

Historical Changes in Aquatic Invertebrates and Charophytes of the Great Salt Lake, Utah
(USA): The Effects of Wastewater Inputs, Water Diversions and Barriers

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Executive Summary

To assess how the Great Salt Lake has changed over the last 150 years following settlement, sediment cores were retrieved from Farmington and Gilbert Bays. One core was collected at the southern end of Farmington Bay and thus characterized the environment at the interface between the Jordan River delta and the open water. The record of invertebrates, crustaceans and aquatic plants indicates that before major anthropogenic disturbances in the mid-1880s this site was mesotrophic. Charophyte meadows were present until the 1950s, which created a habitat favorable for a wide variety of chironomids and ostracods. In the early 1900s, when raw sewage from Salt Lake City began to be discharged in increasing amounts to the bay the system became more eutrophic and the abundance of invertebrates tolerant of hypoxic conditions increased. By 1940-50 the charophyte meadows, which are effective nutrient sinks, had disappeared and the insect community was dominated by species tolerant of organic pollution and a salt-tolerant chironomid that indicates that either increasing nutrients or salinities could have played a role in the near disappearance of Charophytes in Farmington Bay. Increasing populations and concomitant increases in sewage inputs, a shift to secondary treatment of sewage in the mid-1960s, and construction of the Automobile Causeway that reduced water exchanges, have all likely exacerbated eutrophication at this site. Diversion of water via the Surplus Canal also has decreased flushing of the bay during spring runoff. Since the 1960s the bay has remained eutrophic with dominance by relatively few species of nutrient-tolerant chironomids and air-breathing invertebrates.

In Gilbert Bay, higher salinities have precluded the presence of many of the invertebrates found in Farmington Bay. Only remains of *Artemia* (brine shrimp) and *Ephydra* (brine flies) were recorded in Gilbert Bay cores; and these have increased markedly since ~1950 when railway construction allowed salinities in the south arm to decrease. Although the low abundances of *Artemia* and *Ephydra* prior to 1950 may represent poor preservation or reduced sedimentation of these organisms in the dense brine, it is more likely that higher salinities prior to this time resulted in decreased populations.

Introduction

Excess nutrients discharged into lakes and estuaries can cause eutrophication, defined as an excessive production of algae relative to natural or background conditions. This excess production can cause a number of water quality problems including de-oxygenation of the water column, taste and odor problems (Bell and Wurtsbaugh 2007) and production of toxic algal blooms (Schindler 2006). Algal-associated toxins can kill birds, livestock, and dogs, as well as cause liver dysfunction, gastric distress, and possibly cancer (Murphy et al. 2003). On the other hand, eutrophication can also increase ecosystem productivity and favor production of commercially-important organisms such as fish or invertebrates, including brine shrimp and flies, which also support avian production. The latter is of particular interest with regard to Farmington and Bear River bays of Great Salt Lake (GSL), Utah (Figure 1), both of which host large populations of shorebirds, waterfowl and other avian taxa (Paul and Manning 2002).

Eutrophication processes in the Great Salt Lake (GSL) may be particularly complex as the lake is divided by several causeways, which restrict natural hydrologic circulation (Figure 1; Table 1). In particular, impoundment of individual embayments may influence eutrophication by reducing circulation, isolating contaminants, and altering natural salinities

in individual sub-basins. For example, Farmington and Bear River bays are shallow and receive substantial river inflows that dilute salts to near-freshwater levels during spring runoff. However, as those flows subside, evaporation and intrusion of salts from adjoining bays can increase salinities. Presently, Farmington Bay can reach salinities of 90 g/L which is 2½ times saltier than the ocean (35 g/L), whereas those of Bear River Bay can be even higher (Wurtsbaugh and Marcarelli 2006). Currently, Gilbert Bay has a salinity of (140 g/L), although during the floods of 1984-85, salinities decreased to 50 g/L (Figure 2). Gunnison Bay receives its water primarily from Gilbert Bay and often evaporates to the point that salts precipitate out of the water column. As a result, salinity in that basin is nearly 300 g/L. Despite these differences, the Beneficial Uses designated by the State of Utah are similar for all bays and are defined as, “Protected for infrequent or frequent primary and secondary contact recreation, waterfowl, shore birds and other water-oriented wildlife including their necessary food chain”.

Some basins of Great Salt Lake are experiencing symptoms of severe cultural eutrophication (Wurtsbaugh and Marcarelli 2006), likely reflecting multiple sources of human-derived nutrients. For example, GSL receives wastewaters from 1.4 million people in the greater metropolitan area of Salt Lake City, and additional pollutants enter from diffuse or non-point sources associated with the Jordan, Bear and Ogden/Weber river systems (Baskin et al. 2002). Repeated analysis of Farmington Bay water has shown that it is characterized by extremely high nutrient concentrations and frequent severe algal blooms (Wurtsbaugh and Marcarelli 2006, DWQ STORET database). Nutrient levels are also high in Gilbert and Bear River bays (Wurtsbaugh et al. 2006), but the degree to which this is due to nutrient inputs, other human activities, or the natural concentrating effect of water evaporation is unknown.

Eutrophication and salinity interact to control the organisms that survive in GSL, and this interaction may add complexity to the mechanisms degrading water quality in individual embayments. For example, Gilbert Bay has a limited diversity of phytoplankton (algae in the water column) and periphytic (bottom-dwelling) algae, and includes only two metazoans—brine shrimp (*Artemia*) and brine flies (*Ephydra*). Similarly, the salt-saturated waters of Gunnison Bay support only a few types of algae, bacteria and Archaea (a bacteria-like organism), and presently includes very few invertebrates. In addition, the high spatial and temporal variability of salinities in Farmington and Bear River bays may cause significant changes in the biotic composition throughout the year. For example, fish are present and biotic diversity (algae, invertebrates) is high in both bays during the period of maximum spring runoff. However, as summer progresses, evaporation increases lake-water salinity, and toxic algae such as the cyanobacterium (blue-green algae) *Nodularia spumigena* can grow in profusion. Furthermore, decomposition of algal blooms in Farmington Bay reduces oxygen content of sediments and overlying water, resulting in inhospitable conditions for aquatic life (Wurtsbaugh and Marcarelli 2006). Due to the complexity of historical changes, and the absence of long-term records of invertebrate community structure, it has been difficult to evaluate how degradation of water quality, ecosystem management and natural climatic variability may interact to affect aquatic invertebrate community composition and production.

The objective of this study was to quantify historical changes in invertebrate abundance and community composition using well-preserved remains of select taxa found in sediments of GSL. In other lakes around the world, sedimentary concentrations of well-preserved invertebrate remains are correlated strongly with measured historical changes in past invertebrate densities (e.g., Leavitt et al. 1989, 1993; Hann et al. 1994). Such fossil

assemblages, however, include species not commonly collected by routine limnological monitoring (e.g., benthic taxa such as chironomids), and frequently lack fossils from taxa with thin exoskeletons or those with low chitin content (e.g., rotifers, copepods). Less is known of the degree of preservation of fossil invertebrates from benthic habitats or those in saline lakes, although remains of sedimentary invertebrates (Cladocera, chironomids, *Artemia*) have proven particularly useful in reconstruction of historical changes in lake level, salinity, oxygen content, production and former food-web structure (Spencer et al. 1984, Verschuren et al. 1999a, 1999b, 2002; Quinlan et al. 2002). Furthermore, quantification of invertebrate species composition along a gradient of saline lakes demonstrates that climate and associated changes in lake water chemistry (salinity, nutrient content) are paramount controls of community composition and abundance (Wissel et al. 2011). Of the nine sites initially explored in GSL, quantitative analysis of past invertebrate communities was attempted at one location in Farmington Bay and two sites in Gilbert Bay where reliable sediment chronologies had been established (Figure 1) (see Leavitt et al. 2012).

Methods

Study Area

Great Salt Lake is a large terminal lake located in northern Utah (USA). The Salt Lake City greater metropolitan area borders much of the southeastern and eastern side of the lake (Figure 1). Historically, the lake has had an area that ranged from 2,470 km² to 5,490 km² and a hydrologic watershed of greater than 89,000 km² (Belovsky et al. 2011). The lake receives riverine input from the Jordan, Weber, and Bear rivers, which all enter on the east side of the basin. Such freshwater inflow serves to both dilute lake waters, and transport solutes including nutrients, dissolved salts, and contaminants, which become concentrated in the closed-basin as water is lost through evaporation. Since the 1850s maximum lake depth has ranged from 7.6 m to 13.7 m with elevations of 1277 m to 1284 m (Belovsky et al. 2011). GSL is shallow for its size, with a maximum depth of 10.1 m at the historical mean lake elevation. Similar to other hydrologically-closed sites, lake depth and salinity are highly variable (Figure 2) and fluctuate in response to variations in precipitation and evaporation. In particular, lake levels decrease and salinity increases when effective moisture (precipitation - evaporation) declines. Overall, lake levels have varied ~ 6 m since the 1800s, whereas salinity has ranged between 60 g/L and > 250 g/L in the main lake (Figure 2). At present, Farmington Bay exhibits hypereutrophic conditions, and the entire lake has elevated concentrations of nutrients (Belovsky et al. 2011).

Although humans have occupied the Great Salt Lake basin for 1000s of years (Madsen 1980), there is little evidence of human impacts prior to the arrival of ancestral Europeans in 1847 (Adler 1999). Since that time, the lake and its watershed have been subjected to profound modifications (Table 1) including the Surplus Canal in 1885, which diverted a portion of the Jordan River from Farmington Bay to Gilbert Bay, thus diminishing flows of water to Farmington Bay. The installation of a permeable railway causeway (1902) that crossed Bear River Bay and the main body of the lake, and its replacement with an impermeable barrier in 1959, divided the lake into a southern Gilbert Bay and a northern Gunnison Bay (Stephens 1998). In 1952, an automobile causeway was completed connecting the southern shore of Antelope Island with the mainland, whereas a second causeway partially enclosed Farmington Bay in 1969 by linking northern Antelope Island with the east side of the GSL basin. Construction of these structures has modified hydrologic exchanges

among basins within GSL, and effectively divided the lake into four quadrants of differing salinities, ranging from relatively freshwaters to hypersaline: 1) Bear River and Willard Bay; 2) Farmington Bay; 3) Gilbert Bay, and 4) Gunnison Bay (Figure 1). Separation of the bays, and differential inflows of nutrients, has also resulted in different trophic states: Gilbert and Bear River Bay are mesotrophic and Farmington Bay is hypereutrophic (Wurtsbaugh and Marcarelli 2006; Wurtsbaugh, unpublished data, 6). As noted in Leavitt et al. (2012), construction of these barriers modified algal production and chemical characteristic in individual basins.

Land clearance, agriculture and mining began almost immediately following Mormon settlement. Most notably, the on-going Bingham Mine began operating in 1863 and smelting operations commenced by 1892. Populations along the Wasatch Front expanded rapidly, reaching 1.5 million by 1992 and forecast to reach ~5 million by 2050 (Quality Growth Efficiency Tools Technical Committee, 2020). Such population expansion increased urban development in the GSL watershed, and introduced wastewater into the lake (Figure 1, Table 1). Sewer lines to the Jordan River were initiated by 1889, resulting in rapid increases in wastewater loading ($\sim 52 \times 10^6 \text{ L d}^{-1}$ by 1908) (Hooton, no date). In 1911 a Sewer Canal to Farmington Bay was completed and discharge of raw sewage into the Jordan River was discontinued, but transferred into the canal; by 1965 it was discharging $121 \times 10^6 \text{ L d}^{-1}$ into the south end of Farmington Bay. In 1922 the Northwest Oil Drain was joined to the Sewer Canal to receive refinery and other industrial wastes. Between 1959 and 1962 several additional sewage districts were formed, which now discharge secondary-treated wastewater into Farmington Bay and its tributaries (Figure 1). Total discharge of this secondary-treated sewage is $1.16 \times 10^9 \text{ L d}^{-1}$, representing a 22-fold increase in waste waters to Farmington Bay in 50 years.

Core collection

Cores were collected in the summers of 2009 and 2010 when the elevation of Gilbert Bay was near 1278.7 m (4195 feet). Nine sites were selected for lake sediment collection to span the different environments and salinities that exist in the lake (Figure 1). Short (<75 cm) cores were collected manually by inserting a Plexiglas tube into the sediments or by using a Kajak-Brinkman type gravity corer (Glew et al. 2001) at three sites in each of Farmington, Gilbert and Bear River bays to assess spatial variability in lake water eutrophication and metal deposition. Cores from Farmington Bay were located along a gradient of salinity, from the southern part of the bay where waters are fresher to the northern end proximate to the causeway. The most southern Farmington Bay site (Site 1) was in a 1 km wide and 15 cm deep channel, about 1 km from the discharge of the Sewer Canal.

All sites were collected from the central channel of Farmington Bay where the sedimentary sequence (stratigraphy) is assumed to experience the least disturbance by hydrological modification or turbulence. Cores from Gilbert Bay were taken from regions known to have a high rate of sediment deposition, as described by Johnson et al. (2008), including a southern-most core located 12 km from the outfall of the Kennecott mine. Cores from Bear River Bay were obtained from a transect between the GSL Mineral Bridge and the northeast section of Willard Spur. In addition to the master (dated) core at each site, two undated support cores were retrieved at each site and used for estimation of parameters that require a large sediment volume for accurate quantification of fossils (fossil invertebrates, brine shrimp cysts). Cores were stored vertically and most were sectioned into 5-mm increments in the field using a Glew extruder (Glew et al. 2001). In a few cases, support

cores were sectioned in the laboratory. All samples were kept at ~4 °C and in darkness using coolers as they were transported from the field to the laboratory. Depending on the parameter, subsequent sediment analyses were conducted on either every section or alternate sections.

Sediment chronology

Chronological analyses were conducted on ~15 samples per core at the University of Regina Environmental Quality Analysis Laboratory (Sites 1-3, 6) and the University of Waterloo (Sites 4, 5) using identical procedures and equipment. In all cases, sediment dating was based on ^{210}Pb activity measured by gamma spectrometry (Appleby et al. 1986; Schelske et al. 1994) using an Ortec High-Purity Germanium (HPGe) Coaxial Well Photon Detector System. After freeze-drying, samples were homogenized with a mortar and pestle and transferred into pre-weighed polyethylene tubes (15 x 80 mm) at the University of Regina. Individual tubes were filled to a height of 55 mm (equivalent to the depth of the HPGe well) and the sample weight recorded by re-weighing the sampling tubes. Samples were then sealed with a 5-mm layer of epoxy resin and set aside for at least 21 days to achieve equilibrium of the native ^{224}Ra and its decay products. Supported ^{210}Pb activity, expressed as ^{226}Ra activity, was based on average activities of ^{214}Pb (295.1 keV and 351.9 keV) and ^{214}Bi (609.3 keV). Unsupported ^{210}Pb activity was calculated by subtracting proxy estimates of supported ^{210}Pb from the total ^{210}Pb activities (46.5 keV). ^{137}Cs activity was measured at 661.7 keV to identify the period of maximum fallout from atmospheric nuclear weapons testing and validate ^{210}Pb dates. Sediment age-depth relations were calculated using the CRS (constant rate of supply) model (Appleby and Oldfield 1983), which is the model of choice when changes in sediment accumulation rate are suspected (Oldfield and Appleby 1984; Binford 1990). Counting errors were estimated by first-order approximation, assuming that gamma disintegrations are described by a Poisson distribution (Schelske et al. 1994). Bulk sediment accumulation rates ($\text{g cm}^{-2} \text{ yr}^{-1}$) were computed from output of the CRS model (Appleby and Oldfield 1983) and represent the mass of sediment deposited 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 depth-age relationships.

Invertebrate and macrofossil analyses were performed on the two undated support cores taken from each site. Depth-age relationships for each of the support cores were determined by correlation of visible litho-stratigraphic change in the dated master core to each support core. The major litho-stratigraphic changes in the dated cores were used as tie-points to correlate the depths in the support cores back to the equivalent depth and age in the master core. Dates in between tie-points were interpolated. Where litho-stratigraphic changes occurred at the same depth in each core (master and support cores, e.g., Site 3) no correction was made to the depth-age relationship of the support cores.

Invertebrates and macrofossil analysis

Sediment samples were processed for benthic invertebrates and macrofossils following a modified version of the chironomid processing method outlined in Walker (2001). Samples were weighed, deflocculated in hot 10% KOH and washed through a 90- μm mesh with distilled water. The entire 0.5-cm interval was processed in this manner for each sample analysed. Samples were then transferred to a Bogorov counting tray and examined for chironomid head-capsules and other macrofossils under a dissection microscope at 50 X

magnification. All chironomid head-capsules were transferred into a small pool of distilled water on a glass slide. The head capsules were allowed to dry for 48 hours. After drying, a drop of Euparal[®] was added before positioning a glass coverslip over the head-capsules. Slides were dried for 48 hours at 60°C in a drying oven to set the mountant. Chironomids were identified using a transmission light microscope at a magnification of 400 X with the aid of publications by Brooks et al. (2007) and Epler (2001). Other fossil remains of flora and fauna were identified and enumerated in the Bogorov counting tray during the picking of chironomid head-capsules. These other remains were identified using publications by Mathis and Simpson (1981), Cash-Clark and Bradley (1994) (Ephydra); Stearns and Krieger (2008), (Corixidae); Vandekerkhove et al. (2004) (cladoceran ehippia morphotypes); Wood et al. (1998) (statoblasts); de Winton et al. (2007) (charophyte oospore morphotypes).

Chironomid head-capsule counts were expressed as a percentage of the total head-capsule count. Counts for all other remains were expressed as concentrations per gram of dry sediment. The sediment dry weight was determined by drying sediment sub-samples in a drying oven and correcting for the weight of salts present in the sediment pore water. Historical changes in past chironomid assemblages (stratigraphy) were plotted with respect to ²¹⁰Pb-estimated age using the computer software package C2 (Juggins 2003).

The tolerance for each chironomid taxon for “organic waste” (a value between 0 and 10) was estimated following methods of Bode et al. (2002) and Hauer and Lamberti (1996) and presented in Appendix 1. These values were then used to calculate a modified Hilsenhoff Biotic Index (FBI) for each fossil sample (Hilsenhoff 1987) according to:

$$BI = \frac{\sum x_i t_i}{n}$$

where

x_i = number of individuals within a species

t_i = tolerance value of a species

n = total number of organisms in the sample

Taxonomic richness (the number of taxa) and diversity (Shannon Weaver index) were also calculated for each fossil sample. Historical changes in chironomid assemblages were also used to estimate past lake-water salinity (g TDS L⁻¹) in Farmington Bay based on known preference (optima) and tolerances of these taxa in other North American saline lakes and the weighted-averaging approach of Walker et al. (1995).

Brine shrimp cyst abundance and hatching

Almost the entire sediment samples from one of the support cores were used for these analyses. Samples were frozen at -20°C in plastic bags for three months to allow viable cysts to break diapause (Lavens and Sorgeloos 1996). For Site 1, 54 of the 74 samples, and for Gilbert Bay sites 3 and 4, all samples, were selected for enumeration and hatching of cysts. The sediments were dried in tared weighing pans until a constant weight was achieved (3-4 days) at 30-33°C, a temperature that does not harm *Artemia franciscana* embryos (Triantaphyllidis et al. 1994). To extract the cysts from the sediment, samples were placed in 500-ml jars and 200 ml of 240 g L⁻¹ NaCl brine was added. They were incubated at 12°C and stirred daily for one week to facilitate the release of cysts from the sediments. The cysts were then decanted from the jars and counted at 10X magnification with a dissecting microscope. Intact cysts and cyst cases were enumerated separately. The density of intact cysts and

hatched cyst cases were highly correlated ($r^2 = 0.95$ at Site 3, and $r^2 = 0.92$ at Site 4). Cyst densities are expressed per gram of dry sediment after salt corrected. Cysts were stored at 4°C in saturated NaCl brine for seven months until hatching analyzes could be carried out.

To hatch cysts, they were placed in 150 cm² rectangular cell culture flasks (Corning[®] Model #430823) with 150 mL of a 35 g L⁻¹ solution of 60% NaCl and 40% Instant Ocean[®] sea salt. They were incubated for four days at 25°C with constant illumination of 150 $\mu\text{E m}^{-2} \text{sec}^{-1}$. Hatched *Artemia* nauplii were counted on days 2-4 by attracting them to one side of the flask with intense illumination from a microscope lamp. When numbers of nauplii were >100 counts became approximate. The maximum number of nauplii observed during the 3 days was used to calculate the proportion of intact cysts that hatched. We did not attempt to hatch cysts from the Farmington Bay core as they were too few in number.

Results and Discussion

Chronology

Detailed analysis of sedimentary radioisotopes and chronological estimates are provided in Leavitt et al. (2012). Briefly, only one core from Farmington Bay (site 1) and two from Gilbert Bay (sites 3, 4) exhibited declining ²¹⁰Pb radioactivity suitable for establishment of a reliable estimate of past ages or rates of sediment deposition. Specifically, ²¹⁰Pb activity declined to background levels from peaks in the upper 0-2 cm at sites 1 and 4, whereas an intermediate peak was noted at ~6 cm depth at site 3, representing a change in the rate of sediment accumulation. Similarly, specific activities of ¹³⁷Cs reached expected maxima during the early 1960s at sites 1 and 4, whereas maximal ¹³⁷Cs activity precedes expected dates by ~20 yr at site 3. In general, there was no consistent evidence of severe post-deposition mixing of the sediment column at any sites. As a result, application of dating models suggested that cores from all 3 sites (1, 3, and 4) spanned ~200 years, that sediment age generally increased smoothly with burial depth, and that accurate estimates of historical changes in invertebrate abundance could be derived from estimates of either fossil concentration or accumulation rates (Leavitt et al. 2012).

Depth-age relationships for each of the support cores were determined based on correlation of visible litho-stratigraphic changes from the dated master core to each support core. We are confident of the accuracy of the depth-age models for the support cores due to the small differences in the depths of all major litho-stratigraphic units in each core (< 1.5 cm). There was no difference between the depths of the major litho-stratigraphic changes in the master and support cores at site 3.

Farmington Bay fossil invertebrates

A total of twenty chironomid taxa (including species and morphotypes) were recovered from site 1 in Farmington Bay. The relative abundance (%) of the more common types (> 2% in at least one sample) is presented in Figure 3. A list of all chironomid taxa, their tolerance for organic pollution (Biotic Index), salinity optimum, maximum and mean abundance is provided in Appendix 1. Prior to 1860, fossil taxonomic richness tended to be low (~9 taxa), but typical among saline lakes worldwide (Hammer 1986). The early chironomid assemblage was composed mainly of *Paratanytartus* and *Crictopus* type. In

general, density of *Artemia* cysts was elevated prior to Mormon settlement in the mid-1800s, although values were low; Ostracoda were present at moderate densities, whereas cladocerans (*Daphnia*, *Bosmina*) were either absent or greatly diminished (Figure 4). *Artemia* cysts were absent or in low concentrations in Farmington Bay between 1850 to 1960, even though salinities in Gilbert Bay were suitable for *Artemia* over some portions of this time period and the automobile causeway did not separate Farmington Bay from the main lake until 1969 (Table 1). The reason for the low abundance at this time, and subsequent increase is unclear.

Interpretation of invertebrate species composition suggests that Farmington Bay was mesotrophic prior to the mid-1800s. For example, members of the genus *Paratanytarsus* are considered representative of moderate nutrient loading and algal productivity (Figure 3). Similarly, the Biotic Index, based on the chironomid fauna, was moderate (6-7), indicating mesotrophic conditions (Figure 4). *Paratanytarsus* is commonly associated with macrophytes (Buskens 1987; Brodersen et al. 2001), consistent with the presence of charophyte (stonewort) oospores in sediments deposited prior to 1880 (Figure 4). The charophyte remains that were recovered from Farmington Bay were predominantly from the genus *Chara*, with some oospores from *Nitella* also present. Finally, the absence of cladoceran remains and freshwater bryozoans statoblasts prior to the mid 1800s, combined with elevated densities of *Artemia* suggests a site of high salinity, elevated herbivory by zooplankton, and low algal abundance (Leavitt et al. 2012), although interpretation of overall salinity in the basin is difficult because of the proximity of site 1 to the inflow of the Jordan River.

Analyses of fossil chironomid assemblages identified substantial changes in benthic invertebrate community composition after ~1860 (Figure 3). At that time, head capsules from chironomids typical of more productive lake conditions, including *Chironomus anthracinus* and, after 1930, *Tanytarsus clavatus*, increased. Concurrently, the concentrations of fossil remains from brine shrimp decreased (Figure 4) indicating a period of decreased salinity. These faunal changes occurred concomitant with the construction of the first sewer line into the Jordan River in 1889 (Table 1), and appear to track an increase in wastewater influx to Farmington Bay. Consistent with an interpretation of elevated production, absolute densities of charophyte, chironomid, and ostracod fossils also increased to a peak in the early 20th century (Figure 4). Similarly, the Chironomid Biotic Index increases steadily and reached ~ 8.5 ca. 1911. This value is characteristic of 'severe organic pollution' (Table 3). *Chara*, an aquatic macrophyte, may have buffered the lake from initial rises in nutrient influx (Kufel and Kufel 2002), although *Chara* does not survive in highly eutrophic systems (Rodrigo et al. 2010) either due to detrimental effects of elevated phosphorus concentrations (Martin et al. 2002) or shading from blooms of buoyant phytoplankton (Blindow 1992).

In general, chironomid densities increased to a maximum at ~1935 before declining sharply to baseline levels for ~12 years (Figure 3). Absolute densities of ostracods and *Chara* remains reached maxima at about the same time, but then decline (Figure 4). This interval corresponds to a prolonged period of substantially reduced lake level (Figure 2). Consistent with this interpretation, the dominance of *Cricotopus intersectus* type at this time indicates increased lake water salinity, as this taxa has a relatively high tolerance to dissolved salts (up to ~22 g/L) (Walker et al. 1995). The persistence of obligately aquatic invertebrates and benthic charophytes demonstrate that site 1 was never completely desiccated during the GSL low stand, but it was likely cut off from the Jordan River inflows. Although the chironomid biotic index indicates an improvement in water quality during this time (Figure 4), we should be cautious interpreting these values because inferred salinity is relatively high

meaning the Biotic Index, which is based on lakes with lower salinities, may be less accurate. Interestingly, the concentration of algal pigments in the Farmington Bay core declines at the same time, however, this could also be due to increased salinity (Leavitt et al., 2012; Figure 4).

Increased lake levels, followed by hydrologic restriction of Farmington Bay after 1969 and a switch to secondary sewage treatment appears to have re-established elevated populations of benthic and planktonic invertebrates, but favoured hyper-eutrophic taxa over the natural assemblages characteristic of the 1800s (Figures 3, 4). For example, substantial increases in deposition of fossils from pelagic cladoceran, such as Daphnids and *Bosmina*, after ca. 1960 coincided with the return of chironomid densities to historical maxima, complete loss of the original benthic fauna, and persistently elevated biotic indices of organic pollution (>9.5). At this time, fossil chironomid assemblages were composed largely (>50%) of *Tanypus clavatus* type. To the limits of our chronological accuracy, these changes were coeval with hydrologic restriction of Farmington Bay by the northern causeway to Antelope Island, an event which coincided with a 10-fold increase in algal production and deposition in Farmington Bay (Leavitt et al. 2012). After the causeway construction densities of brine shrimp cysts also returned to levels characteristic of the early 1800s (Fig. 4), despite inferences of reduced lake-water salinity in this bay (e.g., reductions in *Cricotopus intersectus* type, increased *Chironomus anthracinus* type and *Tanypus clavatus*) and the observation that this taxon does not tend to co-occur with freshwater Daphnids (Wissel et al. 2011).

Overall, analysis of modern invertebrate assemblages reveals that Farmington Bay is characterized by extremely high trophic status, with biotic indices now approaching hypereutrophic conditions (HBI ~10.0; Figure 4). This biotic assessment is consistent with recent measurements of eutrophication in the bay (Wurtsbaugh and Marcarelli 2006, Wurtsbaugh et al. 2006, Wurtsbaugh 2011). Although *Artemia* have occasionally been observed in Farmington Bay, native chironomid species have been eliminated, and the waters are characterized by abundant freshwater cladocerans. The altered faunal assemblage, combined with elevated algal abundance (Figure 4) (Leavitt et al. 2012) demonstrates that the entire food web of Farmington Bay has been fundamentally altered by increased influx of nutrients and altered hydrologic exchange associated with human activities.

Gilbert Bay fossil invertebrates

Fossil invertebrate assemblages in sediments from both locations in Gilbert Bay were composed mainly of hyper-saline brine flies (*Ephydra*) and brine shrimp (*Artemia*). In both cores all remains from *Ephydra* and *Artemia* increased from baseline densities during the 20th century, although timing differed somewhat between the two sites (Figure 5). Specifically concentrations of fossils from both brine flies and shrimp increased steadily after ~1940 at site 3 in north-central Gilbert Bay, but only increased after ~1960 in the more southern site 4 location (Figure 5). *Artemia* cysts, however, only reached significant numbers (> 50 cysts g⁻¹) after 1960 (Figure 6). Patterns of historical change in density were similar for all lifestages of *Ephydra*, although densities of larval parapods were typically 10-fold greater than those of adult or pupal stages. The *Ephydra* population increase inferred from the fossil record spanning the 20th century is consistent with data published by Winget et al. (1972) indicating rapidly increasing fly populations in the late 1960s and early 1970s. Interestingly, the low concentrations of *Ephydra* remains from site 4 prior to ~1900 is inconsistent with reports of Aldrich (1912) who notes a large density of brine fly adults and pupae in 1890, and could

suggest that this taxon may not be accurately represented or preserved poorly at that site prior to 1900. Accurate representation could be affected by the presence of the deep brine layer that formed in Gilbert Bay after the Railway Causeway construction. This layer may influence the number of fossils because particles of intermediate density can concentrate at the boundary layer (chemocline) and not sink further (Jones and Wurtsbaugh, unpublished data).

Comparison of historical changes in brine shrimp and flies suggest that population densities of taxa native to GSL were responsive to changes in both primary production and lake-water salinity. For example, concentrations of remains from *Ephedra* and *Artemia* increased steadily at site 3 after 1900, in a pattern similar to past algal abundance at that location (Figure 5) (Leavitt et al. 2012). Density of fossil invertebrate and algal remains seemed to plateau or even decline slightly after ca. 1960, consistent with a correlation between primary and secondary production. In contrast, densities of fossils from hypersaline invertebrate taxa increased at southern site 4 only after separation of Gilbert and Gunnison bays by an impermeable railway causeway in 1959, whereas estimated algal abundance appeared to decline at that location after the mid-20th century (Figure 5). Thus, although earlier reports on GSL have suggested that *Ephedra* populations exhibit pronounced inverse responses to long-term fluctuations in salinity (Collins 1980), our analysis suggests a more complex interaction between nutrient concentrations and salinity.

Documented declines in the surface water salinity of Gilbert Bay following replacement of the railroad causeway in 1959 appear to correspond well to coeval increases in deposition of fossils from freshwater corixids (Figure 5). Specifically, elytra from Corixidae were recovered at significant densities only after ca. 1960, and have apparently remained elevated at both core locations. These invertebrates colonize Great Salt Lake during wet intervals and salinities $< 80 \text{ g L}^{-1}$ and have been proposed as important predators of native hypersaline species (Wurtsbaugh and Berry 1990; Wurtsbaugh 1992). Instead we see that increased densities of corixids correspond to historical maxima of both *Artemia* and brine flies, although we note that our cores have low temporal resolution, especially at site 4, therefore ecological interactions should be interpreted with caution.

Owing to the economic importance of the brine shrimp industry, we also tested the viability of *Artemia* cysts to further our understanding of the brine shrimp ecology and evolution. Cyst viability was much higher in more recent sediments (40-80% hatch rate) than in deeper sediments, but cysts as old as 250 years hatched in our experiment (Figure 6c). In general cysts were more viable from Site 3 than from site 4, but the same general decline in viability in older sediments was observed in both cores. Cysts have been found in 600,000 year old sediments in the Great Salt Lake (Spencer et al. 1984), as well as in cores from other Great Basin Lakes (Bradbury et al. 1989; Clegg and Jackson 2002). Clegg and Jackson (2002) found cysts with good ultrastructural configuration, but could not hatch them because they had been dried and exposed to oxygen, which is toxic to them prior to hatching. Others have hatched copepod eggs that were estimated to be >300 years old (Hairston et al. 1995). The dormancy of cysts in the sediments likely represents a genetic store that may allow *Artemia* to respond to different conditions (e.g. salinity, eutrophication, metal resistance) when internal waves redistribute the cysts so that they can hatch (Hairston and De Meester 2008; Derry et al. 2010).

Synthesis and Conclusions

Analysis of sedimentary fossils revealed fundamental changes in secondary production and composition of invertebrate communities of GSL during the past 200 years (Table 2). In Farmington Bay, which was initially mesotrophic, there has been a loss of many of the native benthic invertebrates as a result of increasing nutrient pollution and altered hydrologic exchanges. Specifically, benthic communities in Farmington Bay were initially composed of charophytes and chironomids characteristic of moderately productive waters, which is consistent with the interpretation from fossil algal pigments and biochemical markers at this site (Leavitt et al. 2012). Following influxes of waste waters in the late 1800s, native taxa, such as *Paratanytarsus* type, declined precipitously, to be replaced first by *Chironomus anthracinus* and then *Tanytus clavatus*, both indicators of highly eutrophic waters, particularly after hydrologic restriction of Farmington Bay in the 1960s. Thereafter, previously absent Daphnids and *Bosmina* became important members of the invertebrate assemblage, although *Artemia* were also present during the past 50 years. Given these patterns, it appears that a combination of continued nutrient influx and restricted circulation of water with other basins, has increased primary production and restructured the invertebrate assemblage of Farmington Bay. Construction of the railway causeway and the subsequent decreasing salinity in Gilbert Bay, combined with increasing nutrient levels appear to have increased the abundances of *Artemia* and brine flies, but have not influenced the community assemblage.

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Figure Captions:

Figure 1: Map of Great Salt Lake Utah, including core collection locations. This report provides detailed analysis of master cores collected at Farmington Bay site 1 (09GSL01), Gilbert Bay Site 3 (09GSL03), and Gilbert Bay site 4 (09GSL04).

Figure 2: Record of lake level elevation and salinity from Great Salt Lake. Salinities prior to 1964 were calculated based on the known salt content of the lake and the lake volume (personal communication, Craig Miller, Utah Department of Water Resources). Subsequently, salinities were based on measured concentrations by the U.S. Geological Survey.

Figure 3: Chironomid taxa from Farmington Bay Site 1 presented as a percentage of the total head-capsule count from each down core sample. The lowermost sample only contained a single head capsule, thus any inferences based on this sample should be viewed with caution. Chironomid taxa are color coded based on their tolerance for organic pollution; red being very tolerant and purple being very sensitive to organic pollution or eutrophication. The chironomid record is divided into four periods with similar assemblages (zones) based on CONISS in PSIMPOLL (Bennett, 2005). The biggest change (greatest dissimilarity) occurs just before 1955AD. Chironomid biotic index values (Appendix 1) were used to calculate the biotic index values in Figure 4. Major events in the Great Salt Lake catchment are presented on the right of this figure for the sake of comparison.

Figure 4: Concentration of major plant and animal macrofossils, chironomid-based environmental inferences (water quality and salinity), sediment phosphorus and algal pigment concentrations for Site 1 in Farmington Bay. Major events in the Great Salt Lake catchment are presented on the right of this figure for the sake of comparison.

Figure 5: Invertebrate stratigraphy, sediment P concentrations and salinity observations from Great Salt Lake sites 3 and 4 in Gilbert Bay. Densities of brine shrimp cysts include both intact and hatched cases.

Figure 6: A. Salinity in the Great Salt Lake over the 1850-2003 recorded record. The arrow on the left axis shows the date of construction of the Railway Causeway across the lake that has caused Gilbert Bay to become fresher. The vertical dotted line shows the approximate salinity where *Artemia* reproduction declines (Clegg and Conte 1980; J. Clegg, personal communication). B. Intact *Artemia* cysts densities from sites 3 and 4 in Gilbert Bay and site 1 in Farmington Bay over 170-370 year time spans. C. Percentage of cysts from the two Gilbert Bay cores that hatched. Note log scale which exaggerates the lower cyst densities in the earlier part of the sediment record.

Tables:

Table 1. Major hydrologic, industrial, and wastewater events in the Great Salt Lake basin, 1847-2000.

Year	Event
1847	Mormon pioneers settle Salt Lake Valley
1873	Lake reaches high level (1283.5 m), but salinity only decreases to ~136 g/L
1885	Surplus Canal constructed that diverts flood flows of Jordan River directly to Gilbert Bay, thus directing riverine nutrients away from Farmington Bay
1889	First sewer line in Salt Lake City discharges to the Jordan River
1911	Salt Lake City Sewage Canal completed to Farmington Bay. Discharges reduced into the Jordan River. Oil drain connected 1922.
1930	Farmington Bay Waterfowl Management Area constructed, which captures Jordan River flows. Diking helps capture sediments and nutrients.
1952	South causeway to Antelope Island constructed, partly to prevent raw sewage from reaching swimming beaches at south end of Gilbert Bay.
1959	Railroad Causeway completed that separates Gilbert and Gunnison Bay, which causes salinity to generally decrease in Gilbert Bay, although a deep brine layer begins to form
1959– 1962	Sewer districts formed in Davis Co. to discharge wastewater into Farmington Bay.
1963	Lake reaches lowest recorded level (1277.8 m).
1965	Secondary treatment facility completed in Salt Lake City.
1969	Automobile causeway to Antelope Island completed, partially isolating Farmington Bay. Maximum elevation 1282.2 m
1984	Automobile causeway to Antelope Island flooded by high water. Salinities increase in Farmington Bay. Flooded until mid-1989.
1985	Gilbert Bay reaches 1282.9 m, salinity declines to 58 g/L
1992	Automobile causeway rebuilt

Table 2. Changes in Farmington Bay (Jordan River delta) and Gilbert Bay during the last 170 years inferred from remains of invertebrates and plant fossils.

Period	Inferred environment	Salinity	Descriptors
Farmington Bay (Jordan River Delta)			
1840- 1880	Mesotrophic	Max. 5 g/L	Moderate nutrient loading. Low planktonic productivity. Charophyte (stonewort) meadows present in Farmington Bay.
1880-1950	Increasing to eutrophic	Moderate (<16 g/L)	Low planktonic algal production. Increase in macrophyte standing crop and chironomid populations after the completion of the outlet sewage canal into Farmington Bay in 1911. Moderate salinities (< 16 g/L) in Farmington Bay concurrent with low lake levels and high salinity in the rest of Great Salt lake between 1930 and 1960AD.
1950-1965	Increasing to hypereutrophic	~ 5 mg/L	Loss of chironomid diversity. Construction South Antelope Island Causeway (1952) and then the Antelope Island causeway (1969) reduces exchange with Gilbert Bay. Chironomid assemblages indicate severe nutrient loading. High abundances of cladocera remains and bryozoans statoblasts indicate a large concentration of suspended organic particles (planktonic algae).
1965-Present	Hypereutrophic	Variable with rise and fall of lake	Low chironomid diversity and assemblages indicate severe nutrient loading. High abundances of cladocera remains and bryozoans statoblasts. Predacious corixidae appear. Increase in <i>Artemia</i> , perhaps associated with mid-80s flooding of delta with Gilbert Bay water.
Gilbert Bay			
1820-1950	Oligo-mesotrophic	Generally high (150-300 g/L)	Relatively low abundances of <i>Artemia</i> and <i>Ephydra</i> (brine flies)
1950-Present	Mesotrophic	Variable, 60-250 g/L	Higher abundances of <i>Artemia</i> & <i>Ephydra</i> after Railway Causeway construction in 1959 lowers salinities. Perhaps increased food for <i>Artemia</i> . Predatory Corixidae appear in low densities.

Figures:

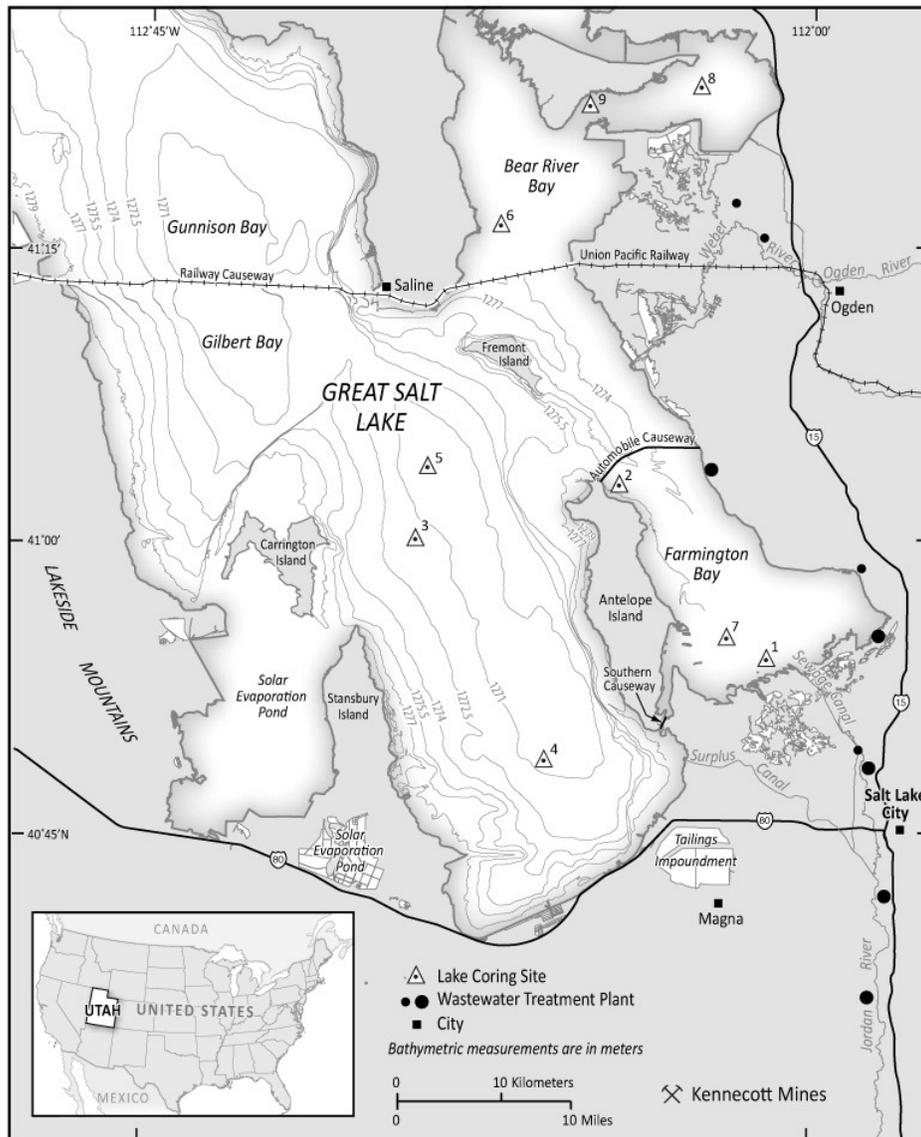


Figure 1

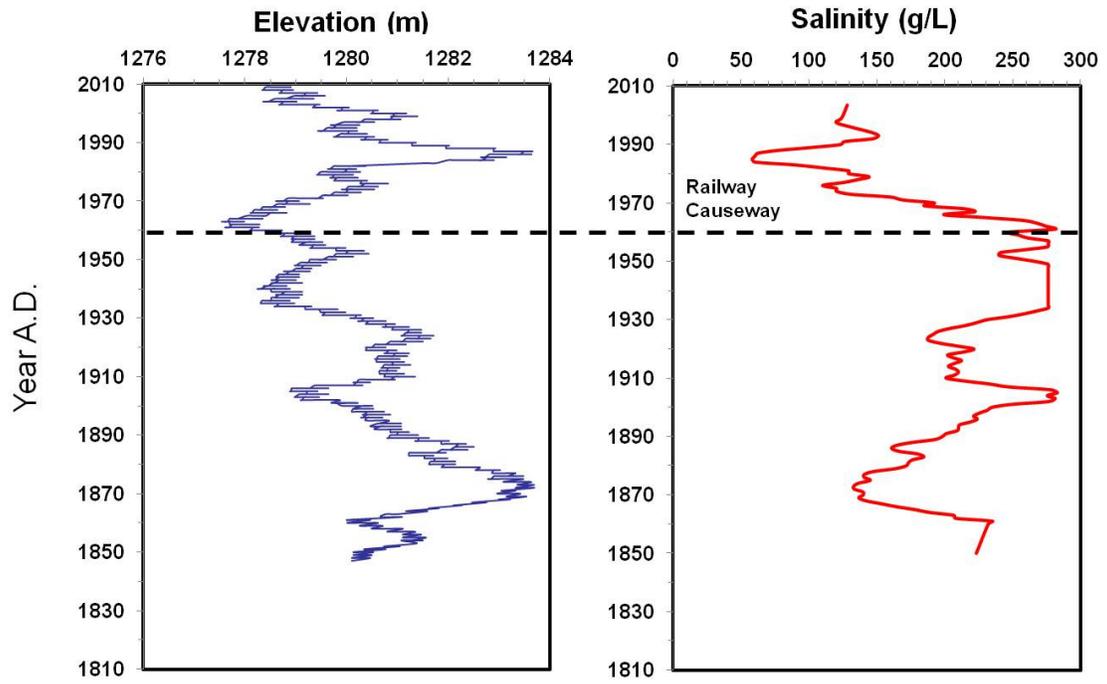


Figure 2

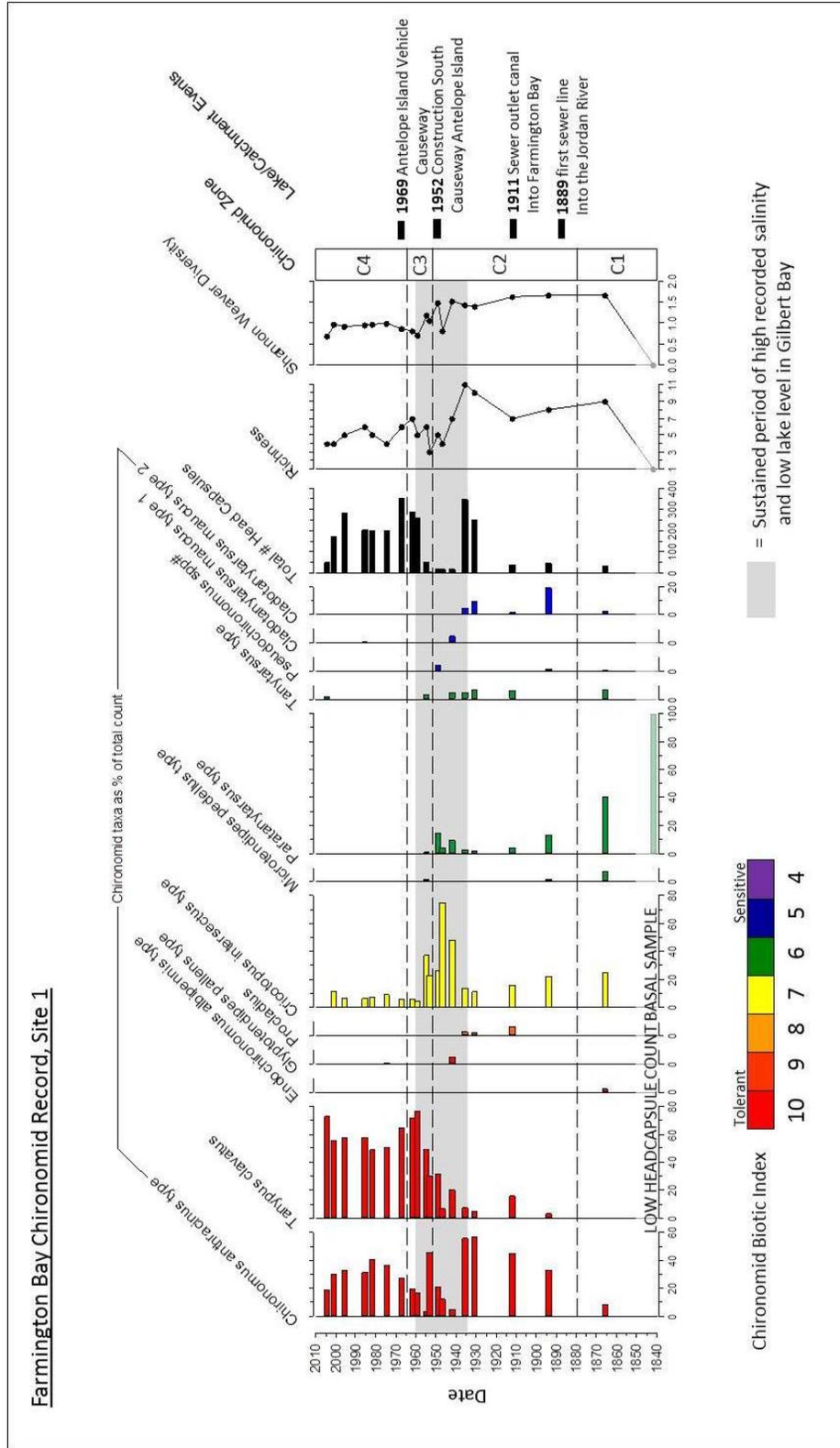


Figure 3

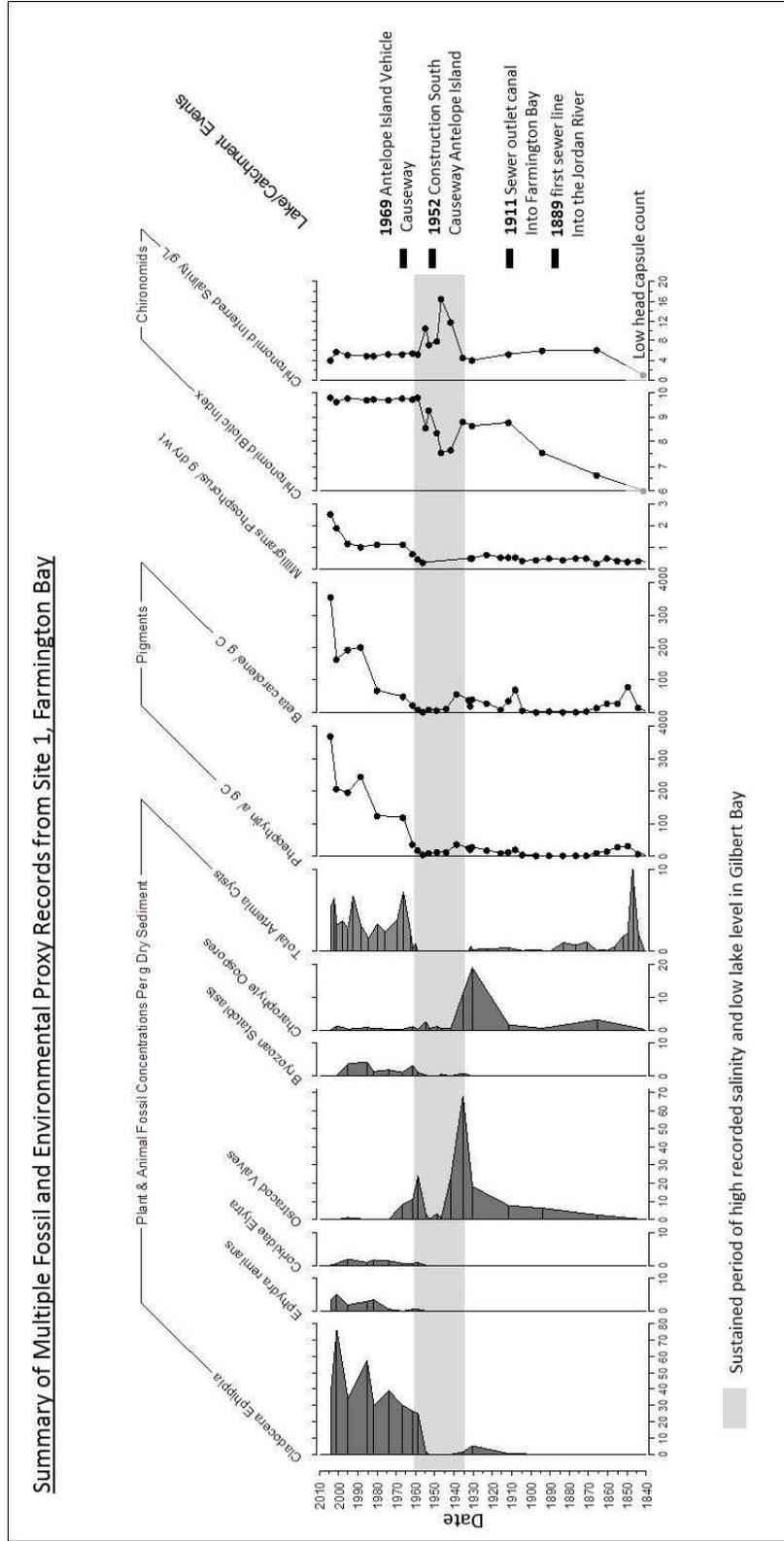


Figure 4

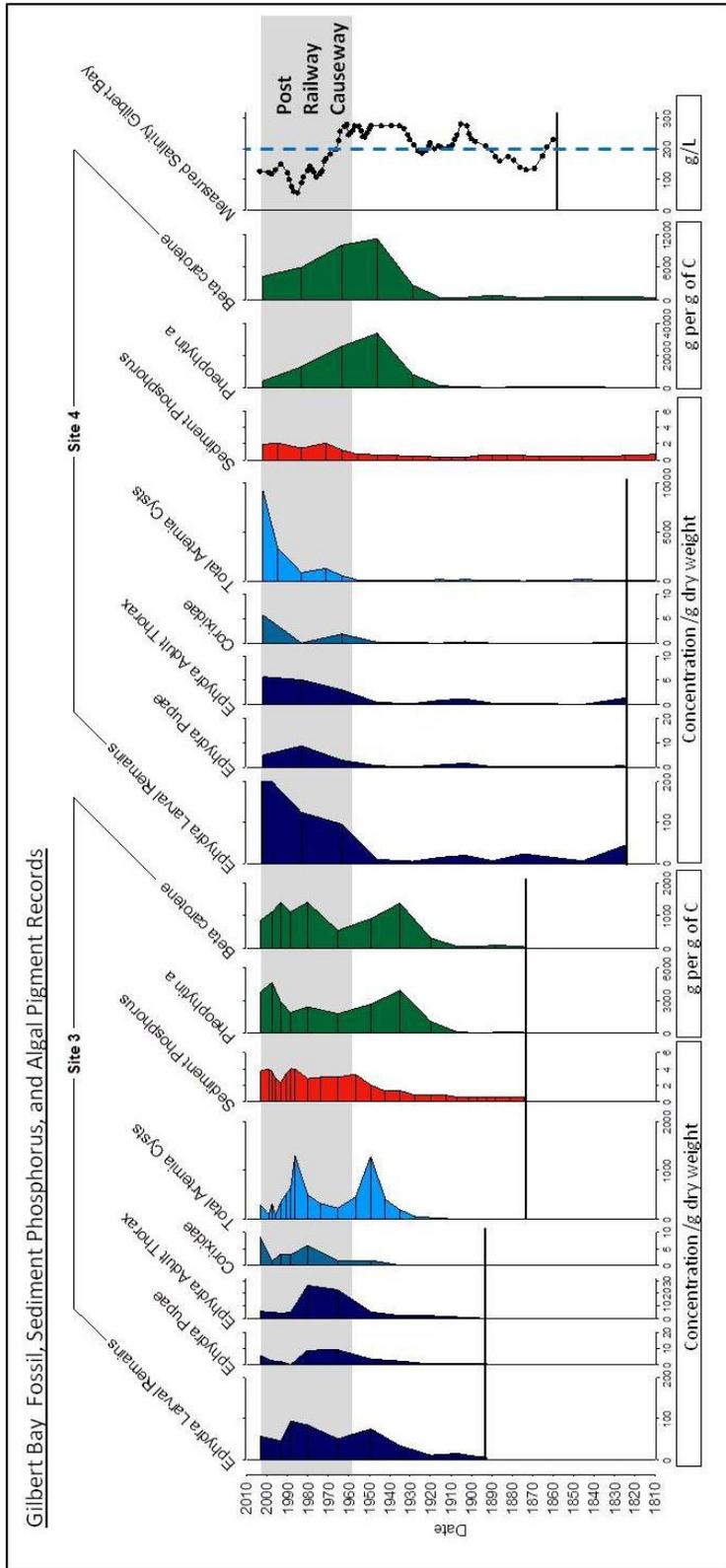


Figure 5

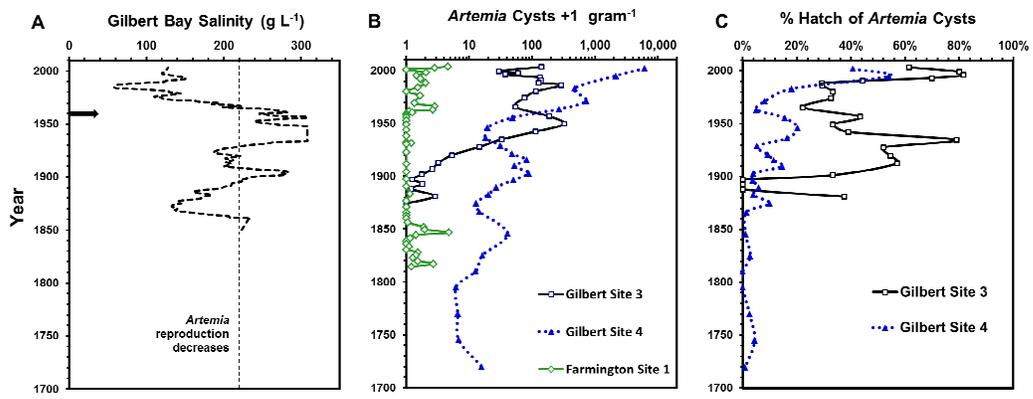


Figure 6

Appendix 1

A full list of all chironomid taxa recovered from Farmington Bay, Site 1. The Tolerance (Biotic Index), salinity optimum, maximum abundance and mean abundance are also provided for each taxon. Ceratopogonidae remains (Diptera) were also recovered from the sediments and data for this group are included with the Chironomidae.

Taxon	Tolerance (Biotic Index)	Salinity Optimum mg/L	Maximum Abundance %	Mean Abundance %
<i>Corynoneura arctica</i> type	4	634	1.8	0.1
<i>Corynoneura edwardsi</i> type	4	634	2.5	0.1
<i>Eukiefferiella devonica</i> type	4	*	0.5	0.02
<i>Cladotanytarsus maucus</i> type 1	5	1064	19.8	0.3
<i>Cladotanytarsus maucus</i> type 2	5	1064	7.5	2
<i>Pseudochironomus</i> spp.	5	186	4.8	0.4
<i>Microtendipes pedellus</i> type	6	104	7.5	0.6
<i>Tanytarsus</i> type	6	1064	7.5	1.9
<i>Paratanytarsus</i> type	6	1064	100	9.8
Ceratopogonidae	6	*	2.4	0.1
<i>Cricotopus intersectus</i> type	7	21135	75	18.2
<i>Microspectra insignilobus</i> type	7	*	0.4	0.02
<i>Procladius</i>	9	1211	7.1	0.7
<i>Chironomus anthracinus</i> type	10	1738	57.2	27.6
<i>Chironomus plumosus</i> type	10	1738	0.5	0.04
<i>Tanypus clavatus</i>	10	5070	76	36.6
<i>Endochironomus albipennis</i> type	10	178	2.5	0.2
<i>Glyptotendipes pallens</i> type	10	1072	5.2	0.5
Tanypod type 1	*	*	0.6	0.05
<i>Gillotia</i> spp.	*	*	1.9	0.1
<i>Neozavrelia</i> spp.	*	*	1.7	0.09

* indicates that no data is available

