Effects of wastewater influx and hydrologic modification on algal production in the Great Salt Lake of Utah, USA[1]

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Executive Summary

Analysis of sedimentary pigments, geochemistry, and algal microfossils (cyanobacteria, chlorophtyes, diatoms) revealed a consistent pattern of eutrophication in Farmington and Gilbert Bays, the two southern basins of Great Salt Lake (GSL), Utah. Remains from bloom-forming cyanobacteria (*Anabaena*, *Gloeotrichia*) were present in 200-year old lake sediments, demonstrating that GSL was naturally productive. However, biogeochemical reconstruction of algal abundance at three sites with reliable chronologies demonstrated that water quality degraded during the late 1800s, concomitant with the 1889 construction of septic systems to introduce wastewater directly into GSL. Overall, increases in algal abundance during the first 50 yr of eutrophication were much more pronounced at Gilbert Bay (Sites 3 and 4) than in Farmington Bay (Site 1), possibly because of enhanced nutrient influx via the Surplus Canal (constructed 1885) and more pronounced hydrologic exchange among southern basins early in the 20th century. Thereafter, the relative degree of eutrophication of Farmington and Gilbert Bays appear to have been altered by a combination of continued nutrient influx, lake-level decline, causeway construction, and associated changes in water circulation within GSL. Specifically, while algal abundance increased in Farmington Bay during the early 20th century, the most rapid eutrophication at this site occurred after ca. 1960, coincident with diminished lake levels and sequential hydrologic closure of Farmington Bay by the southern (1952) and northern (1969) causeways to Antelope Island. Similarly, algal abundance appears to have declined at the southern-most Gilbert Bay site just as that of Farmington Bay increased. It is of note that establishment of the secondary wastewater treatment facilities in Salt Lake City in 1965 has not notably improved water quality or reduced algal biomass in Farmington Bay. Instead, causeway construction appears to have constrained the most severe eutrophication to Farmington Bay and may have reduced the degree of eutrophication at some Gilbert Bay locations. Although changes in water influx and circulation will continue to modify algal production in GSL, there appears little opportunity for substantial water quality improvement until nutrient influxes are more effectively controlled.

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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 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 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 support avian production. This issue is of particular interest with regard to Farmington and Bear River bays of Great Salt Lake (GSL), Utah, both of which host large populations of shorebirds, waterfowl and other avian taxa which rely on high production of invertebrates (Paul and Manning 2002).

Eutrophication processes in GSL may be particularly complex as the lake is divided by several causeways which restrict natural hydrologic circulation (Fig. 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. Farmington Bay can reach salinities of 9% (by mass) which is 2½ times saltier than the ocean (3.5%), while those of Bear River Bay can be even higher. Currently, Gilbert Bay has a salinity of 14%, although during the floods of 1984-85, salinities decreased to 5%. 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, water in that basin is nearly 30% salt, by mass. 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”.

Great Salt Lake is experiencing symptoms of severe cultural eutrophication in some basins (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 (NAWQA; 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 very high in Gilbert and Bear River bays (Wurtsbaugh et al. 2008), but the degree to which this is due to nutrient inputs, 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 and brine flies. Similarly, the salt-saturated waters of Gunnison Bay support only a few types of algae, bacteria and Archaea (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 may reduce oxygen content of sediments and overlying water, resulting in inhospitable conditions for aquatic life.

Intermittent monitoring suggests that surface waters in Farmington Bay have been eutrophic for decades (Coburn and Eckhoff 1972, Sorensen et al. 1988) with modern estimates of Trophic State Index among the highest of any measured water body in Utah – frequently 5 to 10 times greater than the 'hypereutrophic' designation (Wurtsbaugh and Marcarelli 2006). Similarly, concentrations of biological toxins from cyanobacteria have been observed at 5-10 times the maximum level recommended to protect human health by the World Health Organization (2003), and 20 times higher than concentrations shown to cause bird mortalities (Lopez-Rodas 2008). Eutrophication in the other bays of the lake has not been quantified frequently, and little is known of the historical changes in algal production in either Gilbert or Bear River bays. However, because terminal salt lakes also concentrate nutrients naturally during the process of water evaporation (Javor 1989), it is not known whether these bays are more eutrophic than they would be under natural circumstances. Similarly, Farmington Bay, like many estuaries, may also have been naturally productive and supported cyanobacterial blooms prior to settlement of the Wasatch Front.

The objectives of this study were to use a diverse suite of biological and geochemical metrics to quantify the timing, extent and trajectory of historical changes in algal abundance in the main basins of GSL. Of the ten sites initially explored, dating was attempted at six sites, and only one location in Farmington Bay and two sites in Gilbert Bay allowed quantitative estimation of sediment age and deposition, the prerequisite conditions for reliable evaluation of historical eutrophication. As a result, this report presents detailed sedimentary records of these historical changes in algal abundance (fossil pigments), cyanobacterial composition (akinetes and other morphological fossils), diatom community composition (siliceous remains), as well as diverse geochemical estimates of overall lake production (carbon[C], nitrogen [N], phosphorus [P] contents; stable N and C isotopes). Together, these analyses demonstrate that Farmington and Gilbert bays are experiencing eutrophication of their surface waters, most likely due to on-going influx of incompletely-treated wastewaters. In addition, timing of severe algal outbreaks appears to differ among embayments due to changes in water circulation associated with lake-level change and construction of causeways. In contrast,

wastewater management strategies appear to have had limited beneficial effects in controlling algal growth, probably because the urban plants lack modern Biological Nutrient Removal (BNR) technologies to remove growth-limiting nutrients (N, P).

Methods

Core collection

Short (<75 cm) cores were collected manually by inserting a Plexiglas tube into the sediments or by using a Kajak-Brinkman gravity corer (Glew et al. 2001) at several sites in Farmington, Gilbert, and Bear River bays to assess spatial variability in lake water eutrophication (Fig. 1). 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. 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 near 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 elevated sediment mass for accurate quantification of fossils (brine shrimp cysts, fossil invertebrates). Ages of support cores are pending and will be estimated using analysis of stable isotopes from all sediment columns (see below), as well as visible litho-stratigraphic changes in physical properties of the sediments (color, texture, inclusions, etc.) obtained from field photography. 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 strata.

Sediment chronology

Chronological analyses were conducted on ~15 samples per core at University of Regina Environmental Quality Analysis Laboratory (Sites 1-3, 6) and 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 it 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 (g cm^{-2} 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.

Stable isotopes and phosphorus

Stable isotopic compositions of the sediments were analyzed from freeze-dried samples using a Thermoquest (Finnigan MAT) Delta^{Plus} XL stable isotope ratio mass spectrometer equipped with a continuous flow (ConFlo II) and a Carlo Erba NC-2500 elemental analyzer, following the standard methods of Savage et al. (2004). Sediments were analyzed directly without treatment with HCl (1N HCl, ~36 h) to remove inorganic carbon. Samples of 2-10 mg dry mass were packed into tin capsules and introduced into the NC-2500 elemental analyzer. N and C components of sediments were completely oxidized at 1000°C in a furnace in order to convert organic constituents into simple nitrogen-based gases and CO_2. Elemental ratios were estimated as mass N or C relative to dry mass of sediment combusted. Stable isotope ratios (δ values) were calculated relative to the international standards including Pee Dee Belemnite (PDB) for C isotopes (δ^{13}C) and atmospheric nitrogen gas for N isotopes (δ^{15}N). Stable isotopic composition was expressed as δ notation where $\delta = (R_{sample}/ R_{standard} -1) \times 1000$, R_{sample} represents $^{13}C/^{12}C$ or $^{15}N/^{14}N$ in the sample, and the $R_{standard}$ is the corresponding isotope ratio from a standard. The precision of repeated measurements of a laboratory reference (inter-calibrated freshwater lake sediment) was 0.3‰ or better.

Phosphorus (P) content was determined on sediment subsamples of 5 g wet mass at the NAPT-certified Colorado State University Soil, Water and Plant Testing Laboratory. Briefly, water content was determined gravimetrically, then air-dried sediment samples were digested completely using a combination of concentrated nitric (HNO_3) and perchloric ($HClO_4$) acids prior to filtration and analysis of solute content by inductively coupled plasma mass spectrometry. Concentrations (μg P g^{-1} dry mass) were calibrated using technical blanks, 10% duplicates, spike recoveries, NIST certified samples, and an in-house standard.

Pigment Analyses

Sedimentary pigments were extracted, filtered and dried under N_2 gas following the procedures of Leavitt et al. (1989). Briefly, lipid-soluble pigments were extracted from the bulk sediments by soaking freeze-dried sediments in a mixture of acetone : methanol : water (80 : 15 : 5, by volume) for 24 h in darkness and under an inert N_2 atmosphere at 4°C. Pigment concentrations were quantified by reversed-phase high performance liquid chromatography (RP-HPLC). Specifically, carotenoid, chlorophyll (Chl), and pigment-derivative concentrations were quantified using an Agilent 1100 HPLC system following the reversed-phase procedure of Leavitt and Hodgson (2001). The Agilent 1100 system was equipped with a C-18 column (5-µm particle size; 10 cm length), and an Agilent model 1100 scanning photodiode array spectrophotometer (435-nm detection wavelength). An internal reference standard (3.2 mg $\cdot$ L^{-1}) of Sudan II (Sigma Chemical Corp., St. Louis, MO) was injected in each sample.

Pigments isolated from sediments were compared to those from unialgal cultures (Leavitt et al. 1989) and authentic standards obtained from US Environmental Protection Agency and other suppliers. Tentative pigment identity was based mainly on spectral characteristics and chromatographic mobility of pigments from all sources (Leavitt et al. 1989). Not all fossil pigments were positively identified. Consequently, we restricted our analysis to carotenoids characteristic of the following algal groups; cryptophytes (alloxanthin), siliceous algae (diatoms chrysophytes, some dinoflagellates) (fucoxanthin), mainly diatoms (diatoxanthin), chlorophytes (pheophytin b), chlorophytes and cyanobacteria (lutein-zeaxanthin), all cyanobacteria (echinenone), filamentous or colonial cyanobacteria (myxoxanthophyll), Nostocales cyanobacteria (canthoxanthin), purple sulfur (S) bacteria (okenone), and the major *a*, *b*, and *c*-phorbins (chlorophyll derivatives). Pigment concentrations for this report were expressed as nmol pigment $\cdot$ g^{-1} total C (TC), consistent with previous studies of large lakes (Bunting et al. 2007, 2011). Estimates of TC content were derived from stable isotope determinations. Finally, past UVR penetration was measured as a ratio of UVR-absorbing pigments : algal carotenoids, an index which is linearly related to the depth of UVR penetration in whole-lake experiments (Leavitt et al. 1997), while estimates of post-depositional pigment degradation were derived from analysis of ratios of precursor Chl *a* to product pheophytin *a*, as described by Leavitt and Hodgson (2001).

Algal microfossils

Cyanobacterial akinetes (resting stages) and morphological remains of chlorophyte algae were isolated from refrigerated whole sediments and prepared for microscopy following the modified protocol of Crumpton (1987). Whole-sediment samples (~1 g) were diluted with 20 mL distilled water, sonicated three times, and preserved with glutaraldehyde (0.2 mL). Samples were homogenized and ~10 aliquots (~0.10 mL) per interval were individually removed, diluted with distilled water, and fossils filtered onto a 0.45-µm pore membrane filter. Filters were mounted on cover slips using hydroxypropyl-methacrylate (HPMA) resin, air dried for 24 h, and permanently mounted onto glass microscope slides with HPMA resin. For each sample, ~100

cyanobacterial akinetes were identified and enumerated by counting random fields using an Olympus BX51 compound microscope equipped with Nomarski and phase-contrast optics, and epifluorescent detection ($\lambda_{excitation}$ = 450-480 nm). Chlorophyte microfossils were also recorded. Microfossil concentrations were estimated as fossils (akinetes, cells or colonies) g^{-1} dry mass of whole sediment. Fossils were identified to the level of genus and taxonomic identities were based on references from Bunting et al. (2007) and a standard reference collection.

Fossil diatoms

A total of 109 sediment samples from four sites (Site 1 = 47; Site 2 = 29; Site 4 = 25; Site 6 = 8) (Fig. 1) were prepared following the procedures of Batterbee et al. (2001). For each sample, a known mass of whole sediments was suspended for 24 h in 10% aqueous HCl solution to remove carbonate minerals, then washed repeatedly with deionized water before digestion of organic matter for 24 h using a mixture of concentrated nitric (HNO_3) and sulfuric acids (H_2SO_4). Residual acids were removed by repeated washes with deionized water. Samples were prepared for light microscopy by evaporating a small aliquot of the resulting diatom slurry onto a glass coverslip which was then affixed onto a glass slide using Naphax® mounting medium with a refractive index (R.I.) better than 1.74.

Preliminary determinations of the degree of diatom preservation and approximate fossil density were conducted by microscopic inspection along a single central transect in samples collected at 1-cm intervals in the master core from each site. Samples were examined using oil immersion at 1000x on a Nikon Eclipse E600 microscope equipped with differential interface contrast optics. If two unbroken diatom valves (upper or lower cell wall) were observed in the preliminary transect, then the entire slide was enumerated. If a given slide contained fewer than 200 valves, counts were discontinued because of insufficient density for accurate determination of species composition (e.g., Site 4). If sufficient fossils were present, identification and enumeration was continued until at least 480 valves (equivalent to 240 frustules) were quantified. Diatom taxonomy was based mainly on Cumming et al. (1995) to ensure consistent taxonomy between the present study and a previous comprehensive analysis of diatoms from saline lakes located in arid regions of western Canada. Appendix 1 lists taxonomic identity and relevant authority of diatoms recovered from all cores.

Results and Discussion

Radioisotope analyses and sediment chronology

All master cores were analyzed for specific activities of ^{210}Pb and ^{137}Cs (Fig. 2). At sites 1 and 4, ^{210}Pb declined in a monotonic fashion to background levels (Fig. 2a, c), whereas at site 3, an intermediate peak was noted at ~6 cm depth (Fig. 2b), representing a change in the rate of sediment accumulation. Such well-defined declines in ^{210}Pb activity suggest that sediment mixing was relatively unimportant at these Farmington and Gilbert

Bay sites, an interpretation confirmed by distinct peaks in ^{137}Cs activity at those sites (Fig. 2d-f). In this latter case, elevated specific activities of ^{137}Cs were noted in the early 1960s at sites 1 and 4, consistent with maximum atmospheric deposition of this isotope due to open-air tests of atomic weapons (~1964). In contrast, maximal ^{137}Cs activity appears to precede expected dates by 10-20 yr at site 3, either due to isotope migration, low sampling resolution (1 sample per ~15 years), or difficulty fitting ^{210}Pb regressions due to a mid-core peak of ^{210}Pb. At both sites 3 and 4, ^{137}Cs activities declined to near baseline values in surface sediments, whereas modern deposits in Farmington Bay exhibited slightly elevated ^{137}Cs activity. These latter patterns suggest either low levels of sediment mixing (but see ^{210}Pb profile above) or some degree of post-depositional migration of ^{137}Cs under conditions of profound anoxia (see below).

In contrast to sites 1, 3 and 4, there were no significant declines in ^{210}Pb activity in sediments obtained from other locations in Farmington (Site 2), Gilbert (Site 5) or Bear River Bays (Site 6) (Appendix 2). Similarly, no discrete peaks in ^{137}Cs deposition were noted at these latter sites. Finally, there were no obvious geochemical patterns within either pigment or stable isotope analyses at these sites (data not shown). Taken together, these patterns demonstrate conclusively that sediments obtained from sites 2, 5 and 6 were highly mixed and could not be used to establish either basic chronology or historical changes in algal production within Great Salt Lake. Such high variability in sediment deposition and mixing is expected in large shallow lakes (Hambright et al. 2004; Engstrom et al. 2006).

Application of the CRS dating calculation suggests that sediment cores from sites 1, 3, and 4 each spanned ~200 years, despite substantial differences in the depth of sediment collected (10 vs. 30 cm) among coring locations (Fig. 2 g-i). In general, sediment age increased smoothly with burial depth, with the exception of a slight increase in mass accumulation rates at Site 3 since ca. 1980 (Fig. 2h), and a slower rate of sediment accumulation prior to the 20th century at Site 4 (Fig. 2i). Although errors associated with sediment age increased exponentially with burial depth in all cores due to rapid declines in absolute specific activity (dpm g^{-1} dry mass), the linear nature of depth-age relationships demonstrates that all major metrics of past algal abundance (fossils g^{-1} dry mass, fossils g^{-1} total C, fossils cm^{-2} yr^{-1}) will provide equivalent information on the timing and magnitude of historical changes in lake productivity. In most cases, we have used a gravimetric estimate of past algal abundance, as these metrics are linearly related to measured changes in algal biomass in whole-lake experiments and multi-decadal time series (reviewed in Leavitt and Hodgson 2001, Bunting et al. 2007).

Given the unreliable nature of cores from sites 2, 5 and 6, the remainder of this final report will focus only on historical patterns of nutrient geochemistry (C, N, P) and algal production (pigments, chlorophyte and cyanobacterial microfossils, diatoms) derived from master cores collected at sites 1 (Figs. 3, 6), 3 (Fig. 4, 7) and 4 (Fig. 5). Together, these analyses form a coherent and convincing record of eutrophication of the southern portions of Great Salt Lake arising from a combination of wastewater influx, lake-level change, and hydrologic management.

Stable isotopes and nutrient geochemistry

Geochemical analysis of elemental composition (% dry mass) and isotopic ratios of carbon (C) and nitrogen (N) revealed common patterns at all three coring locations, each of which is consistent with recent and substantial eutrophication of GSL. In general, $\delta^{13}C$ values in the 19th century were enriched, relatively stable, and characteristic of carbonate minerals (ca. 0 to -5‰) (Figs. 3-5; panel t). At each site, these C isotope ratios became substantially depleted towards values characteristic of algae (ca. -20 to -30‰), with the most pronounced change commencing early in the 20th century and accelerating after ca. 1960. Although modern C isotopes are still enriched relative to pure algal matter, the shift in C isotopes is consistent with increased deposition of algal organic matter (-20 to -25‰ in GSL; Wurtsbaugh et al. 2008). In addition, concomitant increases in the sedimentary content of N (%N) (panel q) and declines in the mass ratio of C : N of bulk sediments (panel u) of all three cores are also characteristic of elevated deposition of organic matter (high N content) of algal origin (algal C : N <12 : 1). Similar changes have been recorded in other large lakes experiencing increased algal production (Leavitt et al. 2006, Engstrom et al. 2006, Bunting et al. 2007). The remarkable extent and similarity of change among cores suggests a pattern of eutrophication that is consistent throughout much of the southern half of GSL, although variations in exact timing of the major changes suggest variation in onset of algal response among Gilbert and Farmington bays (see below).

Sedimentary concentrations of phosphorus (P) also increased during the 20th century relative to values recorded before 1900 (panel p in Figs. 3-5). In general, P content in sediments from Gilbert Bay increased 50-75% early in the 20th century, reached a plateau between ca. 1930-1980, then declined slightly during the past 20-30 years (Figs. 4-5). In contrast, P concentrations within the Farmington Bay core (Site 1; Fig. 3) increased only after ca. 1960, to reach values nearly three-fold greater than historical baseline levels in recently deposited sediments. Because P can be mobile in modern (<5 yr) sediments or those with low oxygen content in overlying water, interpretation of the most recently deposited material should be cautious. Nonetheless, consistent, coeval and pronounced increases in sedimentary concentrations of both P and N (see above) are consistent with elevated nutrient influx resulting in increased production and sedimentation of organic material.

Stable isotope ratios of N ($\delta^{15}N$) exhibited little systematic variation with burial depth in any core (panel s in Figs. 3-5). During the 19th century, values at all sites were enriched (ca. 10-15 ‰) relative to those seen in sediments of unpolluted lakes (<6‰). However, similar elevated values have been recorded for arid regions in previous paleoecological analysis (Rusak et al. 2004), and presumably reflect increased N cycling in dry climates, leading to progressive loss of N due to denitrification, ammonia volatilization, or other processes. The absence of pronounced historical trends in $\delta^{15}N$ values suggests that elevated content of ^{15}N does not arise from pollution of the lake with urban or agricultural N, as has been recorded elsewhere (Leavitt et al. 2006, Bunting et al. 2007). However, because N isotope values during the early 19th century are similar to

those recorded in both modern samples and those characteristic of N pollution by humans (10-20‰), it appears that the overall degree of isotopic enrichment of ^{15}N in GSL arises mainly as a result of the prevalent (arid) climatic conditions.

Fossil pigments

Analysis of sedimentary carotenoids, chlorophylls and their derivatives revealed consistent evidence of eutrophication of the southern embayments of GSL, as well as marked differences among Farmington and Gilbert bays in terms of the timing of the maximum extent of algal population expansions (Figs. 3-5). At all three sites, concentrations of pigments from most algal groups increased by between 3- and 10-fold early in the 20th century relative to the mid-1800s. These patterns are not consistent with degradative processes (first-order loss with increased age) and instead suggest that historical variation in concentrations arose from true increases in algal abundance rather than an exponential degradation of sedimentary pigments following burial (reviewed in Leavitt 1993). Consistent with this interpretation, ratios of labile precursor Chl *a* to chemically-stable product pheophytin *a* varied little with burial depth at Gilbert Bay sites (panel l in Figs. 4-5), demonstrating that there was little change in pigment preservation though the 200 year fossil record. Although there was evidence of post-depositional transformation of pigments within the Farmington Bay core (Fig. 3l), signatures of potential degradation were restricted to the uppermost 3-4 samples (~25 yr), and could not account for changes in fossil pigment abundance observed in deeper samples. Taken together, these fossil patterns are very similar to those observed for other lakes undergoing substantial and sustained increases in algal production during eutrophication (reviewed in Leavitt et al. 2006; Bunting et al. 2007).

Despite similarities in the timing of the onset of eutrophication (late 1800s), comparison among basins suggests that eutrophication during the early 20th century was more severe in Gilbert Bay than in Farmington Bay, but that water quality may have improved in Gilbert Bay since ca. 1970 as a result of changes in water exchange between the two embayments. For example, most pigment concentrations in cores recovered from sites 3 and 4 in Gilbert Bay (Fig. 4, 5) increase ~10-fold to maxima in the early 20th century after wastewater was initially released into the lake (1880s; Table 1). During this 50-yr interval, algal populations expanded at near-exponential rates, whereas indices of the degree of pigment preservation actually exhibited a modest decline (panel l). Overall, pigment levels in Gilbert Bay remain elevated until ~1960, after which time fossil concentrations decline either slowly (Site 3) or more dramatically (Site 4) to lower, but still elevated concentrations. Given the limits of temporal resolution of our analysis (1960 ± 4 yr), the partial recovery of southern Gilbert Bay appears to coincide with hydrological closure of Farmington Bay due to a combination of lake-level decline and construction of causeways at both the south (1952) and the north (1969) end of Farmington Bay (Table 1). Consistent with this interpretation, the most rapid increase in fossil pigment concentrations at Farmington Bay also occurred only after 1960 (Fig. 3, panels a-k) concomitant with its more complete isolation from the main lake basins and the formation of North, South and Central Davis sewer districts to deliver wastewater to Farmington Bay (1959-1962).

Historic variations in fossil pigment abundance at a given core location were similar among major taxonomic groups of algae (panels a-k in Figs. 3-5). For example, timing and magnitude of changes in past abundance at Farmington Bay were similar among pigments derived from diatoms (Fig. 3b), cryptophytes (Fig. 3c), Nostocales cyanobacteria (Fig. 3g) and indicators of colonial cyanobacteria (Fig. 3f) or combined chlorophyte-cyanobacterial pigments (Fig. 3e), whereas labile compounds from siliceous (Fig. 3a) and total algae (Fig. 3i) exhibited more pronounced changes in surface sediments consistent with post-depositional degradations (Fig. 3l). Interestingly, analysis of biomarkers representative of total cyanobacterial abundance (Fig. 3h) suggested that these algal have been common in Farmington Bay for much of the past 200 years. Similar agreement among groups of pigments was also recorded in Gilbert Bay cores, with labile compounds (fucoxanthin, Chl *a*) exhibiting a more limited decline towards baseline conditions after 1960 relative to patterns exhibited by other, more chemically-stable fossil pigments (Fig. 4, 5; panels a, i).

Changes in lake-level elevation associated with climatic variability and catchment-scale management of hydrologic fluxes do not appear to have substantially biased the sedimentary record of water-quality change at any core location. Specifically, although concentrations of many fossil pigments (panels a-k) were correlated negatively with historical changes in water-column depth (panel o in Figs. 3-5), these relationships were not statistically significant ($P > 0.10$) for all pigments at all sites with the exception of myxoxanthophyll (colonial cyanobacteria) and alloxanthin (cryptophytes) at Site 4. As demonstrated through whole-lake mass balance analyses (reviewed in Leavitt 1993; Leavitt and Hodgson 2001), negative correlations between water-column depth and fossil pigment concentration are expected and can arise because most pigment degradation occurs during sinking of moribund algae to the sediments. In such a case, deeper water engenders more pigment mineralization during sinking and less fossil preservation for a given level of algal production. Fortunately, these mechanisms would be expected to alter pigment deposition only ~25-50% over the range of water column depths observed in GSL (~5 m since 1840) (Leavitt and Hodgson 2001). This range of variation is much less than the 10-fold range in pigment levels observed in GSL sediments during the past 200 years, and demonstrates that most variation in fossil concentration could not be attributed to lake-level alterations.

Prolonged declines in water levels during 1940-1960 may have altered the irradiance regime and oxygen penetration into the sediment-water interface of shallow Farmington Bay (Fig. 3). For example, comparison of historical lake elevation with modern water-column depth at coring Site 1 suggested that Farmington Bay was extremely shallow during several years of the mid-20th century (Fig. 3o). Concomitant with the timing of this low stand ca. 1940-1960, benthic cyanobacteria deposited UVR-absorbing photo-protective pigments at concentrations typical of extremely UVR-stressed environments (Leavitt et al. 1997). Because of the high metabolic costs associated with synthesis and export of these extracellular pigments, cyanobacteria only produce photo-protectant compounds when cells cannot escape intense irradiance through deepwater refugia (Leavitt et al. 1997, Leavitt and Hodgson 2001). Although Farmington Bay is

presently rich in dissolved organic matter, these compounds are typically poor at UVR-attenuation in saline lakes (Vinebrooke et al. 1998). Instead, it appears that low lake levels in Farmington Bay may have aided oxygen penetration into the sediments, thereby constraining growth of obligate anaerobes such as purple sulfur bacteria (Fig. 3m). For example, concentrations of their biomarker pigment okenone increased early in the 20th century, and remained nearly constant until present day, with the exception of a 10-fold decline in fossil concentration during the Farmington Bay low stand. Overall, okenone concentrations were substantially lower than those seen in strongly-stratified lakes with well-lit zones of permanent anoxia (Leavitt et al. 1989). However, the continuous presence of this compound in sediments since ca. 1900, combined with stable indices of pigment preservation (Fig. 3l), strongly suggest that this core location has not experienced complete desiccation during the past 100 years.

Algal Microfossils

Sediments from Farmington Bay Site 1 revealed morphological remains from nine genera of algae, including three cyanobacteria (*Anabaena*, *Gloeotrichia*, *Nodularia*) and six chlorophytes (*Cosmarium*, *Pediastrum*, *Scenedesmus*, *Teilingia*, *Tetrahedron*, *Xanthidium*). Of these taxa, four genera occurred at low densities (< 5000 fossils g^{-1} dry mass) and in only 1-2 of the 35 samples enumerated, including *Nodularia* (1992, 2002), *Scenedesmus* (1978, 1985), *Telingia* (1978), and *Tetrahedron* (1931, 1966). In contrast, the cyanobacteria *Anabaena* (Fig. 3v) and *Gloeotrichia* (Fig. 3w) were common through much of the analytical record, particularly during the first half of the 20th century when the remaining chlorophytes were also abundant (Fig. 3x). The presence of appreciable densities of potentially-N_2-fixing *Anabaena* and *Gloeotrichia* spp. since 1800 suggests both that Farmington Bay was naturally productive prior to Mormon colonization of the catchment, and that N supply may have limited historical growth of algae at that site.

Sharp declines in concentrations of morphological fossils from *Gloeotrichia* (Fig. 3w) and green algae (Fig. 3x) after ca. 1970 coincided with greatly elevated concentrations of biochemical fossils from many algal groups (Fig. 3a-k). This sequence of replacement is consistent with that observed in other large, shallow lakes undergoing progressive eutrophication with N and P (Bunting et al. 2007; Bunting et al. 2011). As reviewed in Bunting et al. (2007), initial stages of eutrophication is often marked by increased densities of *Gloeotrichia*, a taxon capable of acquiring nutrients from sedimentary sources and translocating them into the water column where they further fuel increases in primary production. However, with continued increase in nutrient influx, these meroplanktonic (part benthic, part planktonic) taxa appear to be outcompeted by positively-buoyant or low-light adapted cyanobacteria, such as *Planktothix*, *Microcystis*, and *Nodularia*. Consistent with this scenario, remains of *Nodularia* were recorded only in sediments deposited since 1992, although at present it is not possible to determine whether this pattern represents low abundance of this taxon at Site 1, relatively recent onset of these blooms, relatively poor preservation of *Nodularia* in Farmington Bay sediments.

Sediments from Gilbert Bay preserved few morphological fossils from cyanobacteria or chlorophyte algae (Fig. 5 v-w) relative to those of Farmington Bay (Fig 3 v-x). This pattern is particularly remarkable give that fossil pigment concentrations were nearly 1000% greater at Gilbert Bay than at the Farmington site. We speculate that the relatively high rate of sediment accumulation at the Farmington Bay location (Fig. 2g) relative to that at Sites 3 and 4 in Gilbert Bay (Fig. 2h-i) may have buried organic matter more rapidly within oxygen-poor surface sediments, thereby aiding preservation of delicate fossils. Consistent with this interpretation, ratios of labile precursor Chl *a* to chemically-stable product pheophytin *a* (Chl *a*: Pheophytin *a*) were generally greater in Farmington Bay sediments (~1; Fig. 3l) than in Gilbert Bay deposits (<0.5; Figs. 4l, 5l).

Fossil Diatoms

Sediments deposited in GSL during the past 200 years contained a total of 146 species of diatoms (Appendix 1), including ~50 species in most samples, and a predominance of non-planktonic diatoms characteristic of benthic habitats or other substrates (Fig. 6). In general, the observed level of species richness (24-56 species within ~500 enumerated specimens) is low relative to that seen in shallow freshwater lakes of moderate productivity, but is similar to that observed in other saline lakes (e.g., Rusak et al. 2004). Unfortunately, overall preservation of fossil diatoms was poor at most coring locations in GSL, with reliable densities of valves present only in Farmington Bay sediments deposited since ca. 1960 (Fig. 6) and Gilbert Bay Site 3 sediments deposited after ca. 1970 (Fig. 7). Recognizable remains were nearly absent from all sediments deposited before 1960, despite generally excellent preservation of pigments at all sites. In fact, of the 109 samples examined with light microscopy, only 18 contained sufficient densities of adequately-preserved diatoms to quantify community composition. Although degradation of diatoms in saline lake sediments is under complex multi-factorial control (Barker et al. 1994), diatom preservation tends to improve under conditions of profound anoxia, such as might occur during the most intensive phases of algal production and eutrophication (i.e., post-1960 in Farmington Bay; Fig. 3a-k), following development of the deep brine layer in Gilbert Bay, or in recently deposited sediments. We interpret that the absence of diatoms early in the 20th century is due to a lack of frustule preservation, rather than the absence of diatoms from the lake's flora, because diatom-specific pigment diatoxanthin was present at concentrations above detection limits throughout cores from all sites (Figs. 3b-5b).

Diatom community composition changed rapidly in Farmington Bay during the past 50 years (Fig. 6). In particular, *Fragilaria construens v. pumila* declined from 65% of the fossil diatom sum in the mid 1960s to <1% in material deposited since 2004, while several *Navicula* species increased to 10-20% of the sub-fossil assemblage in recently deposited sediments (*N. pupula*, *N. veneta*, *N. menisculus*). In contrast, other representatives of the genera *Fragilaria* (*F. brevistrata*, *F. construens v. venter*, *F. pinnata*) and *Navicula* (*N. cincta*, *N. cryptotenella*, *N. begerii*, *N. sp. 6 PISCES fo. 2*, *N. clementis*), as well as conspecifics of the genera *Amphora* and *Nitzchia*, and the planktonic diatoms *Cyclotella meneghiniana* and *Stephanodiscus parvus*, revealed few pronounced changes in past relative abundance. Taken together, these changes are most

consistent with the effect of historical variation in lake-water salinity (Cumming et al. 1995), possibly arising from declines in lake circulation and altered salinity in the southern basins due to the 1959 construction of the railroad causeway separating Gilbert and Farmington bays from the remainder of the lake, as well as modest increases in Farmington Bay water level due to the1969 construction of the causeway to the north end of Antelope Island. As well, variation in abundance of *F. construens v. pumila* may also indicate recent changes in the metals content of waters in Farmington Bay, as this taxon exhibits high tolerance to metal exposure (Cattaneo et al. 2011).

Evaluation of historical changes in diatom community composition at Gilbert Bay Site 3 was limited to the period ca. 1974-present because of an absence of diatom fossils in older sediments (Fig. 7). Because of low rates of sediment accumulation relative to Farmington Bay, diatoms were recovered from only 7 samples in the 30-yr interval, although species richness was only slightly lower than that observed at Site 1 (Appendix 1). Once again, we interpret that the absence of well-preserved diatoms in sediments preceding ~1970 reflects dissolution of the siliceous frustules (Barker et al. 1994) because high concentrations of diatom- and siliceous algal-specific pigments were abundant at this site from ~1900 onwards (Fig. 4a, b).

In general, the fossil diatom assemblage at Gilbert Bay Site 3 was composed of species with very high tolerance to salt concentrations. In addition, these taxa are known to have salinity optima (preferred conditions) greater than those of many diatoms recovered from Farmington Bay sediments (Cumming et al. 1995). Overall, the fossil assemblage exhibited few dramatic changes in species composition, with salt-tolerant *Amphora acutiuscula* declining from ~50% of the diatom sum to ~30% in the most recently deposited samples (Fig. 7). In addition, subtle variations in species composition suggest that historical changes in fossil assemblages were consistent with declines in taxa that occur in highly saline, nutrient-rich waters (*Navicula cincta*, *Nitzschia communis* and *Navicula* sp. 6 PISCES fo. 2) in favour of those found in less eutrophic conditions (*Aulacoseira ambiqua*, *Cyclotella menegiana*, small benthic *Fragilaria* spp.). Unfortunately, because diatom preservation was restricted to sediments deposited since ~1970, a period following major hydrologic and wastewater management changes (1959, 1962, 1969; Table 1), it is difficult to interpret the ecological meaning of these fluctuations or attribute the variation to specific causal mechanisms.

Synthesis and Conclusions

Taken together, analysis of sedimentary pigments, geochemistry, and soft algal fossils revealed a consistent pattern of eutrophication in Farmington and Gilbert bays of GSL. As well, analysis of fossil diatoms suggests that algal species composition was responsive to changes in lake-water salinity and metal content. Pigment-reconstructed algal abundance at three sites with reliable chronologies increased during the late-1800s (Figs 3-5, panels a-k), concomitant with the 1889 construction of septic disposal systems to introduce wastewater directly into GSL (Table 1). Elevated algal production is indicated also by pronounced depletion of $\delta^{13}C$ isotope values (panel t), declines in bulk

sedimentary C: N ratio (panel u), and increased N content of sediments (panel q) during the early 20th century. Unlike the carbonate-rich bulk sediments of saline lakes, algal biomass is characterized by depleted ^{13}C content ($\delta^{13}C_{GSL\ algae}$ = -20 to -25‰; $\delta^{13}C_{carbonate}$ = 0 to -5‰) and low C : N mass ratios (8-12), and is often the main source of N to the sediments (Bunting et al. 2007 and references therein). Interestingly, while water-quality degradation was restricted to the 20th century, quantification of sedimentary akinetes revealed that bloom-forming cyanobacteria (*Anabaena*, *Gloeotrichia*) have been present in the lake since at least 1800 (Figs. 3, 4), well before substantial social and economic development by non-native colonists.

Overall, initial increases in algal abundance during the early 20th century were apparently more pronounced at Gilbert Bay (Sites 3 and 4) than at Farmington Bay (Site 1). At Farmington Bay, initial eutrophication appears limited to development of cyanobacteria (Fig. 3f, h), particularly *Gloeotrichia* spp. (Fig. 3w) rather than *Anabaena* (Fig. 3v), and only select green algae (*Cosmarium*, *Pediastrum*, *Xanthidium*) (Fig. 3x) rather than entire assemblages of chlorophytes (Fig. 3d). Such transient blooms of green algae and *Gloeotrichia* have been reported for other large shallow lakes undergoing the first stages of eutrophication (Bunting et al. 2007, 2011). In contrast, initial algal expansion at Gilbert Bay (Figs 4-5, panels a-k) was up to 10-fold above than baseline values seen in the 1800s, reaching maxima during the first half of the 20th century.

Causeway construction and lake-level decline may have altered hydrologic exchange between Farmington and Gilbert bays and influenced the initial rates of eutrophication in the two embayments. For example, although algal abundance increased in Farmington Bay during the early 20th century concomitant with deposition of cyanobacterial microfossils, the most rapid phase of eutrophication occurred after 1960. Within the limits of our sample resolution (5 mm) and chronological errors (Fig. 2g), timing of algal expansion is coeval with the construction of the northern automobile causeway to Antelope Island (completed in 1969) and reduced exchange of water between Farmington and Gilbert bays (Fig. 3). Although temporal resolution of Gilbert Bay cores is lower than that of Farmington Bay due to low rates of sediment accumulation (Fig. 2h, i), algal abundance also appears to have declined at Site 4 (Fig. 5a-k) just as that of Farmington Bay increased (Fig. 3a-k). Isolation of the two southernmost locations (sites 1 and 4) may have been further enhanced both by lake-level decline and emergence of mudflats south of Antelope Island, and by construction of the southern causeway to Antelope Island in 1952. In contrast, the more limited algal recovery at site 3 in Gilbert Bay may reflect its closer proximity to Farmington Bay, effects of construction of the railroad causeway in 1959, or other as-yet-unknown factors.

Establishment of secondary wastewater treatment facilities in Salt Lake City by 1965 has not notably improved water quality or reduced algal biomass in Farmington Bay. Algal biomass at all three coring locations remains 5- to 10-fold higher than baseline levels characteristic of the mid-1800s, although southern portions of Gilbert Bay appear to be experiencing an ongoing recovery (Fig. 5). Conventional secondary treatment removes particulate and dissolved organic matter from wastewater, but does not reduce outfall of dissolved inorganic elements including P and N. At present, we cannot

determine whether elevated sedimentary content of N and P reflects this increased influx, or is simply the result of increased deposition of N- and P-rich algal matter. Similarly, it is not possible to evaluate the significance of the sharp increase in bulk sediment $\delta^{15}N$ values in Farmington Bay sediments after ca. 1960 (Fig. 3s), despite declining values in Gilbert Bay cores (Figs. 4-5s). In other large lakes, such enrichment is consistent with volatilization of excess ammonia from N-rich waters, as well as microbial transformation processes, including denitrification which converts excess nitrate to N_2 or N_20 gases (Bunting et al. 2007).

In conclusion, GSL is experiencing continuing substantial and continuing eutrophication of surface waters in Farmington and Gilbert bays, most likely due to on-going influx of incompletely-treated wastewaters. In addition, the timing of algal population expansion among sites appears to be related in part to hydrologic management associated with construction of causeways. In particular, construction of causeways to Antelope Island appears to have constrained the most severe eutrophication to Farmington Bay and may have reduced the magnitude of eutrophication in southern Gilbert Bay. In contrast, wastewater management strategies appear to have had limited beneficial effects in controlling algal growth, probably because the technology associated with secondary wastewater treatment is ~50 year behind state-of-the-art techniques (Biological Nutrient Removal). Although changes in water influx may continue modifying algal production in southern GSL, water quality is unlikely to improve substantially until nutrient influxes are better controlled.

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885 Table 1. Major hydrologic, industrial, and wastewater events in the Great Salt Lake basin, 1847-1992.

Year	Event
1847	Mormon pioneers settle Salt Lake Valley
1863	Copper mining begins at Bingham Mine; intensifies in 1873
1873	Lake reaches high level (1283.5 m); salinity decreases to ~136 g L^{-1}
1885	Surplus Canal constructed that diverts much of Jordan River directly to Gilbert Bay, thus directing nutrients away from Farmington Bay
1889	First sewer line in Salt Lake City (SLC) to the Jordan River; flow increased to ~52 x 10^6 L d^{-1} by 1908
1892	First smelter for gold, silver and lead
1911	Outlet Sewage Canal to Farmington Bay completed; Wastewater discharge into Jordan River discontinued
1952	South causeway to Antelope Island constructed; prevents wastewater from reaching south end of Gilbert Bay
1952	High influx of water from Jordan and Weber Rivers
1959	Railroad Causeway completed to separate Gilbert and Gunnison Bays. Surface water salinity decreases in Gilbert Bay, but deep brine layer begins to form
1959-1962	Sewerage districts formed to discharge wastewater into Farmington Bay and its tributaries including, North Davis Metropolitan Sewer (~72 x 10^6 L d^{-1}), South Davis Sewer Treatment Plants (total ~128 x 10^6 L d^{-1}), and Central Davis Sewer District (~20 x 10^6 L d^{-1})
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 is 1282.14 m
1985	Water level in Gilbert Bay reaches 1282.86 m; salinity declines to 58 g L^{-1}; Automobile causeway to Antelope Island flooded until 1989
1992	Automobile causeway rebuilt

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Figure legends

Fig. 1. Map of Great Salt Lake Utah, including core 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).

Fig. 2. Sediment chronology and radioisotope activity profiles for Farmington Bay (left column), Gilbert Bay Site 3 (centre column), and Gilbert Bay Site 4 (right column). Data presented include ^{210}Pb activity (dpm g^{-1} dry mass) in top row, ^{137}Cs activity (dpm g^{-1} dry mass) in middle row, and estimated year of sediment deposition in bottom row. All ranges represent mean $\pm$ 1 standard error.

Fig. 3. Historical changes in fossil pigment concentrations (nmol pigment g^{-1} total carbon) and other algal parameters since ca. 1800 in sediments collected from Farmington Bay site 1. Pigments include a) fucoxanthin from siliceous algae, b) diatoxanthin from mainly diatoms, c) alloxanthin from cryptophyte algae, d) pheophytin *b* from chlorophytes, e) sum of lutein from chlorophytes and zeaxanthin from cyanobacteria, f) myxoxanthophyll from some colonial cyanobacteria, g) canthoxanthin from Nostocales cyanobacteria, h) echinenone from all cyanobacteria, i) chlorophyll *a* from all algae, j) pheophytin *a* from all algae, k) β-carotene from all algae, and m) okenone from purple sulfur bacteria. Other parameters include l) pigment preservation index (ratio Chl *a* : pheophytin *a*), n) index of exposure to UV radiation , o) estimated lake depth at this coring site, p) sedimentary P concentration (μg P g^{-1} dry mass), q) sedimentary N content (% dry mass), r) sedimentary C content (% dry mass), s) δ^{15}N values for whole sediment (‰), t) δ^{13}C values for whole sediment (‰), u) mass ratio of C:N of whole sediments, v) concentration of akinetes from *Anabaena* spp. (fossils g^{-1} dry mass), w) concentration of akinetes from *Gloeotrichia* spp. (fossils g^{-1} dry mass), and x) concentration of cells or colonies from chlorophyte spp. (fossils g^{-1} dry mass). See text for details.

Fig. 4. Historical changes in fossil pigment concentrations (nmol pigment g^{-1} total carbon) and other algal parameters since ca. 1800 in sediments collected from Gilbert Bay site 3. Pigments include a) fucoxanthin from siliceous algae, b) diatoxanthin from mainly diatoms, c) alloxanthin from cryptophyte algae, d) pheophytin *b* from chlorophytes, e) sum of lutein from chlorophytes and zeaxanthin from cyanobacteria, f) myxoxanthophyll from some colonial cyanobacteria, g) canthoxanthin from Nostocales cyanobacteria, h) echinenone from all cyanobacteria, i) chlorophyll *a* from all algae, j) pheophytin *a* from all algae, k) β-carotene from all algae, and m) okenone from purple sulfur bacteria. Other parameters include l) pigment preservation index (ratio Chl *a* : pheophytin *a*), n) index of exposure to UV radiation , o) estimated lake depth at this coring site, p) sedimentary P concentration (μg P g^{-1} dry mass), q) sedimentary N content (% dry mass), r) sedimentary C content (% dry mass), s) δ^{15}N values for whole sediment (‰), t) δ^{13}C values for whole sediment (‰), and u) mass ratio of C:N of

whole sediments. No soft algal remains were recorded at this site. See text for details.

Fig. 5. Historical changes in fossil pigment concentrations (nmol pigment g^{-1} total carbon) and other algal parameters since ca. 1800 in sediments collected from Gilbert Bay site 4. Pigments include a) fucoxanthin from siliceous algae, b) diatoxanthin from mainly diatoms, c) alloxanthin from cryptophyte algae, d) pheophytin *b* from chlorophytes, e) sum of lutein from chlorophytes and zeaxanthin from cyanobacteria, f) myxoxanthophyll from some colonial cyanobacteria, g) canthoxanthin from Nostocales cyanobacteria, h) echinenone from all cyanobacteria, i) chlorophyll *a* from all algae, j) pheophytin *a* from all algae, k) β-carotene from all algae, and m) okenone from purple sulfur bacteria. Other parameters include l) pigment preservation index (ratio Chl *a* : pheophytin *a*), n) index of exposure to UV radiation , o) estimated lake depth at this coring site, p) sedimentary P concentration (μg P g^{-1} dry mass), q) sedimentary N content (% dry mass), r) sedimentary C content (% dry mass), s) $\delta^{15}N$ values for whole sediment (‰), t) $\delta^{13}C$ values for whole sediment (‰), u) mass ratio of C:N of whole sediments, v) concentration of akinetes from *Anabaena* spp. (fossils g^{-1} dry mass), and w) concentration of akinetes from *Gloeotrichia* spp. (fossils g^{-1} dry mass) . Cyanobacterial and chlorophyte microfossils are considered unreliable due to infrequent occurrence (few samples) and low densities within individual samples. See text for details.

Fig. 6. Relative abundance (% fossil sum) of the main diatom species recovered from sediments collected at Farmington Bay site 1. Note: diatom preservation was poor prior to ca. 1960, and diatom abundance could not be estimated. See text for details.

Fig. 7. Relative abundance (% fossil sum) of the main diatom species recovered from sediments collected at Gilbert Bay site 3. Note: diatom preservation was poor prior to ca. 1970, and diatom abundance could not be estimated. See text for details.

970 Fig. 1. Morphometric map of Great Salt Lake, Utah.

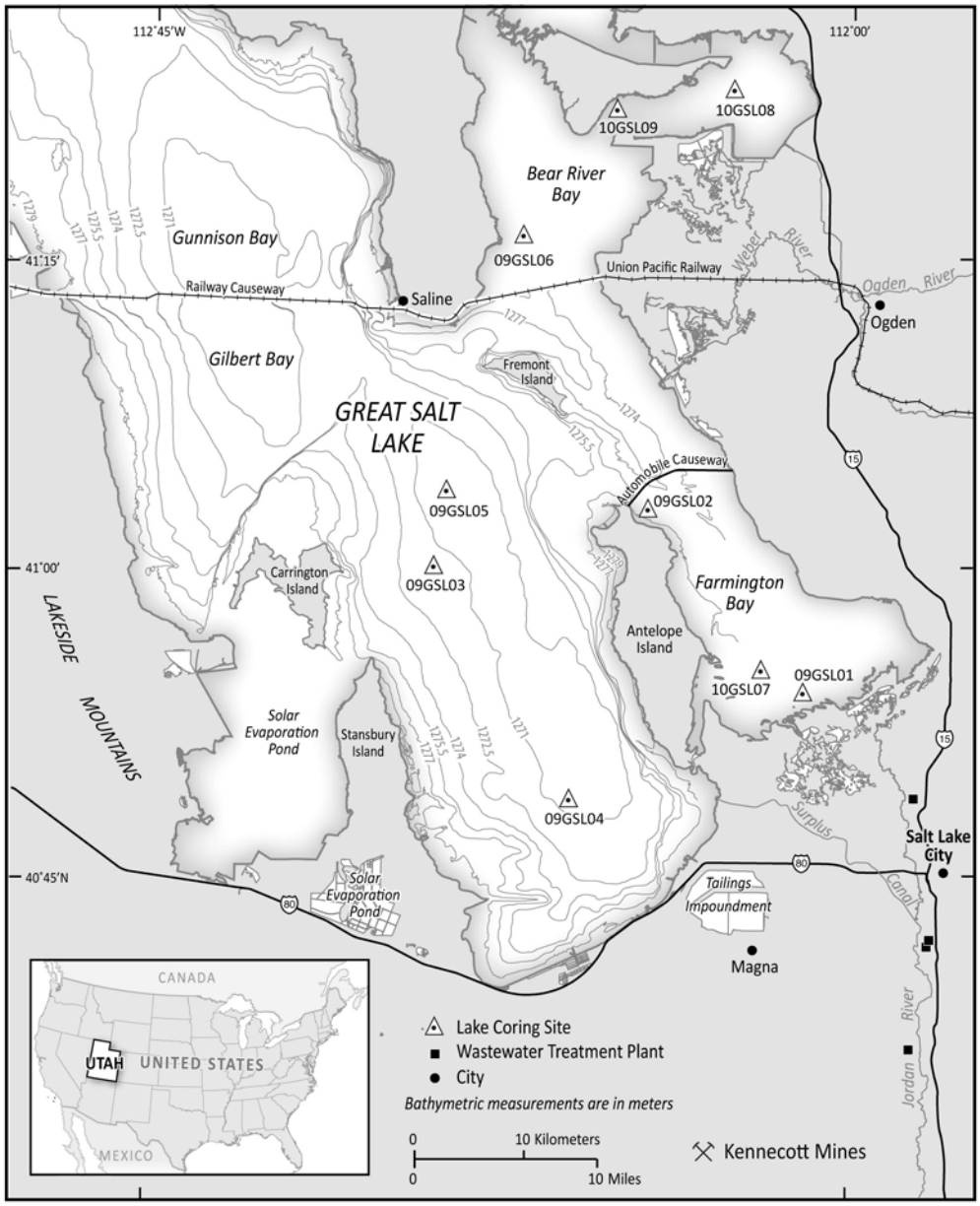


Fig. 2. Sediment chronology

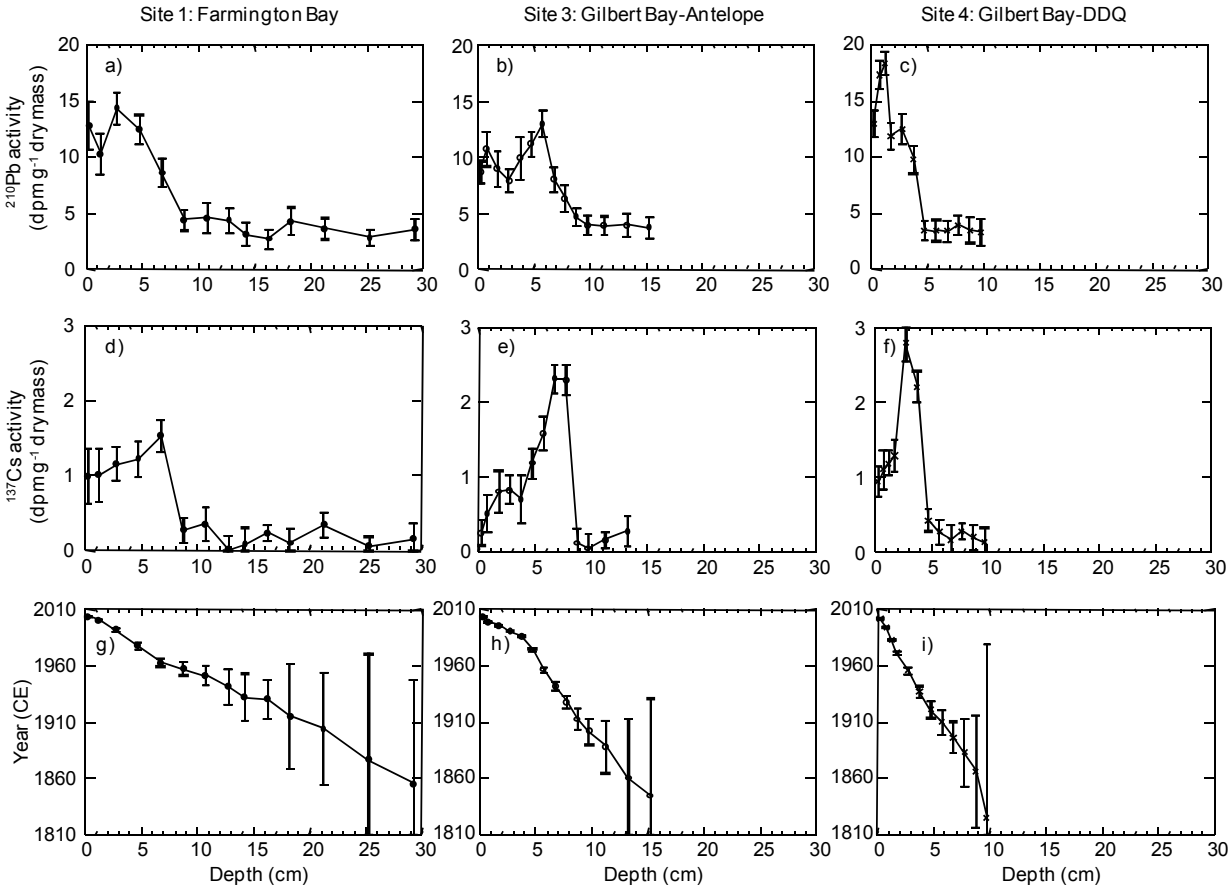


Fig. 3. Pigments Farmington Bay, Site 1

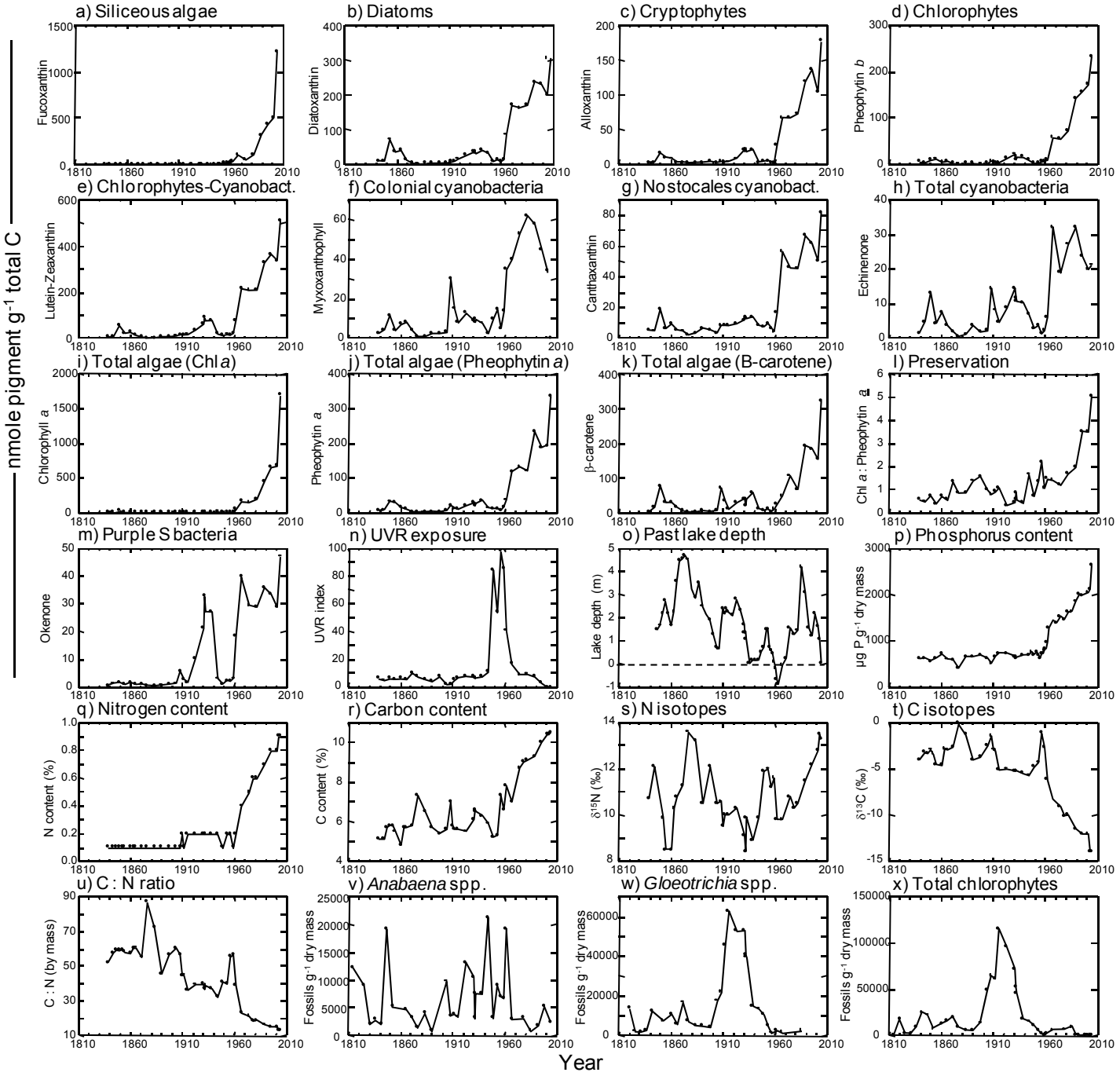


Fig. 4. Pigments
Gilbert Bay, Site 3

975

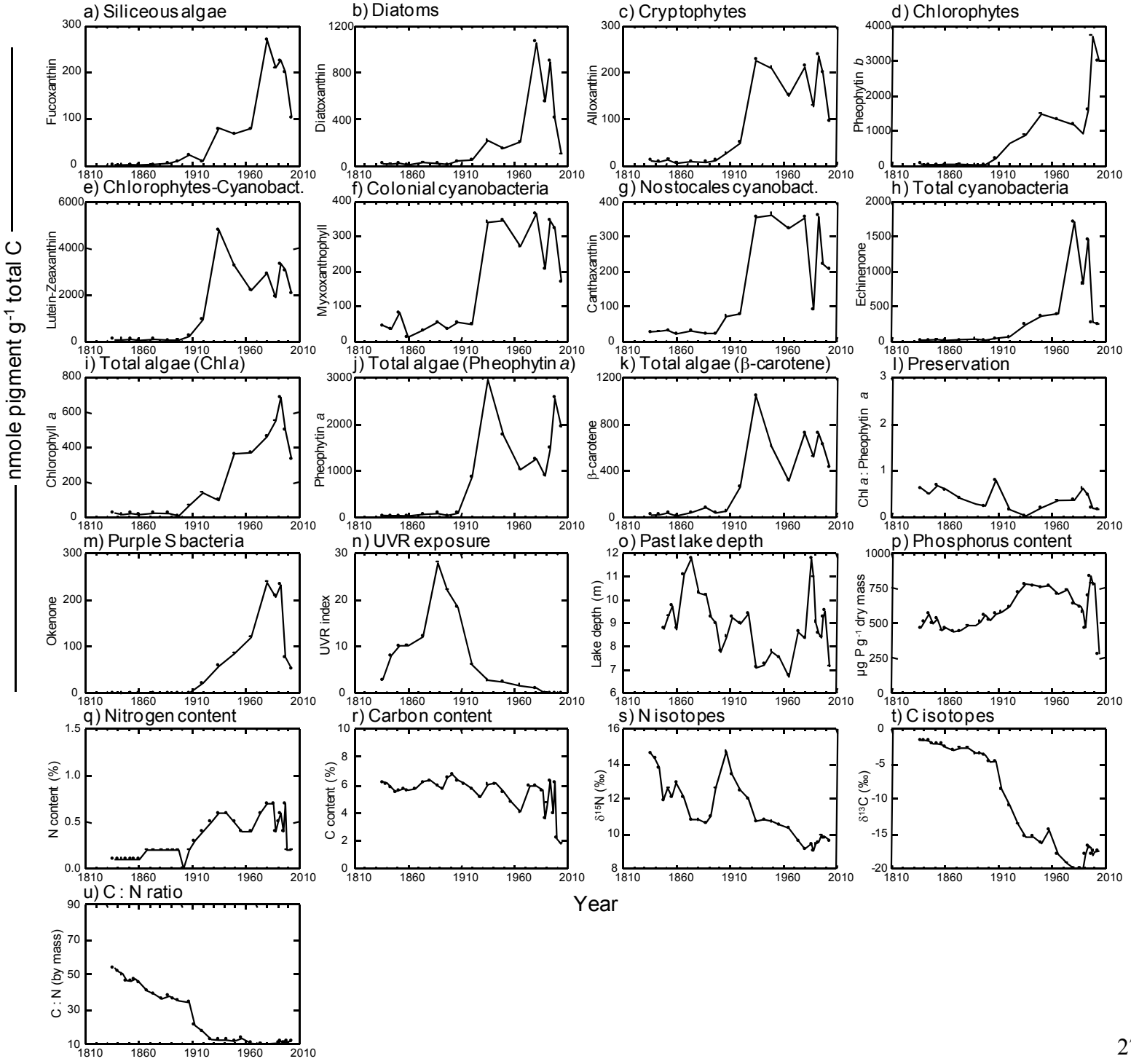


Fig. 5. Pigments
Gilbert Bay, Site 4

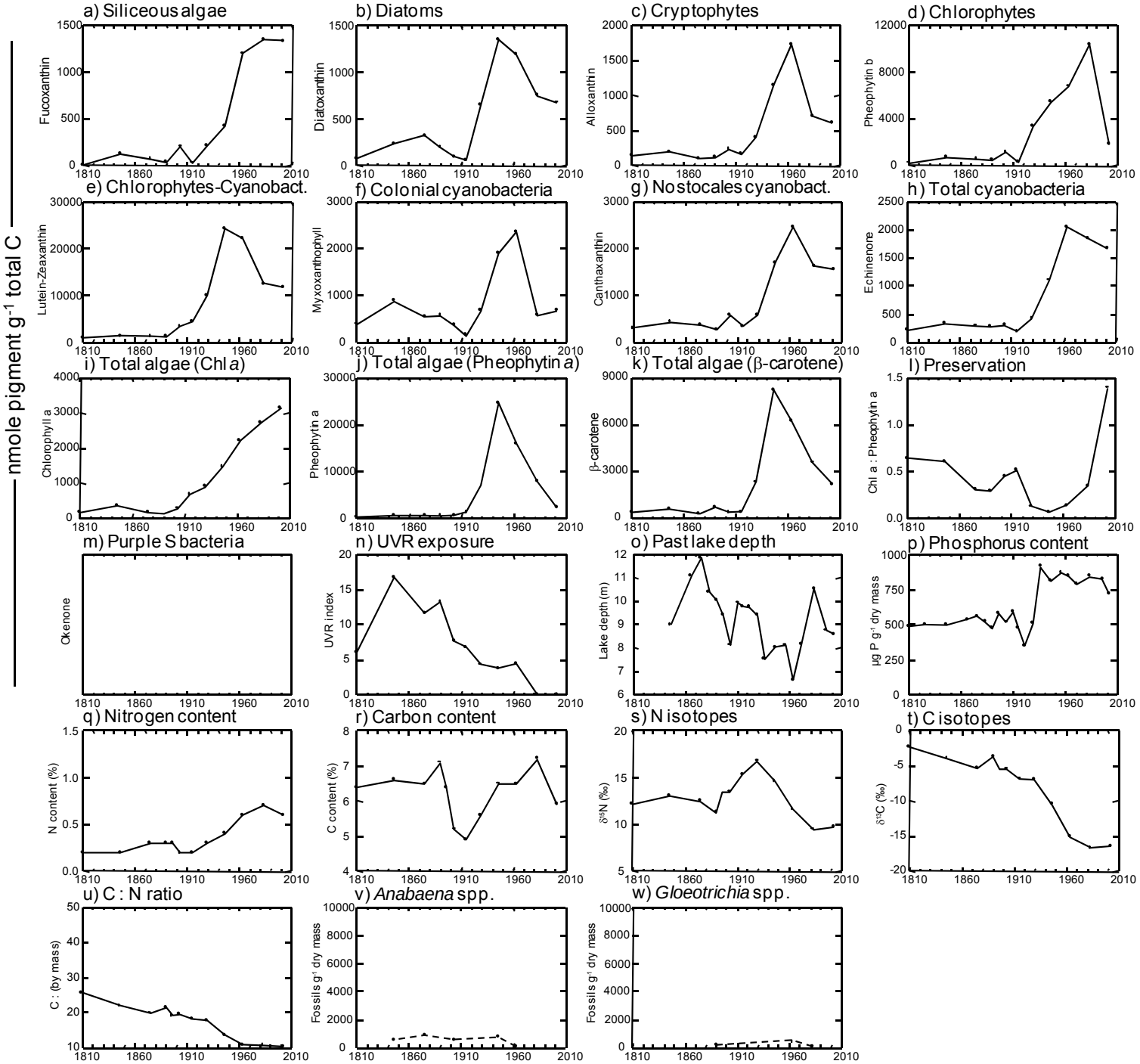


Fig. 6. Sedimentary diatoms, Farmington Bay, Site 1

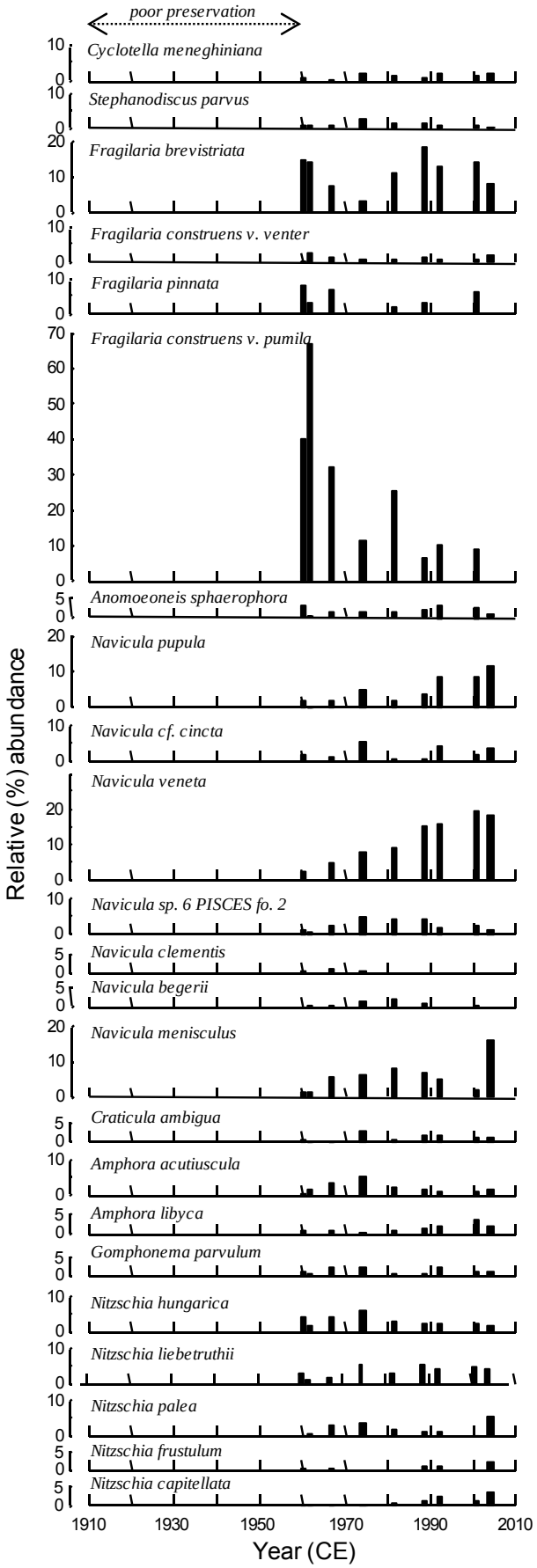
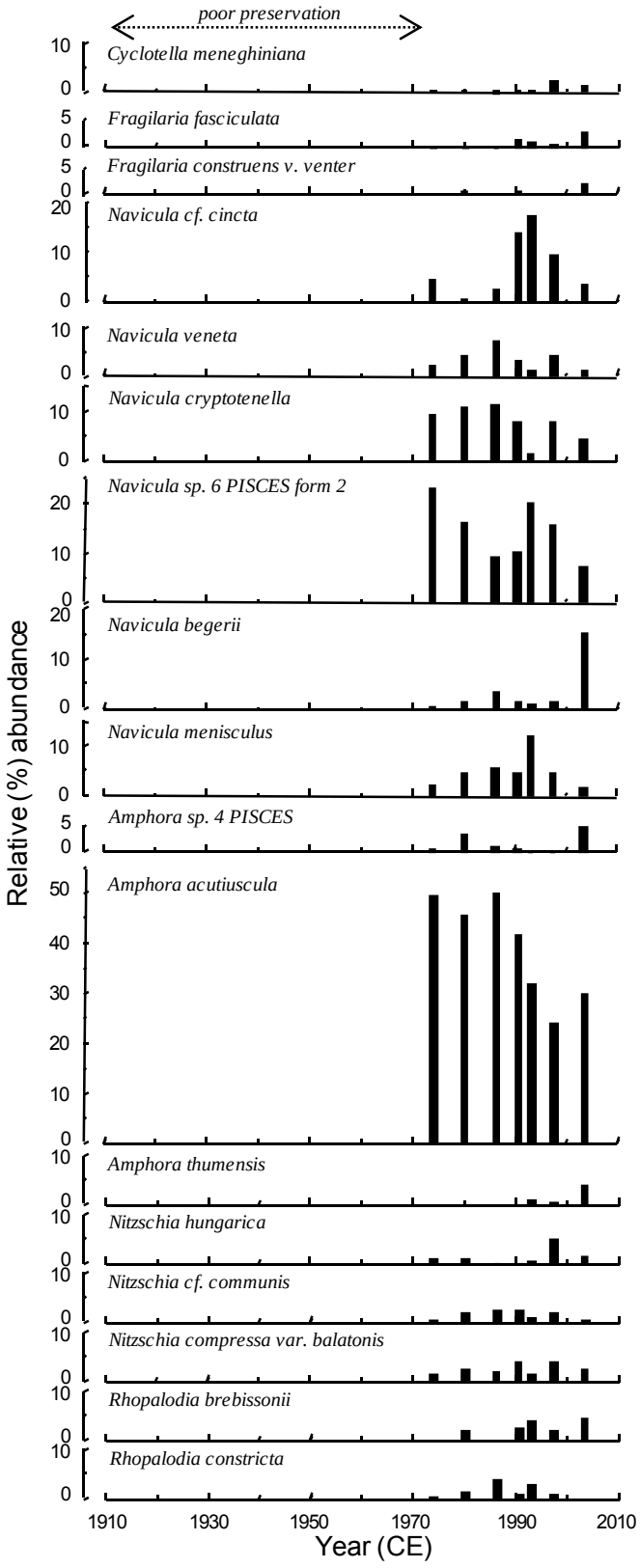


Fig. 7. Sedimentary diatoms, Gilbert Bay, Site 3



List of electronic appendices

Appendix 1. List of fossil diatoms recovered from sediments of Great Salt Lake, Utah.

Appendix 2. Specific activities (dpm g^{-1} dry mass) of ^{210}Pb and ^{137}Cs in sediments of Great Salt Lake, Utah.

Appendix 3. Fossil pigment concentrations, stable isotopes, fossil algal remains in sediments, and metal concentrations in sediments from Farmington Bay site 1.

Appendix 4. Fossil pigment concentrations, stable isotopes, fossil algal remains in sediments, and metal concentrations in sediments from Gilbert Bay site 3.

Appendix 5. Fossil pigment concentrations, stable isotopes, fossil algal remains in sediments, and metal concentrations in sediments from Gilbert Bay site 4.