 | 1367 | 150.8 | 1.352 | 4.86 | 3.33 | 625 | 1.51 | 28.6 | 181 | | | | | | |
| 1955.5 | 2.75 | 0.67 | 0.47 | 7698 | 40.3 | 0.64 | 9.74 | 6.81 | 21.7 | 490.0 | 12694 | 0.49 | 78.36 | 154783 | 21.1 | 1297 | 262.1 | 1.558 | 5.52 | 4.33 | 926 | 1.81 | 29.1 | 208 | | | | | | |
| 1946.0 | 3.25 | 0.73 | 0.47 | 6512 | 38.4 | 0.56 | 7.27 | 5.75 | 16.3 | 369.1 | 10758 | 0.66 | 44.73 | 106439 | 18.5 | 1117 | 204.0 | 1.112 | 3.64 | 3.53 | 1771 | 1.47 | 23.4 | 211 | | | | | | |
| 1936.5 | 3.75 | 0.78 | 0.46 | 6873 | 39.2 | 0.57 | 9.54 | 6.37 | 19.8 | 493.3 | 11395 | 0.42 | 67.40 | 123155 | 19.4 | 1167 | 242.7 | 0.294 | 4.46 | 4.23 | 906 | 1.65 | 25.3 | 237 | | | | | | |
| 1928.8 | 4.25 | 0.84 | 0.31 | 3490 | 23.0 | 0.26 | 5.25 | 2.57 | 6.9 | 204.7 | 5537 | 0.27 | 11.92 | 72676 | 9.6 | 607 | 112.6 | 0.072 | 1.60 | 1.16 | 1075 | 0.66 | 11.3 | 90 | | | | | | |
| 1921.1 | 4.75 | 0.89 | 0.24 | 2647 | 18.6 | 0.21 | 1.68 | 1.97 | 5.2 | 164.1 | 4375 | 0.31 | 9.16 | 41223 | 6.9 | 391 | 62.7 | 0.325 | 1.21 | 0.89 | 854 | 0.58 | 8.6 | 73 | | | | | | |
| 1915.4 | 5.25 | 0.88 | 0.21 | 3806 | 19.5 | 0.30 | 2.62 | 2.79 | 6.6 | 247.8 | 6240 | 0.34 | 7.02 | 56260 | 9.1 | 543 | 124.5 | 0.103 | 1.68 | 1.15 | 1068 | 0.54 | 11.6 | 111 | | | | | | |
| 1909.7 | 5.75 | 0.86 | 0.21 | 4976 | 18.1 | 0.40 | 2.06 | 3.71 | 7.9 | 326.5 | 8211 | 0.48 | 8.41 | 62339 | 13.9 | 688 | 156.4 | 0.198 | 1.62 | 1.44 | 1273 | 0.62 | 14.1 | 172 | | | | | | |
| 1903.1 | 6.25 | 0.86 | 0.24 | 4829 | 13.6 | 0.39 | 1.73 | 3.37 | 7.0 | 167.1 | 7543 | 0.38 | 5.61 | 58897 | 11.8 | 600 | 157.1 | 0.081 | 1.31 | 0.89 | 1311 | 0.45 | 13.1 | 128 | | | | | | |
| 1896.5 | 6.75 | 0.87 | 0.71 | 5792 | 13.2 | 0.51 | 1.48 | 4.06 | 8.3 | 104.7 | 9397 | 0.44 | 5.15 | 56261 | 13.2 | 668 | 163.1 | 0.058 | 1.28 | 0.80 | 1778 | 0.43 | 15.7 | 129 | | | | | | |
| 1889.6 | 7.25 | 0.87 | 0.16 | 5288 | 7.4 | 0.37 | 0.56 | 3.31 | 7.1 | 19.8 | 9001 | 0.28 | 4.33 | 56751 | 10.0 | 544 | 61.3 | 0.058 | 1.01 | 0.37 | 2031 | 0.26 | 13.3 | 55 | | | | | | |
| 1882.7 | 7.75 | 0.87 | 0.13 | 5499 | 8.5 | 0.45 | 0.45 | 3.60 | 7.8 | 19.9 | 9073 | 0.18 | 3.80 | 57672 | 16.2 | 595 | 31.2 | 0.069 | 1.33 | 0.39 | 1817 | 0.29 | 15.1 | 48 | | | | | | |
| 1874.6 | 8.25 | 0.87 | 0.06 | 6637 | 10.5 | 0.49 | 0.34 | 4.11 | 9.4 | 18.1 | 10231 | 0.14 | 2.39 | 58025 | 14.4 | 645 | 14.7 | 0.034 | 1.39 | 0.38 | 1188 | 0.30 | 16.7 | 52 | | | | | | |
| 1866.4 | 8.75 | 0.87 | 0.14 | 6353 | 12.0 | 0.52 | 0.34 | 4.18 | 9.0 | 15.8 | 10555 | 0.10 | 3.51 | 58088 | 16.3 | 619 | 10.9 | 0.057 | 1.15 | 0.37 | 1392 | 0.37 | 15.1 | 51 | | | | | | |
| 1845.6 | 9.25 | 0.87 | 0.02 | 6051 | 12.7 | 0.47 | 0.37 | 3.91 | 8.4 | 13.8 | 10236 | 0.19 | 2.85 | 59084 | 13.1 | 568 | 11.0 | 0.114 | 1.05 | 0.31 | 1578 | 0.33 | 14.3 | 47 | | | | | | |
| 1824.9 | 9.75 | 0.88 | 0.02 | 6055 | 13.3 | 0.50 | 0.31 | 3.53 | 8.3 | 13.2 | 10083 | 0.10 | 2.84 | 56427 | 13.3 | 570 | 8.3 | 0.125 | 0.95 | 0.30 | 1555 | 0.79 | 13.5 | 47 | | | | | | |
| 1810.1 | 10.25 | 0.88 | 0.03 | 5762 | 13.5 | 0.44 | 0.25 | 3.36 | 7.8 | 11.4 | 9508 | 0.05 | 1.77 | 52081 | 10.2 | 554 | 7.5 | 0.045 | 0.88 | 0.27 | 1859 | 0.36 | 13.0 | 41 | | | | | | |
| 1795.3 | 10.75 | 0.88 | 0.01 | 5039 | 11.7 | 0.48 | 0.22 | 3.14 | 7.0 | 9.2 | 8628 | 0.03 | 2.31 | 52467 | 9.1 | 507 | 5.7 | 0.113 | 0.83 | 0.28 | 1832 | 0.73 | 13.1 | 46 | | | | | | |
| 1770.1 | 11.25 | 0.89 | 0.01 | 5346 | 12.3 | 0.46 | 0.25 | 3.22 | 7.3 | 9.9 | 9001 | 0.03 | 2.88 | 51011 | 9.7 | 527 | 6.5 | 0.056 | 0.86 | 0.25 | 1704 | 0.41 | 13.1 | 39 | | | | | | |
| 1745.0 | 11.75 | 0.89 | 0.01 | 5670 | 12.9 | 0.53 | 0.28 | 3.50 | 7.6 | 10.8 | 8961 | 0.09 | 2.62 | 53068 | 9.5 | 578 | 7.6 | 0.056 | 0.95 | 0.28 | 1799 | 0.53 | 14.2 | 42 | | | | | | |
| 1719.9 | 12.25 | 0.89 | 0.01 | 5515 | 12.8 | 0.47 | 0.31 | 3.04 | 6.7 | 10.6 | 7922 | 0.04 | 2.70 | 45537 | 8.9 | 522 | 7.7 | 0.056 | 0.91 | 0.27 | 1903 | 0.54 | 13.1 | 61 | | | | | | |

Appendix 1c. Farmington Bay (Site 1) metal concentrations in the sediment core from the Great Salt Lake. These concentrations are corrected for the amount of salt in the interstitial water, and thus represent concentrations per gram of dry sediment. To calculate the actual values measured in the dried sediment+salt mixture, multiply concentrations by the value "Sediment (fraction of dry wt)".

| 210Pb Year (estimate) | Core Mid-Depth | Sediment (fraction of dry wt) | Salt-Corrected Concentration ($\mu\text{g g}^{-1}$ dry sediment; ppm) | | | | | | | | | | | | | | | | | | | | | | | | | | | |
|-----------------------|----------------|-------------------------------|--|------|------|------|------|------|------|-------|-------|------|-------|-------|------|------|-------|-------|------|------|------|------|------|-----|--|--|--|--|--|--|
| | | | Ag | Al | As | Be | Ca | Co | Cr | Cu | Fe | Hg | Mo | Na | Ni | P | Pb | Sb | Se | Sn | Sr | Tl | V | Zn | | | | | | |
| 2004.0 | 0.25 | 0.91 | 2.69 | 5774 | 19.4 | 0.63 | 2.98 | 5.67 | 37.4 | 155.1 | 9751 | 0.90 | 1.19 | 6358 | 13.4 | 2897 | 134.8 | 0.044 | 2.52 | 2.93 | 1358 | 0.44 | 22.4 | 283 | | | | | | |
| 2002.4 | 0.75 | 0.93 | 1.06 | 5238 | 18.0 | 0.57 | 2.55 | 5.24 | 33.2 | 135.2 | 9059 | 0.85 | 1.21 | 4952 | 12.0 | 2273 | 124.9 | 0.032 | 2.20 | 2.59 | 1314 | 0.42 | 21.2 | 246 | | | | | | |
| 2000.8 | 1.50 | 0.95 | 2.41 | 5463 | 18.7 | 0.57 | 2.82 | 5.57 | 35.5 | 162.5 | 9111 | 0.86 | 0.98 | 4887 | 12.8 | 2154 | 126.9 | 0.042 | 2.27 | 2.59 | 1309 | 0.43 | 23.0 | 251 | | | | | | |
| 1998.0 | 1.75 | 0.95 | | | | | | | | | | | | | | | | | | | | | | | | | | | | |
| 1995.2 | 2.25 | 0.95 | 0.81 | 5446 | 18.6 | 0.57 | 2.49 | 5.18 | 32.4 | 138.0 | 9090 | 0.85 | 1.39 | 4540 | 11.9 | 2086 | 124.7 | 0.032 | 2.17 | 2.54 | 1350 | 0.42 | 21.9 | 240 | | | | | | |
| 1992.3 | 2.75 | 0.95 | 2.36 | 5508 | 20.9 | 0.57 | 2.73 | 5.50 | 35.4 | 136.0 | 9377 | 0.87 | 2.84 | 4198 | 12.6 | 2098 | 113.7 | 0.042 | 2.33 | 2.51 | 1302 | 0.46 | 22.8 | 258 | | | | | | |
| 1988.7 | 3.25 | 0.96 | 2.11 | 5511 | 22.7 | 0.58 | 2.61 | 5.49 | 34.9 | 140.8 | 9470 | 0.84 | 2.95 | 3795 | 12.7 | 1933 | 110.3 | 0.031 | 2.22 | 2.45 | 1337 | 0.46 | 22.9 | 251 | | | | | | |
| 1985.1 | 3.75 | 0.96 | 2.28 | 4990 | 21.2 | 0.52 | 2.56 | 4.77 | 31.5 | 144.3 | 8933 | 0.83 | 2.27 | 3308 | 11.2 | 1701 | 121.9 | 0.062 | 2.30 | 2.53 | 1402 | 0.44 | 21.6 | 240 | | | | | | |
| 1981.6 | 4.25 | 0.97 | 2.19 | 5106 | 21.9 | 0.55 | 2.61 | 4.94 | 33.1 | 153.9 | 9031 | 0.82 | 2.65 | 3582 | 11.7 | 1663 | 123.1 | 0.031 | 2.34 | 2.57 | 1548 | 0.42 | 22.1 | 230 | | | | | | |
| 1978.0 | 4.75 | 0.97 | 2.32 | 4932 | 19.7 | 0.55 | 2.41 | 4.57 | 28.7 | 145.1 | 8608 | 0.81 | 14.67 | 3686 | 10.6 | 1504 | 121.3 | 0.031 | 2.04 | 2.47 | 1480 | 0.41 | 20.7 | 233 | | | | | | |
| 1974.2 | 5.25 | 0.97 | 2.23 | 4923 | 21.1 | 0.54 | 2.52 | 4.79 | 31.0 | 139.7 | 8700 | 0.86 | 3.61 | 3880 | 11.4 | 1565 | 121.5 | 0.031 | 2.27 | 2.72 | 1427 | 0.42 | 21.7 | 230 | | | | | | |
| 1970.4 | 5.75 | 0.98 | 2.41 | 4891 | 21.2 | 0.53 | 2.72 | 5.14 | 36.8 | 143.1 | 8867 | 0.89 | 2.32 | 3636 | 12.2 | 1418 | 118.8 | 0.031 | 1.99 | 3.27 | 1391 | 0.41 | 24.1 | 234 | | | | | | |
| 1966.7 | 6.25 | 0.98 | 1.52 | 5252 | 21.7 | 0.58 | 2.86 | 5.28 | 39.2 | 145.8 | 9633 | 0.84 | 2.41 | 4008 | 12.8 | 1468 | 118.9 | 0.031 | 2.00 | 3.12 | 1371 | 0.42 | 24.4 | 256 | | | | | | |
| 1962.9 | 6.75 | 0.98 | 1.50 | 5916 | 21.9 | 0.67 | 3.58 | 6.03 | 51.6 | 152.3 | 10663 | 1.09 | 4.05 | 3787 | 15.1 | 1291 | 145.1 | 0.041 | 2.19 | 3.79 | 1386 | 0.48 | 26.7 | 296 | | | | | | |
| 1961.6 | 7.25 | 0.98 | 1.04 | 5244 | 23.8 | 0.55 | 2.96 | 4.95 | 30.1 | 148.7 | 9441 | 1.01 | 6.96 | 3528 | 12.0 | 991 | 151.6 | 0.051 | 1.68 | 3.38 | 1419 | 0.66 | 19.7 | 296 | | | | | | |
| 1960.3 | 7.75 | 0.98 | 1.68 | 4886 | 27.0 | 0.46 | 3.03 | 4.85 | 14.9 | 193.7 | 8840 | 0.81 | 4.69 | 3886 | 10.0 | 843 | 191.4 | 0.061 | 1.89 | 3.01 | 1241 | 0.71 | 14.1 | 351 | | | | | | |
| 1958.9 | 8.25 | 0.99 | 0.21 | 5109 | 19.5 | 0.47 | 1.40 | 4.48 | 9.5 | 134.7 | 9054 | 0.51 | 4.27 | 3888 | 9.5 | 807 | 138.0 | 0.041 | 1.50 | 0.98 | 1297 | 0.30 | 12.0 | 261 | | | | | | |
| 1957.6 | 8.75 | 0.98 | 0.38 | 5467 | 21.0 | 0.52 | 0.98 | 4.69 | 9.6 | 121.5 | 9567 | 0.84 | 3.62 | 4467 | 10.2 | 742 | 112.5 | 0.040 | 1.65 | 0.48 | 1189 | 0.21 | 11.5 | 219 | | | | | | |
| 1956.0 | 9.25 | 0.99 | 2.32 | 4986 | 22.0 | 0.46 | 0.54 | 4.06 | 8.1 | 76.9 | 8776 | 0.27 | 2.65 | 4467 | 8.9 | 719 | 71.1 | 0.020 | 1.43 | 0.24 | 1306 | 0.14 | 11.1 | 148 | | | | | | |
| 1954.5 | 9.75 | 0.98 | 0.08 | 4433 | 22.2 | 0.42 | 0.17 | 4.59 | 7.3 | 15.9 | 7841 | 0.22 | 2.54 | 4593 | 8.7 | 631 | 22.8 | 0.030 | 1.55 | 0.11 | 1381 | 0.15 | 12.0 | 54 | | | | | | |
| 1952.9 | 10.25 | 0.98 | 0.01 | 5839 | 33.9 | 0.50 | 0.27 | 4.99 | 9.0 | 12.1 | 9038 | 0.11 | 5.43 | 4968 | 10.3 | 705 | 13.0 | 0.031 | 1.47 | 0.12 | 1050 | 0.26 | 13.8 | 64 | | | | | | |
| 1951.4 | 10.75 | 0.98 | 0.01 | 7462 | 56.4 | 0.63 | 0.31 | 4.78 | 9.8 | 10.5 | 11162 | 0.06 | 8.23 | 7190 | 11.5 | 769 | 8.6 | 0.031 | 1.50 | 0.18 | 907 | 0.24 | 14.3 | 67 | | | | | | |
| 1948.9 | 11.25 | 0.98 | 0.01 | 6570 | 51.8 | 0.49 | 0.35 | 4.15 | 8.6 | 12.1 | 10031 | 0.05 | 5.77 | 6334 | 10.0 | 713 | 11.6 | 0.020 | 1.03 | 0.22 | 955 | 0.22 | 13.2 | 64 | | | | | | |
| 1946.5 | 11.75 | 0.98 | 0.01 | 6696 | 53.9 | 0.59 | 0.41 | 4.21 | 8.6 | 12.9 | 10332 | 0.05 | 4.27 | 6813 | 10.2 | 774 | 14.3 | 0.031 | 1.79 | 0.29 | 925 | 0.29 | 13.1 | 67 | | | | | | |
| 1944.0 | 12.25 | 0.98 | 0.01 | 6393 | 50.0 | 0.58 | 0.32 | 4.28 | 9.0 | 12.7 | 10043 | 0.05 | 2.52 | 7859 | 10.3 | 713 | 12.7 | 0.020 | 0.76 | 0.28 | 902 | 0.31 | 12.9 | 64 | | | | | | |
| 1938.5 | 13.25 | 0.98 | 0.01 | 6380 | 32.5 | 0.66 | 0.20 | 4.18 | 9.1 | 9.9 | 10277 | 0.02 | 0.74 | 9635 | 9.9 | 705 | 6.2 | 0.020 | 0.91 | 0.19 | 1074 | 0.18 | 12.7 | 50 | | | | | | |
| 1932.3 | 14.25 | 0.98 | 0.01 | 6510 | 21.6 | 0.60 | 0.18 | 4.16 | 9.1 | 9.7 | 10990 | 0.01 | 0.42 | 9797 | 9.8 | 658 | 5.7 | 0.010 | 0.80 | 0.21 | 1081 | 0.17 | 12.7 | 50 | | | | | | |
| 1931.4 | 15.25 | 0.98 | 0.01 | 6107 | 19.4 | 0.57 | 0.18 | 4.17 | 9.3 | 9.8 | 9840 | 0.01 | 0.41 | 10189 | 9.8 | 687 | 5.5 | 0.010 | 0.77 | 0.22 | 1043 | 0.17 | 12.8 | 49 | | | | | | |
| 1930.5 | 16.25 | 0.98 | 0.01 | 6520 | 18.5 | 0.59 | 0.18 | 4.42 | 9.5 | 9.9 | 10487 | 0.01 | 0.34 | 10892 | 10.2 | 681 | 5.6 | 0.005 | 0.86 | 0.14 | 984 | 0.17 | 13.3 | 52 | | | | | | |
| 1922.9 | 17.25 | 0.98 | 0.01 | 6906 | 18.1 | 0.58 | 0.19 | 4.60 | 9.6 | 9.8 | 11201 | 0.01 | 0.46 | 11273 | 10.7 | 739 | 5.5 | 0.010 | 0.77 | 0.27 | 1000 | 0.18 | 12.9 | 54 | | | | | | |
| 1915.3 | 18.25 | 0.98 | 0.01 | 7155 | 15.5 | 0.57 | 0.19 | 4.55 | 9.6 | 10.7 | 11698 | 0.01 | 0.39 | 10601 | 10.7 | 745 | 6.0 | 0.010 | 0.76 | 0.18 | 888 | 0.18 | 12.2 | 53 | | | | | | |
| 1911.6 | 19.25 | 0.99 | 0.09 | 5447 | 13.0 | 0.58 | 0.18 | 4.46 | 9.2 | 9.6 | 9450 | 0.01 | 0.44 | 9721 | 10.0 | 671 | 4.8 | 0.020 | 0.63 | 0.10 | 892 | 0.17 | 13.8 | 48 | | | | | | |
| 1907.9 | 20.25 | 0.99 | 0.01 | 4112 | 10.8 | 0.36 | 0.11 | 3.69 | 6.6 | 6.7 | 7458 | 0.01 | 0.80 | 10034 | 8.0 | 514 | 4.0 | 0.010 | 0.63 | 0.07 | 1537 | 0.09 | 11.5 | 35 | | | | | | |
| 1904.2 | 21.25 | 0.99 | 0.09 | 4937 | 25.1 | 0.49 | 0.21 | 4.23 | 8.0 | 8.1 | 8742 | 0.02 | 3.78 | 9090 | 8.9 | 622 | 5.9 | 0.020 | 0.84 | 0.12 | 948 | 0.24 | 12.0 | 44 | | | | | | |
| 1897.3 | 22.25 | 0.99 | 0.01 | 5854 | 41.3 | 0.51 | 0.19 | 4.43 | 9.2 | 8.7 | 10193 | 0.01 | 2.24 | 10495 | 9.3 | 697 | 5.5 | 0.020 | 0.85 | 0.09 | 929 | 0.16 | 12.0 | 44 | | | | | | |
| 1890.3 | 23.25 | 0.99 | 0.01 | 5950 | 22.3 | 0.63 | 0.19 | 4.57 | 9.7 | 8.7 | 9775 | 0.01 | 0.79 | 11639 | 10.0 | 657 | 5.4 | 0.020 | 0.60 | 0.20 | 853 | 0.17 | 12.4 | 48 | | | | | | |
| 1883.4 | 24.25 | 0.99 | 0.01 | 5970 | 16.7 | 0.55 | 0.15 | 4.45 | 9.1 | 7.1 | 9800 | 0.01 | 0.33 | 11332 | 9.3 | 662 | 3.9 | 0.010 | 0.73 | 0.09 | 966 | 0.12 | 11.3 | 44 | | | | | | |
| 1876.4 | 25.25 | 0.99 | 0.06 | 3356 | 14.3 | 0.38 | 0.09 | 3.49 | 8.2 | 4.8 | 8299 | 0.01 | 1.37 | 11630 | 8.8 | 506 | 4.3 | 0.010 | 0.75 | 0.09 | 2064 | 0.08 | 10.9 | 31 | | | | | | |
| 1871.0 | 26.25 | 0.99 | 0.02 | 5421 | 15.5 | 0.43 | 0.16 | 4.08 | 8.6 | 7.4 | 8847 | 0.01 | 0.77 | 11630 | 8.8 | 566 | 4.2 | 0.010 | 0.59 | 0.14 | 1010 | 0.13 | 11.5 | 43 | | | | | | |
| 1865.7 | 27.25 | 0.99 | 0.02 | 4858 | 18.7 | 0.45 | 0.32 | 5.83 | 8.3 | 14.4 | 8291 | 0.02 | 1.30 | 12148 | 8.5 | 605 | 9.1 | 0.020 | 0.84 | 0.13 | 1011 | 0.38 | 20.8 | 60 | | | | | | |
| 1860.3 | 28.25 | 0.99 | 0.01 | 5716 | 50.4 | 0.53 | 0.33 | 4.47 | 9.8 | 20.3 | 10094 | 0.02 | 3.02 | 11063 | 9.7 | 714 | 11.9 | 0.030 | 1.16 | 0.10 | 513 | 0.33 | 12.4 | 48 | | | | | | |
| 1854.9 | 29.25 | 0.99 | 0.01 | 6829 | 44.3 | 0.60 | 0.38 | 4.84 | 11.0 | 24.9 | 12796 | 0.04 | 3.72 | 13269 | 10.4 | 632 | 13.8 | 0.051 | 0.88 | 0.26 | 834 | 0.28 | 14.4 | 55 | | | | | | |
| 1849.5 | 30.25 | 0.99 | 0.05 | 5785 | 27.7 | 0.54 | 0.23 | 4.27 | 9.8 | 15.8 | 10502 | 0.01 | 1.72 | 11337 | 9.5 | 571 | 8.8 | 0.030 | 0.75 | 0.13 | 1052 | 0.21 | 13.4 | 51 | | | | | | |
| 1844.1 | 31.25 | 0.99 | 0.01 | 6181 | 23.6 | 0.51 | 0.24 | 4.68 | 9.9 | 12.0 | 10779 | 0.01 | 2.54 | 13437 | 10.2 | 632 | 8.6 | 0.020 | 0.88 | 0.17 | 831 | 0.27 | 14.4 | 58 | | | | | | |
| 1838.7 | 32.25 | 0.99 | 0.01 | 6280 | 34.4 | 0.59 | 0.21 | 4.30 | 9.4 | 11.1 | 11208 | 0.02 | 1.61 | 14426 | 9.8 | 616 | 7.5 | 0.030 | 0.77 | 0.14 | 984 | 0.21 | 13.7 | 53 | | | | | | |

Appendix 2. Comparison of the chronologies of metal contaminants (copper, mercury, lead), with an indicator of fluvial transport of eroded material (aluminum) at Gilbert Bay Site 3 (A), Gilbert Bay Site 4 (B), and Farmington Bay Site 1 (C). The plotted values are the 3-point running means of the ratio of a metal at a given depth relative to the mean pre-industrial concentration (pre-1860).

