A Report to the Utah Division of Water Quality

Comparative Analysis of Pollution in Farmington Bay and the Great Salt Lake, Utah

Aquatic Ecology Laboratory Class Project 2001
College of Natural Resources
Utah State University

Wayne A. Wurtsbaugh, Instructor[1]
Amy Marcarelli, Teaching Assistant

Students

Cameron Christison - Phytoplankton
Joel Moore - Brine Shrimp growth and survival
Donovan Gross - Farmington bay nutrient budget
Sophia Bates & Sara Kircher - Oxygen and hydrogen sulfide

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Reference:
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Summary

Farmington Bay covers 94 mi^2 (260 km^2) in the SW corner of the Great Salt Lake, and is essentially a separate lake because it is enclosed by Antelope Island and a causeway leading to the island from the mainland. The bay has received wastes from the adjoining Salt Lake City metropolitan area for decades. Because of water quality concerns for Farmington Bay, the Aquatic Ecology Laboratory class at Utah State University studied the bay and a nearby control site (Bridger Bay) in the Great Salt Lake during the fall of 2001. Field sampling and laboratory experiments, as well as other data sources, demonstrated the bay is severely eutrophic and is one of the most polluted water bodies in the state of Utah. A preliminary nutrient loading estimate for the bay indicates that total phosphorus coming into the system is 8-times higher than necessary for the bay to be classed as eutrophic. Sewage treatment plants discharging directly to the bay contribute approximately 50% of the nutrients. Metrics of eutrophication (chlorophyll, Secchi depth and total phosphorus) all indicated that the bay was hypereutrophic and the combined Trophic State Index was 91, higher than any other lake or reservoir in the state. Oxygen was supersaturated in the surface waters of Farmington Bay during the day, but the bottom water was anoxic. During the night, nearly the entire water column became anoxic due to respiratory demand of the biota. The anoxic conditions allowed high concentrations of foul-smelling hydrogen sulfide to be produced. Brine shrimp were not abundant in Farmington Bay and the community was dominated by rotifers. In contrast, water quality in Bridger Bay located on the main lake, was good and brine shrimp were abundant there. Our results, although restricted in scope, corroborate existing monitoring data from this bay.

Water quality characteristics in Farmington Bay do not meet those mandated for the protection of aquatic life. Odor problems from the bay likely impact more people than are affected by any other polluted water body in the state. The impact of eutrophication and anoxia on the biota in Farmington Bay may also be substantial, although inadequate data exists to determine these impacts. There are substantial technical challenges to be overcome if water quality in the bay is to be improved to meet its designated use. However, before these technical issues can be solved, the responsible agencies will need to address the problem, and begin studies that may eventually lead to a solution to this serious water quality issue.

Introduction

The Great Salt Lake of Utah lies next to greater metropolitan Salt Lake City with a population of over one million people. The lake is a tremendous recreational asset, supports a diverse and abundant bird community, and produces commercially important brine shrimp cysts for the world's aquaculture industry. The lake also receives a large portion of wastes from the adjoining community. Much of the city's wastes flow into Farmington Bay at the SE corner of the lake. This "bay" is more like a lake, as Antelope Island to the west, and a causeway joining the island to the mainland greatly restrict mixing of its waters with the much larger Great Salt Lake. The heavy waste load flowing into the bay has been a concern for decades. In 1972 Coburn and Eckhoff (1972) wrote "disregard for the...water quality of Farmington Bay might lead to...a tremendously large mismanaged waste lagoon, upwind from metropolitan Salt Lake City." They also commented that if anaerobic conditions developed in Farmington Bay, the potential for odor problems was at hand.

The objective of our class project was to compare water quality conditions in Farmington Bay with those of a nearby site in the Great Salt Lake. On October 4th and 5th we sampled limnological conditions at these two sites. Additionally, students conducted individual or group projects of their own design. Donovan Gross analyzed nutrient concentrations at the two sites and in tributaries and used data of the USGS, Utah Division of Water Quality and EPA to construct the first nutrient budget for Farmington Bay. Sara Kircher and Sophia Bates measured diel changes in oxygen at the two sites, and experimentally related these results to the production of odor-causing hydrogen sulfide gas. Cameron Christison and Joel Moore conducted an experiment to test how mixing the nutrient-laden Farmington Bay water with Great Salt Lake water would influence phytoplankton growth and species composition, and how this, in turn, would effect the growth, survival and egg production of brine shrimp. The results of the sampling and experiments, albeit limited to a short-term class project, indicate that Farmington Bay is indeed severely eutrophic, causing the production of noxious hydrogen sulfide gas and likely limiting the production of brine shrimp. In contrast, the site in the Great Salt Lake had good water quality and abundant populations of brine shrimp.

Methods

Field Sampling--We sampled at two sites: Farmington Bay, and Bridger Bay in the Great Salt Lake. Both bays are located near the Antelope Island Marina (Fig. 1). The Farmington Bay site was located approximately 1 km SE of the causeway bridge at the NE end of the island (12T 0396982, UTM 4546797). The Bridger Bay site was located off the NW tip of the island (12T 0393509, UTM 4544598) and was also approximately 1 km from shore. Both stations were in located where the depth was ca. 2 m. Weather prior to, and on the first day of sampling (Oct. 4) was warm and there were no winds. Strong winds began at 2000 hr and blew through much of the night and the following morning.

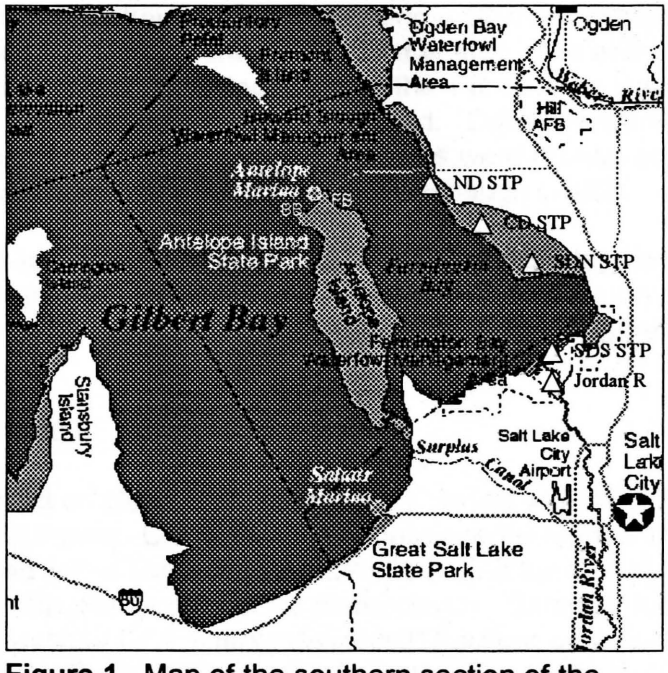

Figure 1. Map of the southern section of the Great Salt Lake showing the sampling sites in Farmington Bay (FB) and Bridger Bay (BB). Triangles show the locations of sewage treatment plants and the Jordan River sampling sites.

At each station we measured vertical profiles of temperature and dissolved oxygen with a Yellow Springs Instrument (YSI) probe that was calibrated for zero oxygen with saturated sulfite solution, and for air saturation. The profiles were done between 1600 and 1800 on 4 October and again before sunrise on 5 October. At each site recording YSI sondes were deployed from buoys at a depth of 1 m and recorded temperature, oxygen and conductivity at 5-minute intervals from the late afternoon until early morning. Light profiles were measured with a LiCor radiometer and a 4π sensor that measured photosynthetic active radiation (PAR). Secchi disk transparency measurements were made with a 20-cm black and white disk. Samples for chemical analysis were collected with a 10-L Kemmerer bottle. The bottle is 0.5-m long, and thus was not ideally suited for the fine-scale stratification encountered in Farmington Bay.

Phytoplankton from these samples were preserved in 5% formalin. In the laboratory they were identified and enumerated using Utermohl settling chambers and an inverted microscope at 400X. 10-20 mL aliquots (depending on expected algae concentrations) were settled in the chambers for at least 24 hours. Algae were grouped into four broad taxonomic categories: (1) Large chlorophytes – mostly *Dunaliella salina, Dunaliella viridis,* and *Carteria sp.* (2) Small chlorophytes – primarily *Oocystis parva.* (3) Filamentous cyanophytes – most likely *Nodularia spumigena*, and; (4) Diatoms – including representatives of the genera *Navicula, Amorpha, and Nitzechia*. At least 100 representatives from each category were counted or, alternatively if densities were low, at least 30 fields were examined. The lengths and widths of 10 individuals from each category were measured in order to calculate average biovolume per individual. Measurements were made only once and applied across all the samples and experimental treatments (see below). Biovolumes were estimated with the geometric volume technique outlined in Wetzel and Likens (2000).

Zooplankton were collected with two top-to-bottom vertical hauls at each site using a 12-cm diameter net with 80-µm netting, and preserved with 5% formalin. They were enumerated at 40X. The entire samples were counted for macrozooplankton (brine shrimp, cladocera). In the Farmington Bay samples, rotifers were very abundant and they were consequently subsampled. Densities were calculated assuming a net efficiency of 50%. Lengths of up to 10 of each species were measured with eyepiece micrometers, and biomasses calculated with the equations of Wurtsbaugh (1992).

Chemistry samples from each depth were stored in acid-washed high-density polyethylene bottles that were repeatedly rinsed with lake and sample water prior to storing the final sample. In the laboratory, density was measured with a hydrometer and salinity was calculated from these values using the equation of Wurtsbaugh and Berry (1990):

$$\text{Salinity (g/L)} = 1430.6\ (\text{density}) - 1430.8$$

Samples (25 ml) for chlorophyll analysis were filtered on glass fiber filters (GF/F), rinsed with ca. 10 ml of deionized water to remove the salt water, and frozen. Chlorophyll was subsequently extracted for 24 hours in 6 ml of 100% methanol and analyzed with a Turner 10AU fluorometer and the method of Welschmeyer (1994) that does not require acidification to correct for phaeophytin. Samples for nutrient analysis were either filtered through acid-washed GF/F filters ("dissolved" fraction) or stored for two weeks as whole water samples (Total N, Total P) at -20°C, but did not freeze due to the high salt content. Soluble reactive phosphorus was measured spectrophotometrically with the acid-molybdate method (APHA 1992) in 10-cm cells. Samples for TN and TP analysis were simultaneously digested using the methodology of Nydahl (1978). Phosphorus was then analyzed with the acid-molybdate method, and nitrogen with the second derivative method of Ferree and Shannon (2001). Arsenic interference with the acid-molybdate method (Linge 2001; Linge and Oldham 2001) was not accounted for and may have led to an overestimate of SRP and TP.

Nutrient Loading to Farmington Bay–The objectives of this aspect of the research project were to calculate a total phosphorus budget, to estimate areal loading, and to quantify the primary sources of phosphorus for Farmington Bay. Although phytoplankton in the bay may be more limited by nitrogen than by phosphorus, we used the latter in our analysis because total nitrogen data is not collected by the state, and because phosphorus budgets are more commonly used in euthrophication analyses. Data used in the total phosphorus (TP) budget were obtained from the EPA *Storet* database located at www.epa.gov/storet. We analyzed inflows from the Jordan River (Div. Water Quality ID #499182), and from the four wastewater treatment plants (WTP) that discharge their secondarily-treated effluents directly to the bay (South Davis South Waste Water Treatment Plant #499181; South Davis North #499078; Central Davis #499027; North Davis #49907). The location of the collection sites is shown in Figure 1. Farmington Bay's area and volume were determined using a hypsographic curve (Fig. 2) constructed from a morphometric map of Farmington Bay (Great Salt Lake Yacht Club).

Total phosphorus concentrations (reported as P) and discharge measurements from discrete time intervals were obtained for 1 Aug 1999 to 31 July 2000. The sample dates used were 24 Aug 99, 30 Nov 99, 11 Jan 00, 29 Feb 00, 28 Mar 00, 18 Apr 00, 4 May 00, 23 May 00, and 6 Jun 00. Nutrient concentrations from discrete sampling dates were averaged to obtain a mean value over that interval. These concentrations were then multiplied by mean discharge values from the same sampling dates and multiplied by the number of days in the interval to arrive at an estimated total nutrient load in that interval. Nutrient loads for each interval were then summed to arrive at an estimated annual nutrient load (kg P y^{-1}). The process was repeated for each inflow used.

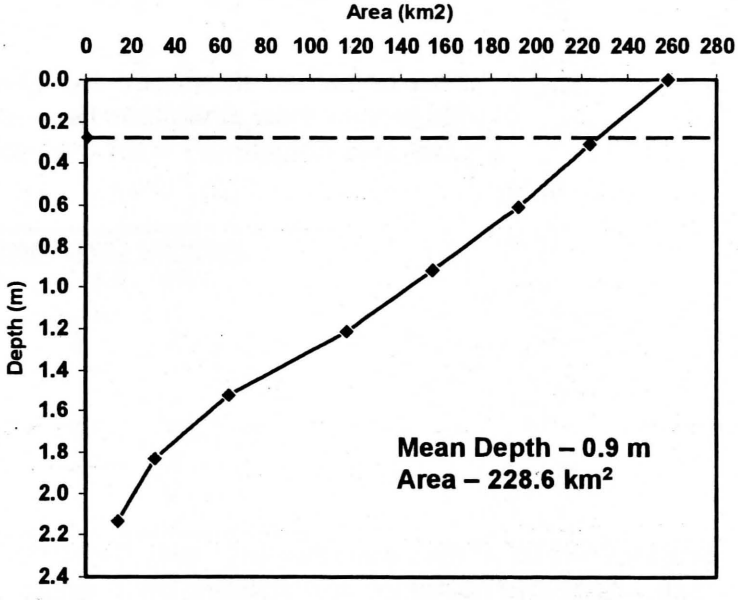

Fig. 2. Hypsographic curve of Farmington Bay constructed from Great Salt Lake Yacht Club map, 1st ed., 1992, 1927 North America datum. Dashed line shows lake elevation (1279.9 m) during the nutrient loading water year that was analyzed.

Annual TP load was then divided by the surface area of Farmington Bay to arrive at annual areal TP loading (mg P*m^{-2} y^{-1}). Areal loading was used in an equation adapted from Vollenweider (1969) by Cooke et al. (1993): $TP = L/z(p+q)$; where L is areal TP loading (kg P*m^{-2} y^{-1}) and z is mean depth (m, 0.9 m); *p* is hydraulic flushing rate (0.77 *y^{-1}) at a lake elevation 4200 ft (1280.2 m) (Austin 1993), and *q* is the sedimentation rate coefficient for P. Vollenweider's (1976) estimate for *q* (10/z, where 10 has units of m/y) was used (Cooke et al. 1993). Occasionally, a parameter (TP concentration or discharge) was missing for a sample date. In these cases, values from the sample date immediately preceding and immediately following were averaged to obtain a value for that date. Due to the size of the data set, the exact values used are not shown here but rather are available from the *Storet* database or from the authors upon request.

Hydrogen sulfide and oxygen analysis sampling and experiment–Hydrogen sulfide produces the rotten-egg smell that often influences Salt Lake communities. To determine what factors contribute to hydrogen sulfide production, we used a 3-way factorial experiment with the following treatments: (1) presence or absence of sediments; (2) station location, and; (3) incubation in light or dark. The

latter simulated what could happen in shallow (light) or deep sediments (dark). Two replicates of each treatment were used. Samples consisting of water or water and sediment were taken from Farmington Bay (12 samples) and Bridger Bay (12 samples). The samples were placed in 470-ml acid-washed canning jars. Water samples from Farmington Bay and Bridger Bay were taken from 1-m depth. In Bridger Bay, sediments and accompanying water were collected by hand by scooping them into the jars and capping them underwater to minimize oxidation of the hydrogen sulfide. The sediments from Bridger Bay were primarily oolitic sands, with some algal material. The amount of sediment in each Bridger Bay sample was approximately 100 ml. Sediments and water from Farmington Bay were sampled using an Eckman dredge and a bucket. The dredge and sample were placed in a bucket underwater and brought to the surface. A canning jar was filled with the overlying water from the dredge. A 60-ml syringe with the end cut off was used to collect 50 ml of the soft, fine sediment and deposit it in the water-filled jar. Plastic wrap was placed over all the sampled jars immediately after sampling to prevent oxygen from entering or escaping, and to protect the lids from rusting due to high salinity. Each sample was placed in a cooler with ice for transportation to the laboratory. They were held overnight in the cooler and the next day (Day 0) four samples from each site were analyzed for oxygen with a YSI probe. Hydrogen sulfide was measured by titration using the iodometric method (APHA 1992).

Table 1. Experimental treatments. All treatments contained water from the respective sites. On day 0, all treatments were without light due to being transported in a cooler. FB = Farmington Bay, BB = Bridger Bay.

Day 0	Treatments	With sediment		W/O sediment	
		FB	BB	FB	BB
	W/ Light	-	-	-	-
	W/O Light	2	2	2	2
Day 5	Treatments				
		FB	BB	FB	BB
	W/ Light	2	2	2	2
	W/O Light	2	2	2	2

The remaining samples were incubated at ca. 22°C for 5 days. The jars were separated into light and dark treatments to determine the influence of photosynthetic production of oxygen on hydrogen sulfide production. Dark treatments were covered using a cardboard box whereas the light treatments were exposed to intensities of 150 μE $cm^{-2}sec^{-1}$ on a 18:6 light:dark cycle. After five days we again measured oxygen and hydrogen sulfide using the methods described previously. These measurements were done at 1400-1600 hrs, after the light samples had been in the light for ca. 7 hours. A three-way ANOVA was conducted to test whether oxygen and H_2S concentrations were significantly influenced by station, the presence or absence of sediments, and by the light treatment. The experimental design is summarized in Table 1.

Effects of dilution of Farmington Bay water with Great Salt Lake water–A potential remedy for the eutrophication problems in Farmington Bay would be to increase mixing between the bay and the much larger southern section (Gilbert Bay) of the main lake, or to discharge sewage effluents directly to the main lake. This could potentially dilute the high nutrient levels of the Farmington Bay, but it would increase nutrients (and possibly other pollutants) in the main lake, and thus potentially harm brine shrimp or other organisms. To assess how diversion of Farmington Bay water into the main lake would influence the plankton, we experimentally mixed different proportions of water from the two habitats in small microcosms, and measured the responses of phytoplankton and brine shrimp for 15 days.

Because both nutrients and salinity levels could influence the plankton, we prepared treatments where the salinity was allowed to vary in relation to the different proportions of water that were mixed from Farmington Bay (FB) and from the main lake (Bridger Bay, BB). In other treatments we added salts to maintain a constant salinity characteristic of the main lake 115 g/L). The experimental design was as follows:

Mix proportion (%)		Salinity (# of replicates)	
FB	BB	Constant	Varying
100	0	2	2
75	25	2	2
50	50	2	2
10	90	2	2
0	100	2	2

Water for the experiment was collected at 1-m at each of the two sites and brought to the laboratory in polyethylene Cubitainers®. Twenty, 1-L jars were filled with 0.85 L of various proportions of Farmington Bay (FBW) and Bridger Bay water (BBW). During mixing, all macrozooplankton were removed from the microcosms using a 153-µm screen. The terminology used to distinguish the treatments is found in Table 2. In the Constant Salinity treatments (CS) we added a 1:1 ratio of NaCl to *Instant Ocean*® salts (Aquarium Systems, Menton, Ohio) to 2 replicates of each mixture so that the resulting salinity would approximate that of 100% Bridger Bay water. Two different sets of jars were used for the Constant Salinity and the Variable Salinity treatments, and this may have influenced the results (see below). At the end of the experiment water density was measured and salinity calculated as discussed above to ensure that there were no significant changes.

Table 2. List of treatments and terminology. CS1 refers to constant salinity / replicate 1. 100/0 mean treatment is 100%FBW and 0%BBW. Salinities shown were measured at the end of the experiment.

Jar Number	Terminology	Salinity (g/L)	% Farmington Bay	% Bridger Bay
1	CS1 100/0	113	100	0
2	CS2 100/0	113	100	0
3	CS1 75/25	113	75	25
4	CS2 75/25	113	75	25
5	CS1 50/50	113	50	50
6	CS2 50/50	111	50	50
7	CS1 10/90	113	10	90
8	CS2 10/90	111	10	90
9	CS1 0/100	113	0	100
10	CS2 0/100	111	0	100
11	VS1 100/0	88	100	0
12	VS2 100/0	90	100	0
13	VS1 75/25	96	75	25
14	VS2 75/25	96	75	25
15	VS1 50/50	100	50	50
16	VS2 50/50	100	50	50
17	VS1 10/90	109	10	90
18	VS2 10/90	110	10	90
19	VS1 0/100	111	0	100
20	VS2 0/100	111	0	100

After the microcosm mixtures were prepared, 15, 2-day old *Artemia* nauplii hatched from cysts were randomly counted into 20 treatment jars. The glass jars were incubated with an 18:6 light:dark cycle, light intensities of 150 $\mu E\ cm^{-2}\ sec^{-1}$, and a temperature of 20°C. The treatments were swirled to mix the phytoplankton and randomized 1-3 times a day for the next 15 days. The lids were removed from the jars once a day during the experiment for 2-5 minutes to allow gas exchange. To determine if the treatments were undergoing anoxia during the dark hours we altered the lighting regime the day before the experiment was concluded. The lights were turned back off after they had been on for 30 minutes of the day cycle and left off for 3 hours befre dissolved oxygen was measured.

On days 0, 5, 10, and 15, 20-ml samples were removed from each microcosm for chlorophyll analysis, filtered, frozen and measured as described above. At the end of the experiment, 100 ml samples were taken from each microcosm and preserved in 3% formalin for the identification and enumeration of algae taxa. Counting and biomass measurements were performed as discussed above.

The number of living *Artemia* in each jar were observed and counted on days 5, 10, 15 of the experiment. On day five counting was difficult in treatments with a high proportion of Farmington Bay water, as the small nauplii were difficult to see in the turbid water. At the end of the experiment the living *Artemia* were filtered out and preserved in formalin for subsequent counting and measuring (formalin was mistakenly not added to the 75%FBW-constant salinity treatment). *Artemia* were measured from the tip of the head to the end of the caudal furca using a micrometer (Basbug and Demirkalp 1997). The number of gravid females was recorded and eggs were counted by dissecting the female egg sacs. We determined individual dry weight of the *Artemia* by using the following length-dry weight relationship of Reeve (1963):

$$\text{Dry Weight (g)} = 0.9 * \text{Length (mm)}^{3.02}$$

Results

Field Sampling

Salinity levels measured during the afternoon profiles indicated nearly uniform conditions throughout the water columns, with a mean salinity of 115 g/L in Bridger Bay and 87 g/L in Farmington Bay. The water in Bridger Bay was moderately clear, with a Secchi depth of 2.0 meters. The light extinction coefficient was 0.62 and 25% of surface light reached the bottom at 2 meters. The projected depth of the photic zone (1% light level) was 7.4 m. In contrast, the water in Farmington Bay was extremely green and the Secchi depth was only 0.1 m. The light extinction coefficient was 6.17 and the depth of the photic zone was estimated to be 0.7 m.

During the afternoon the lake was calm and there was minimal thermal stratification in Bridger Bay. Oxygen levels increased slightly through the water column and were somewhat above saturation (Fig. 3a). By dawn, oxygen levels had dropped slightly, and were uniform through the water column. The heavy wave action moved our boat from the original station to a shallower site so we could not measure oxygen and temperature below 1 m. However, night winds were very strong (see below) and wave height in the morning was estimated at 0.4 m, so it is unlikely the lake was stratified in the littoral area.

In contrast to Bridger Bay, in Farmington Bay there were marked vertical changes in oxygen through the water column and very strong diel changes (Fig. 3b). During the day the surface water was supersaturated with oxygen, with concentrations reaching 16.5 mg/L. Below the photic zone (0.75 m) concentrations dropped rapidly, decreasing to 1.8 mg/L at 2 m. At dawn, most of the water column was anoxic. However, between 2 and 2.5 m concentrations increased to >2 mg/L.

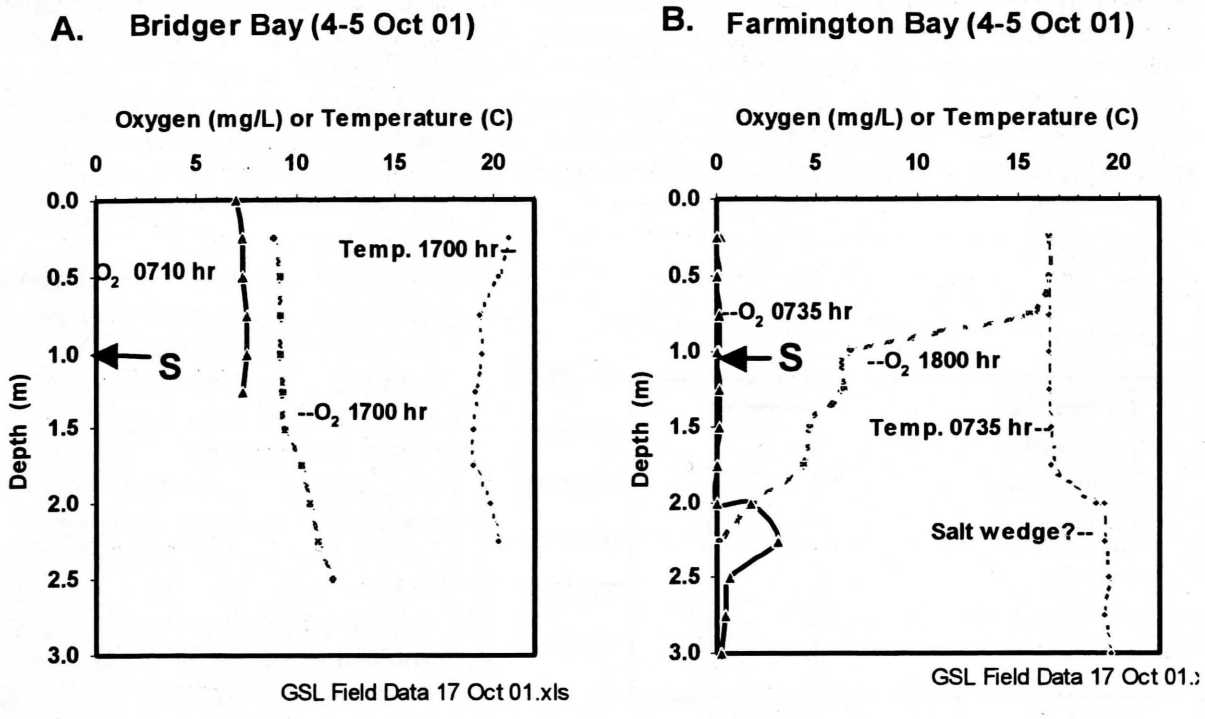

Figure 4. Oxygen profiles in Bridger Bay (A) and Farmington Bay (B) taken during the late afternoon and at dawn on October 4th and 5th. Note that most of the water column in Farmington Bay became anoxic overnight. The bulge in oxygen in the deeper water of Farmington Bay may have been associated with a salt wedge that caused inverse thermal stratification. "S" indicates the depth at which the oxygen-recording sondes were deployed.

Temperatures were warmer in the deeper water than in the surface, suggesting that during the morning profile our station was located over a salt wedge from the main lake that likely entered through the breach in the causeway. Unfortunately, vertical profiles of salinity were not taken during the morning profile to confirm the presence of the wedge. It is unclear whether the nearly complete loss of oxygen occurs nightly in Farmington Bay, or whether this phenomenon occurs only when high winds agitate the sediments, causing increased biochemical oxygen demand. Even if anoxia occurs only during wind events, it points to a severe euthrophication problem in the bay.

The diel changes in oxygen in the two bays was also obvious from data collected with the sondes deployed at 1 m. In Bridger Bay oxygen concentrations varied relatively little between the late afternoon and dawn, with a total range of 7.8 to 5.5 mg/L (Fig. 4). In Farmington Bay, the sonde was located in the region of the water column where

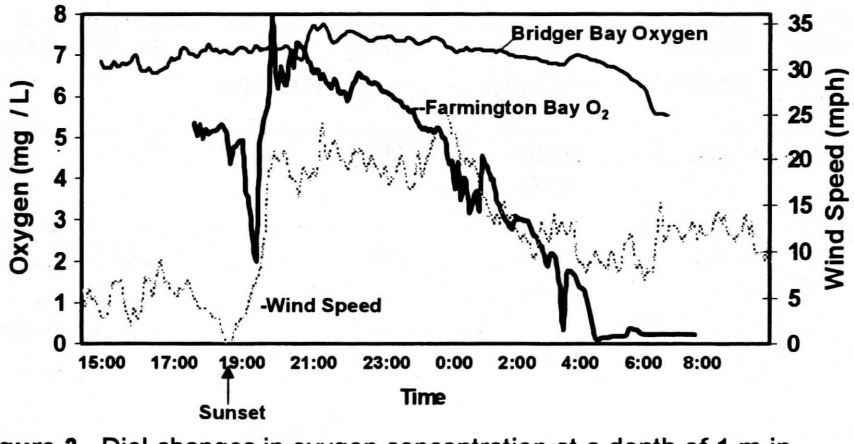

Figure 3. Diel changes in oxygen concentration at a depth of 1 m in Farmington Bay (dark line) and in Bridger Bay of the Great Salt Lake on 4-5 October 2001. Wind speed on Antelope Island is also shown (wind data compliments of Utah Department of Air Quality (http://www.met.utah.edu/)

oxygen levels decreased rapidly (1m). At this site, oxygen concentrations were erratic, but nevertheless showed the overnight loss of nearly all the oxygen in the epilimnion. After sunset, oxygen levels dropped to near 2 mg/L, but then increased rapidly as the northerly winds increased, possibly due to the water layers shifting in a seiche-like manner. Subsequently, oxygen concentrations declined throughout the night, reaching 0.1 mg/L by 0515 in the morning. The sudden reversals in the oxygen trend during the night were related to changes in wind (Fig. 4) and to changes in conductivity (not shown), suggesting that different water masses or strata were being mixed together.

Chlorophyll levels were far higher in Farmington Bay than in Bridger Bay (Table 3). In Bridger Bay there was no vertical stratification of chlorophyll and mean levels through the water column were 7 µg/L. In Farmington Bay mean water column chlorophyll *a* was 141 µg/L. There was some stratification, with levels reaching 180 µg/L at 1.5 m. The biovolumes of the algal communities were consistent with the differences in chlorophyll concentrations (Fig. 5). The average biovolume in Farmington Bay was 74-times higher than in Bridger Bay. In Farmington Bay the community was primarily dominated by green algae (98%) of the genera *Dunaliella* and *Carteria*, whereas the community in Bridger Bay was composed of 89% diatoms, primarily in the genera *Navicula*, *Amorpha*, and *Nitzechia*.

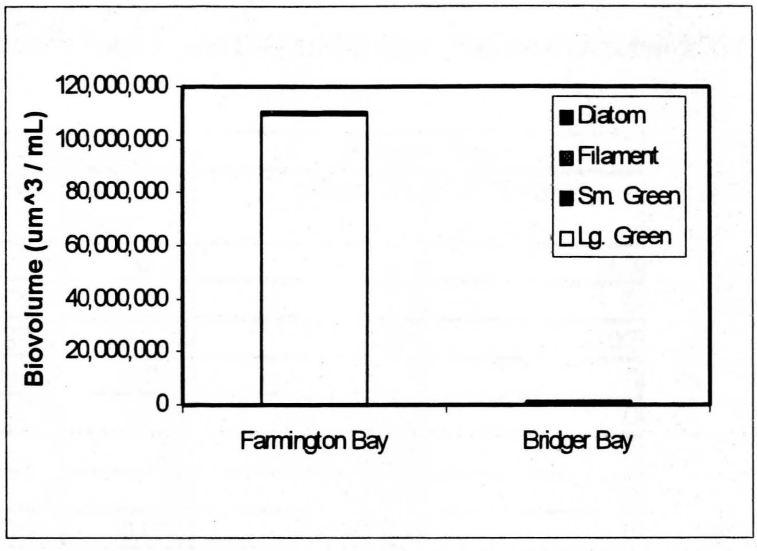

Figure 5. Biovolumes of phytoplankton in Farmington and Bridger Bays on 4 Oct. 2002

Table 3. Chlorophyll, total phosphorus (TP) and total nitrogen (TN) levels in Farmington Bay and the Great Salt Lake (Bridger Bay) on 4 October 2001. Values in parentheses show S.E. Criteria for classification as eutrophic or hypereutrophic (Wetzel 2001) and the trophic state indices of Carlson (TSI) for chlorophyll, TP and Secchi are also shown.

	Chlorophyll a µg / Liter	TSI(chl)	TP µg / Liter	TSI(tp)	TN µg / Liter	Secchi meters	TSI(sd)
Farmington Bay	141 (13)	79	890	102	5290 (220)	0.1	93
Great Salt Lake	7.0 (0.5)	50	320	87	2880 (130)	2.0	50
Eutrophic level	>14		>84		>1875		
Hypereutrophic level	>100		>750		-		

The zooplankton community at the two sites also differed markedly (Table 2). In Bridger Bay, the community was composed almost entirely of *Artemia*. In Farmington Bay the dominant taxa were rotifers with some cladocerans, primarily *Bosmina* sp., and brine shrimp were ten times less abundant than in Bridger Bay (Figure 6a). The samples yielded a single corixid in Farmington Bay and a single brine fly larvae in Bridger Bay. The biomass in each bay reflected the larger species, *Artemia* in Bridger Bay and *Bosmina* in Farmington Bay (Figure 6b).

Table 2. Estimated densities of zooplankton in Farmington Bay and Bridger Bay on 4 October 2001. Mean ± standard error of two replicates at each station.

	Farmington Bay		Bridger Bay	
	Mean #/L	Std. Error	Mean #/L	Std. Error
Brine shrimp adults & juv.	0.4	0.2	3.1	0.3
Brine shrimp nauplii	0.1	0.04	0.8	0
Rotifers	227.1	131.2	0.04	0.02
Harpactacoid copepods	0.02	0.02	0	0
Bosmina	10.4	10.4	0	0
Daphnia	0	0	0.04	0.02
Brine Fly larvae	0	0	0.04	0.02
Corixid	0.02	0.02	0	0

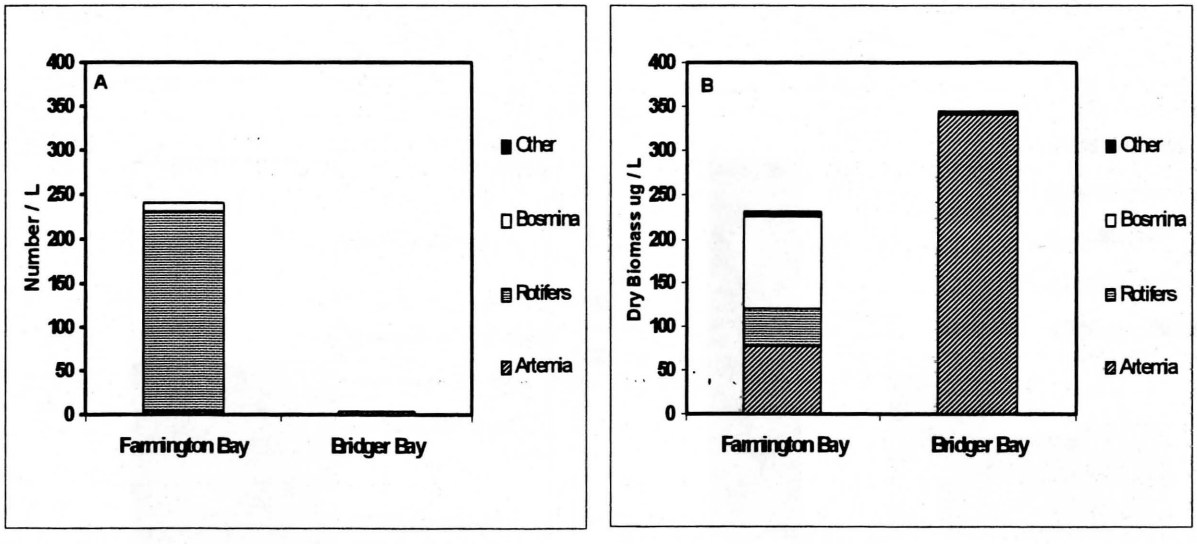

Figure 6. Densities (A) and biomasses (B) of zooplankton in Farmington and Bridger Bays on 4 October 2001.

Temporal analysis of phosphorus discharges indicated that during the dry seasons most of the loading entered Farmington Bay from sewage treatment plants that discharge directly into the bay (Fig. 7). During spring run off, however, loading increases markedly from the Jordan River. Analysis of the TP and discharge data resulted in a mean daily areal P loading of 5.5 mg $P*m^{-2}\ d^{-1}$ or 1.26 tonnes $P*d^{-1}$ entering Farmington Bay and the wetland complex. Annual loading was 157 mg $P*m^{-2}\ d^{-1}$. Of this, direct loading from the sewage treatment plants contributed 48% and the Jordan River contributed 52% (Fig. 8). The nitrogen to phosphorus ratio of Farmington Bay, the Jordan River and the sewage treatment plants, were well below the Redfield ratio (Fig. 9), indicating that the waters in this region may be nitrogen limited. However, the extremely high algal populations and/or inorganic turbidity restrict light penetration so much, it is likely that algal production in the bay is currently light-limited.

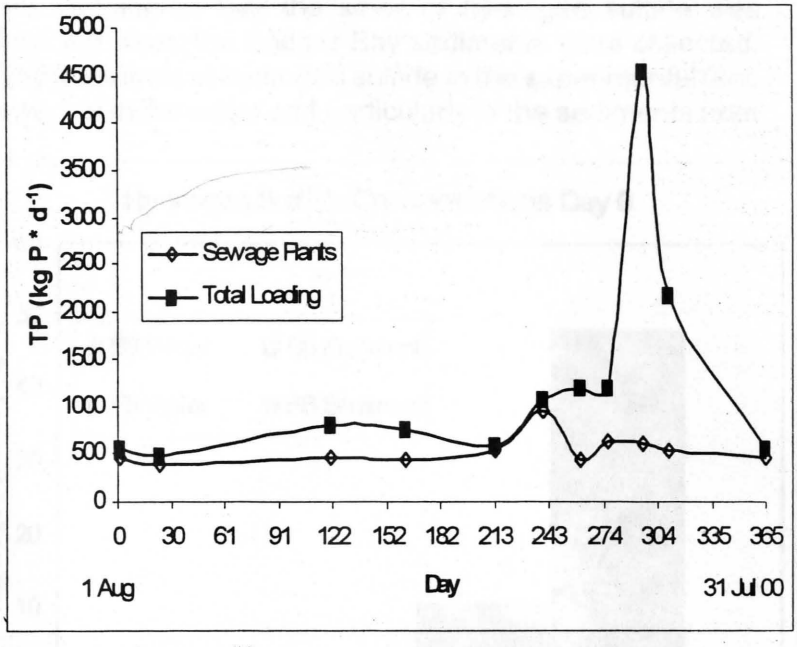

Figure 7. Seasonal changes in nutrient loading to Farmington Bay from sewage treatment plants discharging directlyto the bay and the total loading. The difference between the two lines is primarily the loading from the Jordan River. Sequential days from 1August are on the x-axis.

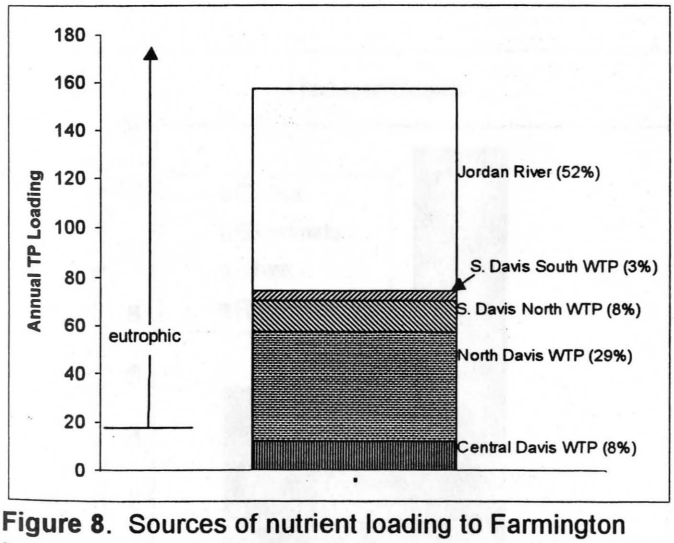

Figure 8. Sources of nutrient loading to Farmington Bay (mg m-2 d-1). Note that phosphorus loading is far greater than necessary to make the bay eutrophic.

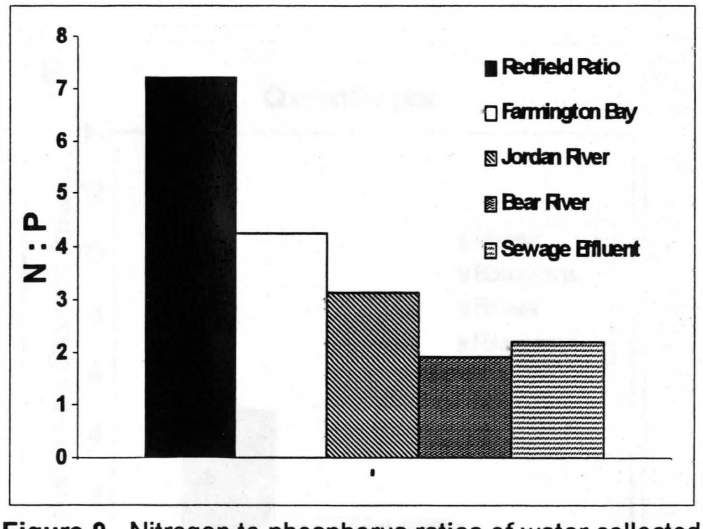

Figure 9. Nitrogen to phosphorus ratios of water collected from the Jordan River, Bear River and sewage treatment plants on 4 October 2001. Also shown is the Redfield ratio, the normal N:P ratio of plankton.

Donvan's FB nutrient loading

Using his Fig. 7 and stated load of 1.26 tonnes P/d is:
1,260 kg P/day entering FB and wetlands

=
459900000 g P/ year
2.01 g/m2/year

However, he also says "annual loading was 157 mg P/m2/d".
If his units are correct and we multiply by 365 d/year = 57,305.00 mg/m2/year
= 57 g/m2/year

Something is amiss!
My guess is that his calculations are correct, but that he isn't showing loading, but rather the estimated P concentration (mg/m3).

When dredge samples were taken from Farmington Bay the smell of hydrogen sulfide was overwhelming, whereas no smell was detected when the Bridger Bay sediments were collected. These differences were reflected in the Day 0 analysis of hydrogen sulfide in the experimental jars. Hydrogen sulfide concentrations were far higher in the water and particularly in the sediments from Farmington Bay than from Bridger Bay (Fig. 10).

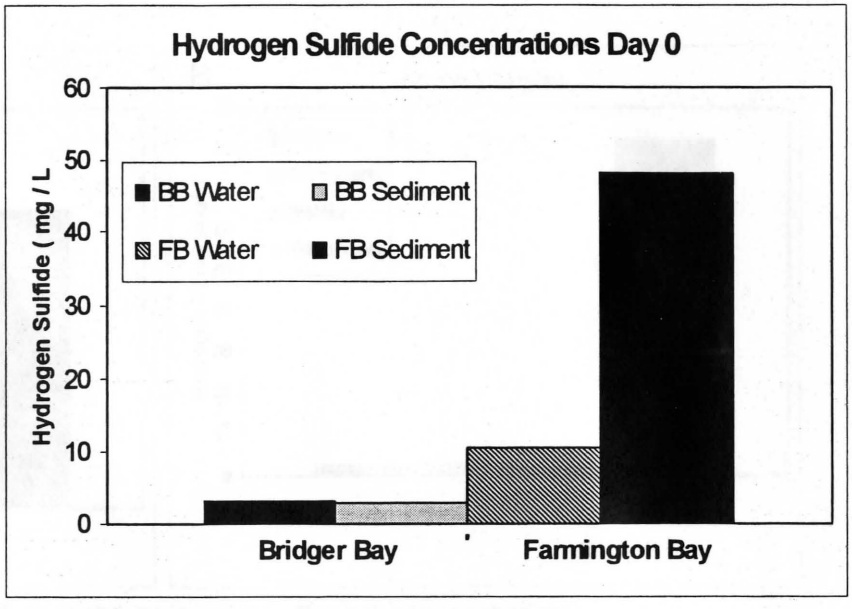

Figure 10. Hydrogen sulfide concentrations in the microcosms at the start of the laboratory experiment (18 h after the samples were collected from Bridger Bay (BB) and from Farmington Bay (FB).

After five days, oxygen concentrations in the different treatments differed markedly (Fig.11), and the effects of the presence or absence of sediments and light were both highly significant ($p < 0.01$). Oxygen concentrations were moderate in the uncovered Bridger Bay samples that were exposed to light. In the Farmington Bay treatments, the water-alone treatment was supersaturated with oxygen, but in the sediment treatment the mean oxygen level was < 0.5 mg L^{-1}. In the covered samples, oxygen was only high in the Bridger Bay treatment without sediments. The Farmington Bay water treatment had ca. 1 mg O_2 L^{-1}, and the water plus sediments had < 0.6 O_2 L^{-1}.

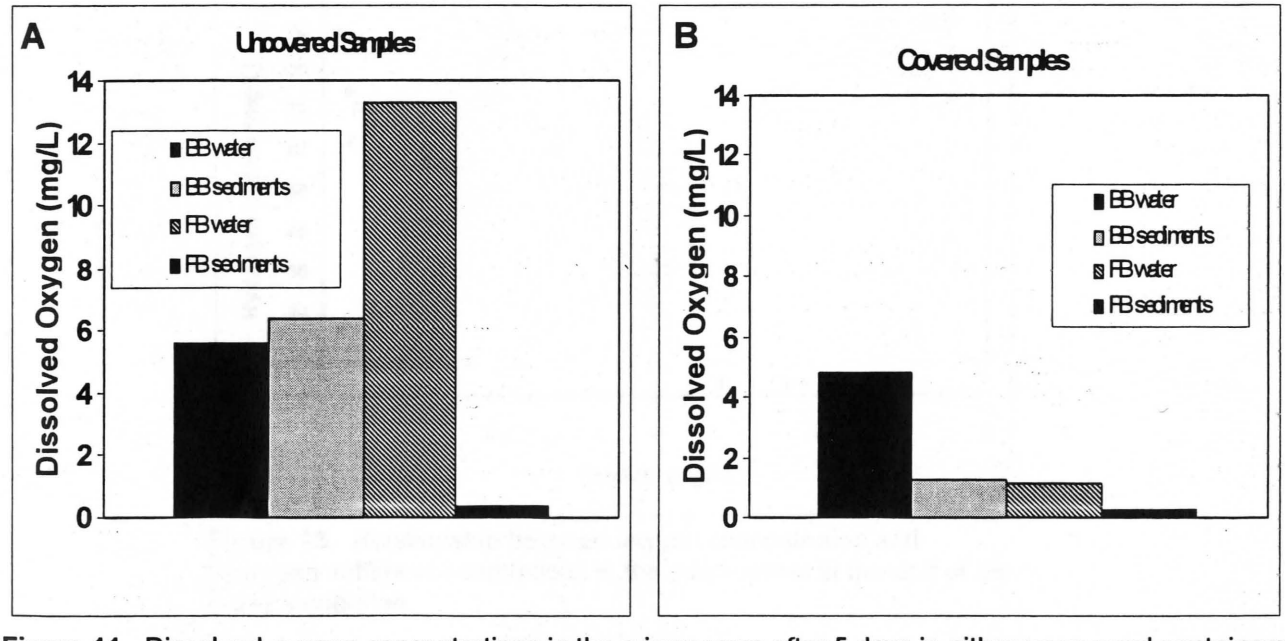

Figure 11. Dissolved oxygen concentrations in the microcosms after 5 days in either uncovered containers receiving light (A) or in covered containers.

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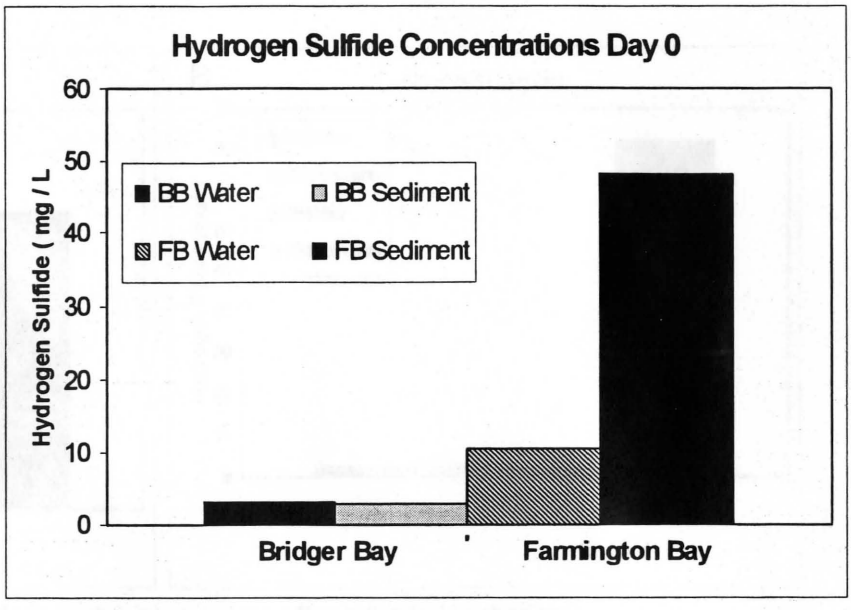

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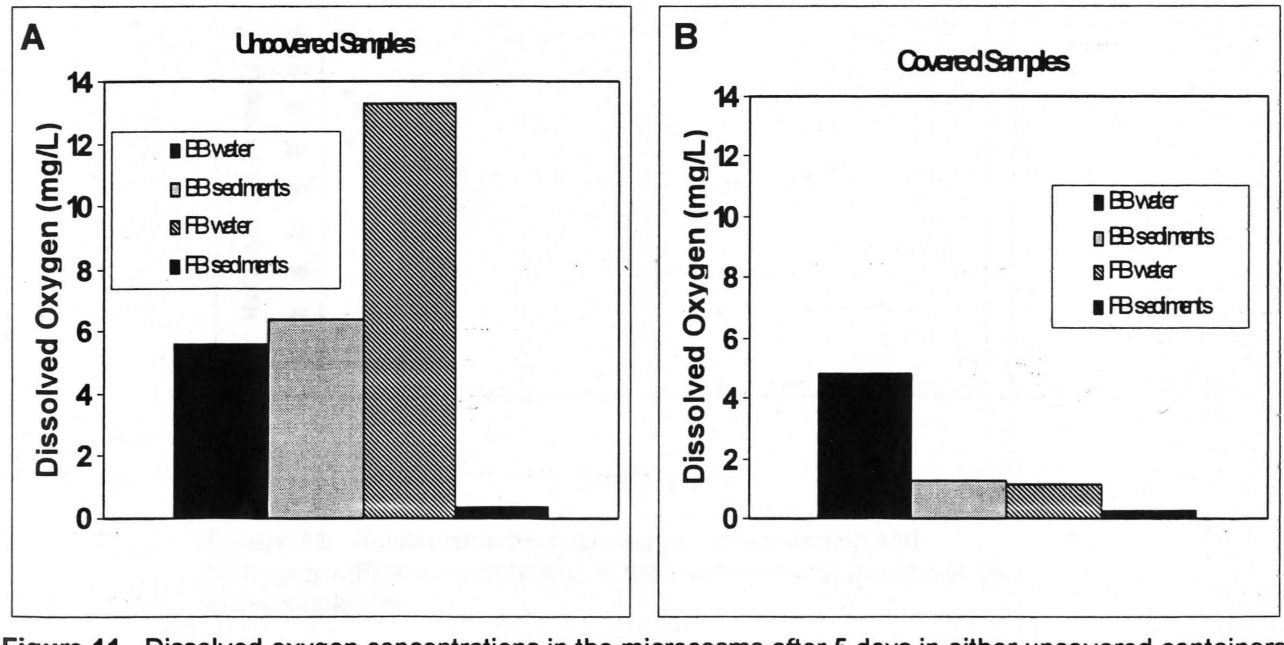

Figure 11. Dissolved oxygen concentrations in the microcosms after 5 days in either uncovered containers receiving light (A) or in covered containers.

At the end of the experiment, the only treatments with large amounts of hydrogen sulfide were those with Farmington Bay sediments, with concentrations reaching 82 mg L^{-1} (Fig. 12). Hydrogen sulfide concentrations in the different treatments were significantly different ($p < 0.001$) between sites ($p < 0.001$) and between substrates (water vs water+sediments) ($p < 0.001$), but the effect of light was not significant ($p = 0.12$). As expected, the amount of hydrogen sulfide present in the microcosms was clearly related to oxygen levels (Fig. 13), since hydrogen sulfide is only produced under anoxic conditions.

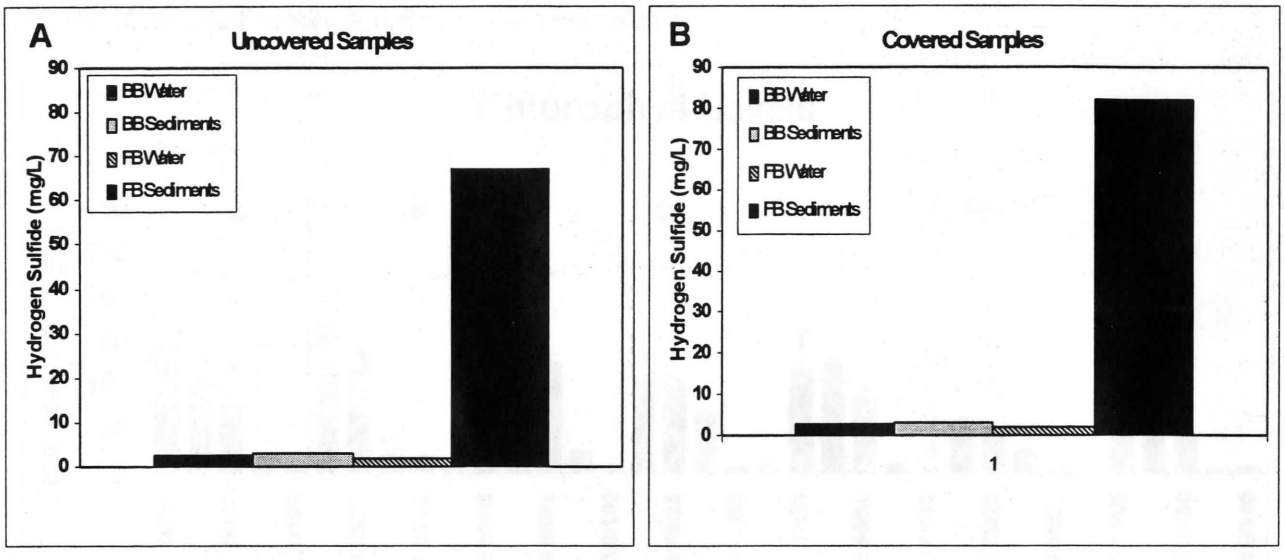

Figure 12. Hydrogen sulfide concentrations in the microcosms after a 5-day incubation.

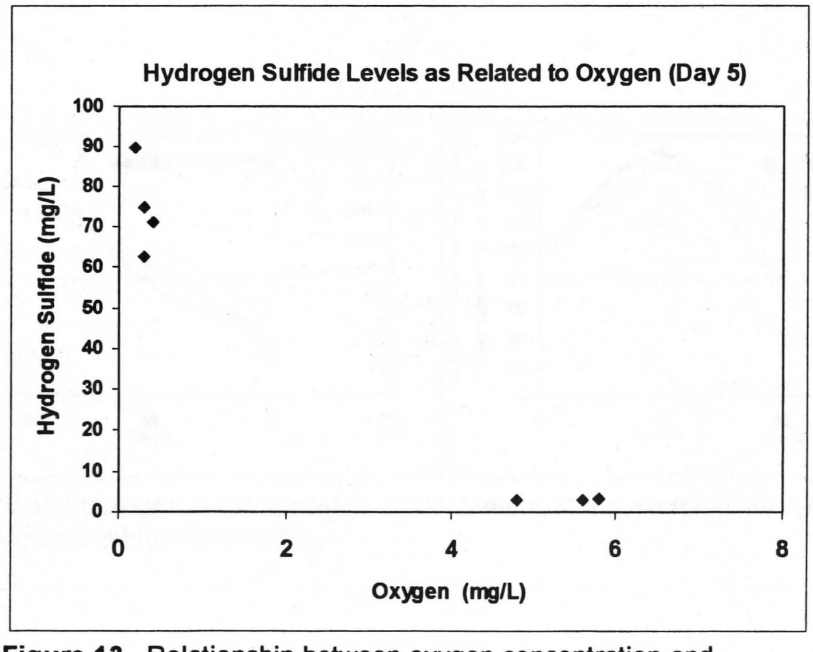

Figure 13. Relationship between oxygen concentration and hydrogen sulfide concentrations in the microcosms at the end of the 5-day incubation.

Effects of diluting Farmington Bay water into the Great Salt Lake: Plankton responses

The initial chlorophyll levels in the microcosms simply reflected the relative mix of the extremely high-chlorophyll water from Farmington Bay, and the moderate chlorophyll levels in Bridger Bay (Fig. 14). In many treatments, particularly those receiving significant amounts of Farmington Bay water, the chlorophyll levels initially increased 25-165% by day 5, and then declined between day 5 and 15. Chlorophyll concentrations reached 240 µg L^{-1} in the 100% Farmington Bay, Constant Salinity treatment on day 5. The initial increase was muted, or even absent, in those treatments composed of only 10 or 0% Farmington Bay water (Fig. 15).

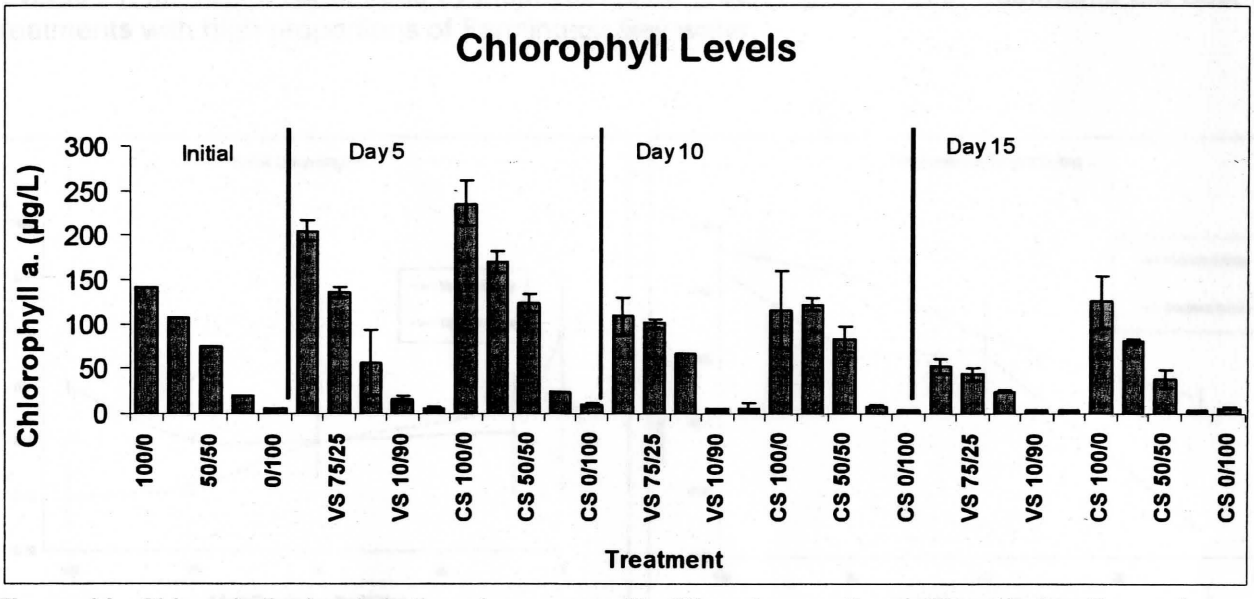

Figure 14. Chlorophyll a levels in the microcosms with different proportions of Farmington Bay and Bridger Bay water. 100/0, for example, describes the proportion of Farmington Bay to Bridger Bay water. VS = Variable Salinity; CS = Constant Salinity.

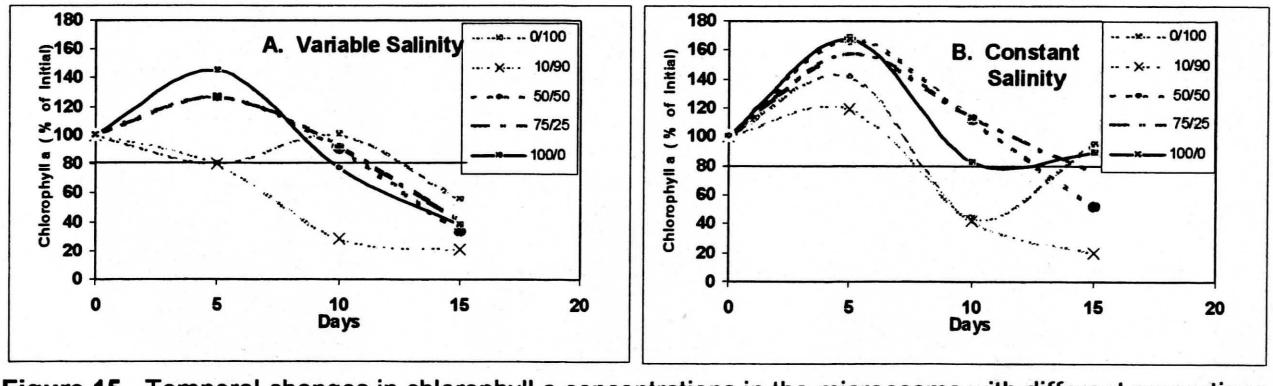

Figure 15. Temporal changes in chlorophyll a concentrations in the microcosms with different proportions of Farmington Bay and Bridger Bay water.

Phytoplankton grew better in the constant salinity treatments than in the variable salinity treatments. This occurred even in the 100% Bridger Bay CS treatment that grew better than the 100% Bridger Bay VS treatment, even though neither had salt added. This suggests that the microcosm containers in the CS treatment may have been contaminated with some nutrient growth factor.

There were definite differences in the growth rates of different taxa in water from the different bays, but the response was different in the Constant Salinity and Variable Salinity treatments (Fig. 16). For example, small chlorophytes grew better in the Constant Salinity treatments as the proportion of Bridger Bay water increased, but in the Variable Salinity treatment this response was absent or reversed (Fig. 16). Filamentous cyanophytes (blue green algae) in both treatments did best in treatments with high proportions of Farmington Bay water.

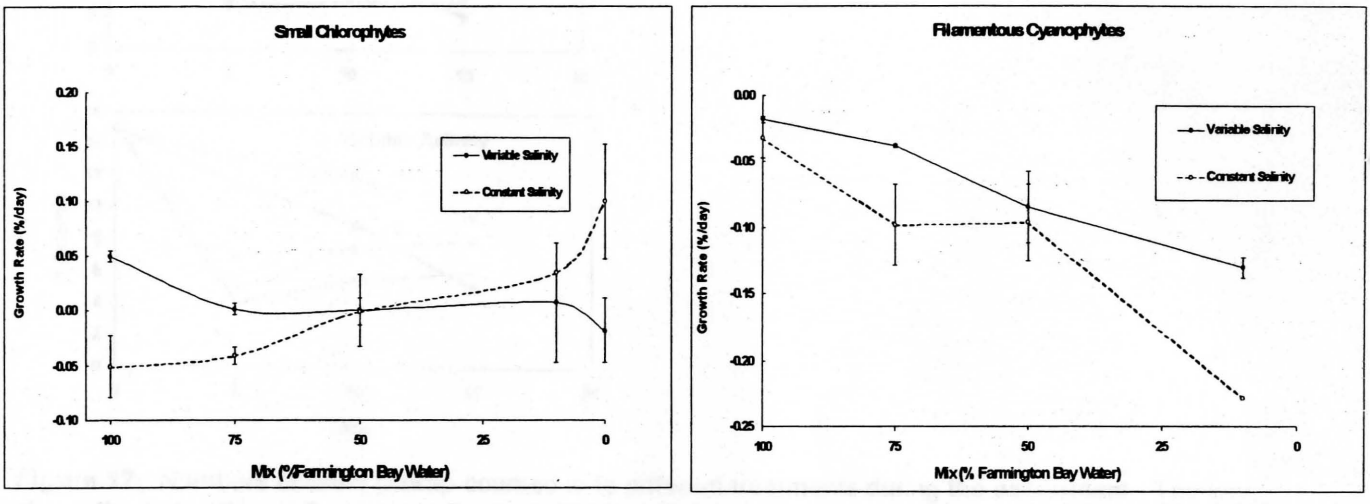

Figure 16. Instantaneous growth rates of small chlorophytes (left) and cyanobacteria (right) over the 15-day experiment in different relative mixes of Farmington Bay and Bridger Bay water.

Initial survival rates of brine shrimp were best in treatments with 10% Farmington Bay water and 90% water from the Great Salt Lake (Fig. 17). However, variability between replicates was high, making it difficult to determine trends. When we calculated instantaneous mortality using data from day 0, 10 and 15, there was no significant difference in survival among salinity treatments, but there was significantly higher mortality in the Constant Salinity treatment than in the Variable Salinity treatments (Fig. 18; $p < 0.001$).

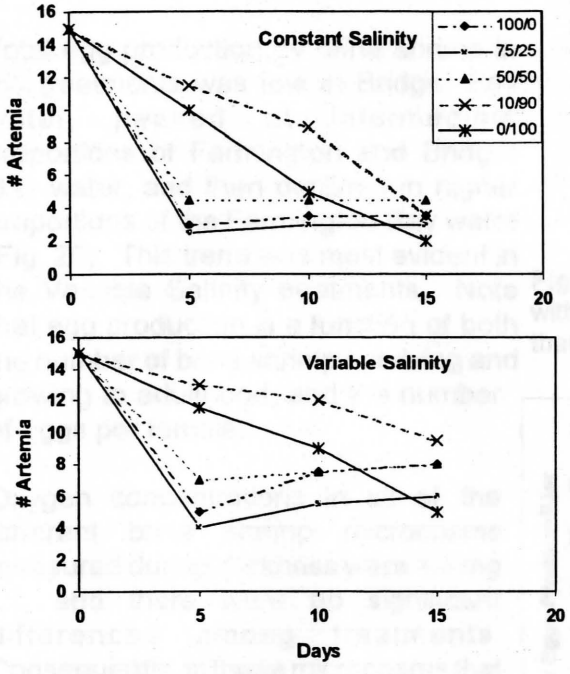

Figure 17. Numbers of brine shrimp counted in te different treatments during the experiment. The key shows the proportion of Farmington Bay water/Bridger Bay water. For example, 100/0 means 100% Farmington Bay and 0% Bridger Bay water. Densities on day 5 may be biased (see textA).

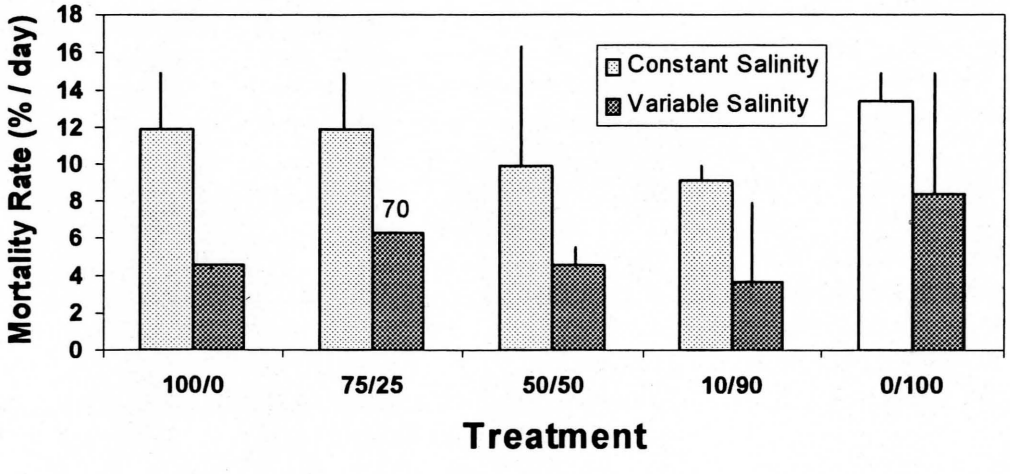

Figure 18. Instantaneous mortality rates +s.d. of brine shrimp during the 15-day experiment in constant salinity or variable-salinity treatments. The x-axis shows the percentages of Farmington Bay (1st number) and Bridger Bay (2nd number water used. In both the constant and variabe salinities, the lowest mortality rates were in mixtures with 10% Farmington Bay water and 90% Bridger Bay water.

Growth rates of brine shrimp were low in the Bridger Bay water with low chlorophyll levels, but increased when only a small portion of chlorophyll-rich Farmington Bay water was added. Additional chlorophyll increased growth little (Fig. 19).

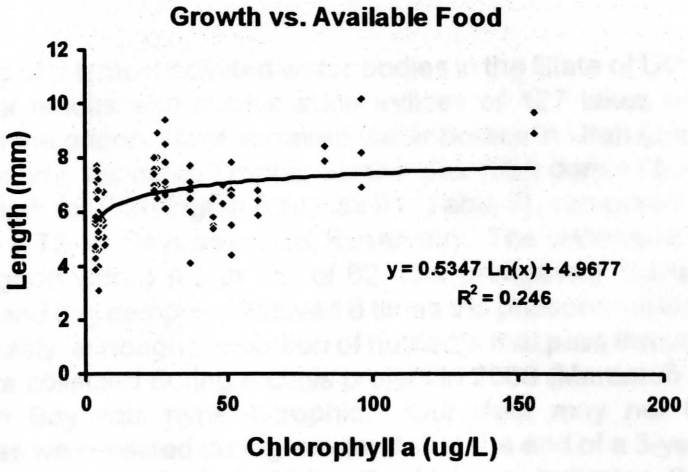

Figure 19. Final lengths of brine shrimp in the experiments with varying mixes of Farmington Bay and Bridger Bay water that produced different chlorophyll levels.

Total egg production by brine shrimp in the treatments was low in Bridger Bay water, peaked at intermediate proportions of Farmington and Bridger Bay water, and then declined in higher proportions of the Farmington Bay water (Fig. 20). This trend was most evident in the Variable Salinity treatments. Note that egg production is a function of both the number of brine shrimp surviving and growing to adulthood, and the number of eggs per female.

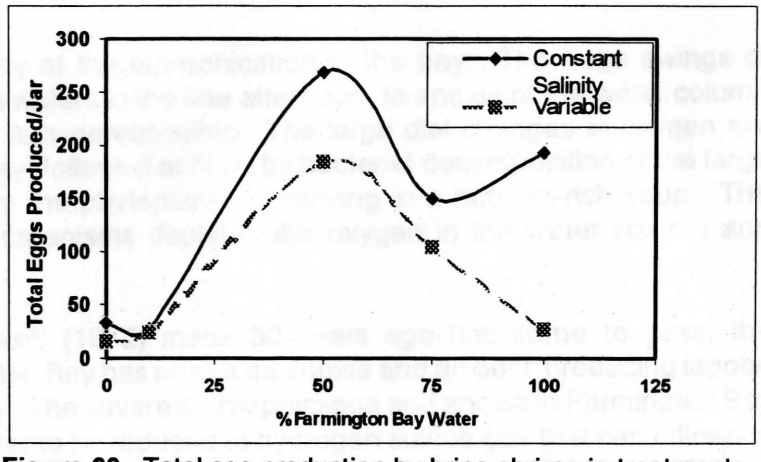

Figure 20. Total egg production by brine shrimp in treatments with variable proportions of Farmington Bay water mixed with Bridger Bay water from the Great Salt Lake.

Oxygen concentrations in all of the different brine shrimp microcosms measured during darkness were > 4 mg L^{-1}, and there were no significant differences among treatments. Consequently, in these microcosms that lacked sediments, the water did not become anoxic overnight.

Discussion

Our data indicate that Farmington Bay is one of the most polluted water bodies in the State of Utah. We base this conclusion by comparing our results with trophic state indices of 127 lakes and reservoirs greater than 50 acres that are on the priority list of impaired water bodies in Utah (Judd 1997). Using the data collected during our study, the mean Trophic State Index (TSI) derived from chlorophyll, total phosphorus and Secchi depth for Farmington Bay was 91 (Table 3), compared to the most eutrophic reservoir on the list with a TSI of 74 (Lower Box Reservoir). The water quality in Bridger Bay, in contrast, was relatively good with a mean TSI of 62. Our preliminary nutrient loading estimate suggests that the wetland and bay complex received 8 times the phosphorus load that is acceptable to maintain good water quality, although the portion of nutrients that pass through the wetlands into the bay is unknown. Data collected during a class project in 2000 (Marcarelli et al. 2001) also indicated that Farmington Bay was hypereutrophic. Our data may not be characteristic of the entire summer period, as we collected during a warm fall at the end of a 3-year drought. However, data collected by the Utah Division of Water Quality also indicates that Farmington Bay has severe water quality problems. Using their Secchi depth and total phosphorus data for 2000 yields a mean TSI of 76, still the highest value of any system in the state. Despite its extremely poor water quality, Farmington Bay is not on the state's priority list of impacted water bodies.

Oxygen levels also indicate the severity of the eutrophication in the bay. The huge swings of oxygen from supersaturation of surface waters in the late afternoon, to anoxia of the water column by early morning indicate that the bay is hypereutrophic. The large diel changes in oxygen are caused by photosynthesis during the day, followed at night by bacterial decomposition of the large amount of organic matter produced by the phytoplankton growing in a nutrient-rich soup. The respiration of the bacteria and other organisms depletes the oxygen in the water column and sediments.

Thus the warning of Coburn and Eckoff (1972) made 30 years ago has come to pass: the degradation of water quality in Farmington Bay has produced anoxia and an odor-producing lagoon upwind from metropolitan Salt Lake City. The severe euthrophication and anoxia in Farmington Bay allows the abundant sulfates in the water to be reduced to hydrogen sulfide gas that can influence metropolitan Salt Lake City. Noxious hydrogen sulfide was present in the water and particularly in the sediments of the bay, and more was produced under simulated anoxic conditions in the laboratory. The production of odor-causing hydrogen sulfide in the bay is not new: Carter et al. (1971) and Israelsen et al. (1985) noted that it was present in the sediments and water. Production of hydrogen sulfide and other odors is not limited to Farmington Bay, as smaller amounts are produced in marshes bordering the lake and were even noted by early explorers visiting the Great Salt Lake (Lazar, in press). The main south basin of the lake seldom, if ever, produces objectionable odors (W. Wurtsbaugh, personal observation). Bear River Bay does not produce the intense, objectionable odors characteristic of Farmington Bay, even though the two bays have similar morphometric and hydrological characteristics (Personal communications, S. Manes, Harold Crane Wildlife Refuge and J. Dolling, Farmington Bay waterfowl Management area). Quantitative analyses of the odor problems in Farmington Bay and elsewhere in the lake are badly needed, but unfortunately, there is no agency directly responsible for this problem (Personal communication, J. Pitkin, Utah DWQ). Despite the lack of quantitative data, the preliminary observations suggest that the odor problems influencing lakeside communities are due to severe water pollution in Farmington Bay, and not to the innate characteristics of the Great Salt Lake.

In addition to odor problems, the hypereutrophic condition of Farmington Bay may deplete invertebrates upon which bird populations depend. Our sampling in October indicated that there were considerably less brine shrimp in Farmington Bay than in the main lake, and that overall

zooplankton biomass was lower in the bay. The microcosm experiments indicated that brine shrimp survival after 15 days was similar in water from the two sites, suggesting that salinity or algal food composition in Farmington Bay may be sufficient for brine shrimp. However, oxygen levels in the microcosms were relatively high, even at night, whereas the anoxia in the bay may preclude brine shrimp from thriving. Instead, the zooplankton population in Farmington Bay was dominated by rotifers, and also by air-breathing corixids. This was also noted in field sampling during the previous year's class project on Farmington Bay (Marcarelli et al. 2001). The corixids, however, may be an important food source for birds (J. Caudell, Utah State University, personal communication), and bird populations in the bay are high (C. Perschon, UDWR, personal communication). Our zooplankton data, however, must be interpreted cautiously. Our sampling site was near the causeway, and salt wedges from the main lake intrude into Farmington Bay. It is possible that the brine shrimp we did encounter in the bay were brought in via this intruding water mass. Additionally, zooplankton populations are highly dynamic, and sampling on a single date can provide biased results. Our laboratory experiment was also compromised, as variability between replicates was high, and we encountered unexplained differences between the constant salinity and variable salinity treatments that could possibly have been due to contaminated containers. Because of its large size, the potential production of brine shrimp and other invertebrates in the bay is very important, both for the commercial brine shrimp industry, and for the birds that depend on the invertebrate prey base. Brine shrimp survival will be influenced not only by eutrophication, but also by the salinity (Hayes, 1971; Wurtsbaugh 1992), which is now influenced by the Antelope Island causeway that impedes mixing between the lake and bay. The relative impact of salinity changes and euthrophication on the invertebrate populations is not understood. Clearly, more research is needed on the zooplankton and benthic community in Farmington Bay to understand how these anthropogenic factors have modified the biotic community of the bay.

Are there solutions to the eutrophication problem in Farmington Bay? Human and industrial wastes have been dumped into Farmington Bay for a century with little regard for the impact on the system. Currently the nutrients and other wastes from more than 500,000 people in the metropolitan area enter the bay. The effluents from 10 of the12 sewage treatment plants in the Salt Lake Valley reach the bay, creating a tremendous nutrient load. The construction of the Syracuse-Antelope Island causeway exacerbates the problem by restricting the exchange of water between the bay and the main lake. Furthermore, Farmington Bay is very shallow so that the nutrients are concentrated in a relatively small volume of water, thus providing optimal conditions for algal growth. Thus the buildup of nutrients and their containment in a restricted area presents real challenges for reducing euthrophication. The peculiar chemical characteristics and the biota in the bay provide additional challenges for investigators.

The greatest challenge for improving the water quality in the bay, however, will be overcoming the neglect it has suffered. Only recently has a monitoring program been initiated by the State and Davis County and by the federal NAQWA program (Giddings and Stephens 1999). The lack of studies is surprising give that the odor problems from Farmington Bay likely impact more people in the state than are affected by any other polluted water body. The impact of euthrophication and anoxia on the biota in the bay may also be substantial, although adequate data to determine these impacts are wanting. Because of the severity of the problem for both human and wildlife welfare, Farmington Bay needs to be added to Utah's list of impacted water bodies following the provisions of the Clean Water Act (303d listing). Before progress can be made in restoring Farmington Bay to a condition closer to the relatively good water quality like that in Bridger Bay on the main lake, considerable efforts will need to be focused by the state and non-governmental groups dedicated to maintaining healthy aquatic ecosystems.

Acknowledgments

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References

APHA. 1992. Standard methods for the examination of water and wastewater. Eighteenth edition. American Public Health Association, Washington, D.C.

Austin, L. 1993. unpublished data. Assistant Director, Utah DWQ.

Basbug, Y. and Demirkalp, F.Y. 1997. A note on the brine shrimp *Artemia* in Tuz Lake (Turkey). Hydrobiologia 353:45-51.

Carter, C.K. (Coordinator). 1971. Some ecological considerations of the Farmington Bay estuary and adjacent Great Salt lake Park. University of Utah, Salt Lake City, Utah.

Carter, J., T. Hoagland and A. Coburn. 1971. Geological and hydrological characteristics of Farmington Bay. Westminster College and University of Utah, Salt Lake City, Utah.

Cooke, G.D., E.B. Welch, S.A. Peterson and P.R. Newroth. 1993. Restoration and management of lakes and reservoirs. 2nd ed. Lewis Publishers.

Coburn, A. 1972. Pollution input from the lower Jordan basin to Antelope Island estuary. M.S. Thesis, Dept. of Civil Engineering, University of Utah, Salt Lake City, Utah

Coburn, A. and D.W. Eckhoff. 1972. Pollution input from the lower Jordan basin to Antelope Island esturary i the Great Salt lake and Utah's water resources. Am. Wat. Res. Assoc., Utah Section, Annual Conf., 1st, Proceedings. Pp. 12–140.

Ferree, M.A. and R.D. Shannon. 2001. Evaluation of a second derivative UV/visible spectroscopy technique for nitrate and total nitrogen analysis of wastewater samples. Water Res. 35: 327-332.

Giddings, E.M. and D. Stephens. 1999. Selected aquatic biological investigations in the Great Salt Lake Basins, 1875-1998, National Water-Quality Assessment Program, U.S. Geological Survey Water-Resources Investigations Report 99-4132, 42 p.

Hayes, C.R. 1971. Distribution, populations, and species diversity of phytoplankton and zooplankton of Farmington Bay. In: Carter, C.K. (ed). Some ecological considerations of Farmington Bay estuary and adjacent Great Salt Lake State Park. University of Utah Library, Salt Lake City, Utah. Pp. E2-E21.

Lazar, S. In press. Great Salt Lake odors. Friends of Great Salt Lake information paper. Salt Lake City.

Linge, K. 2001. Investigations into the arsenic and phosphorus enrichment of Lake Yangebup. http://www.cwr.uwa.edu.au/~linge/methods.htm

Linge, K. L. And C.E. Oldham. 2001. Interference from arsenate when determining phosphate by the malachite green spectrophotometric method. Analytica Chimica Acta 450:247-252.

Marcarelli, A.M., M.D. Mills and W.A. Wurtsbaugh. 2001. The Great Salt Lake doesn't stink...But Farmington Bay does! Friends of Great Salt Lake Newsletter 7(2): 5, 13; 7(4):6.

Nydahl, F. 1978. On the peroxodisulphate oxidation of total nitrogen in waters to nitrate. Water Research 12:1123-1130.

Reeve, M.R. 1963. Growth efficiency in *Artemia* under laboratory conditions. Biological Bulletin 125:133-145.

Vollenweider, R. A. 1976. Advances in defining critical loading levels for phosphorus in lake eutrophication. Mem. Ist. Ital. Idrobiol. 33: 53-83.

Welschmeyer, N.A. 1994. Fluorometric analysis of chlorophyll a in the presence of chlorophyll b and phaeopigments, Limnol. Oceanogr. 39: 1985-1992.

Wetzel, R.G. and G. Likens. 2000. Limnological Methods. 3nd ed. Springer.

Wurtsbaugh, W.A. 1992. Food-web modification by an invertebrate predator in the Great Salt Lake (USA). Oecologia 89:168-175.

Wurtsbaugh, W.A, and Berry, T.S. 1990. Cascading effects of decreased salinity on the plankton, chemistry, and physics of the Great Salt Lake (Utah). Canadian Journal of Fisheries and Aquatic Sciences 47:100-109.Wurtsbaugh, W.A. 1992. Food-web modification by an invertebrate predator in the Great Salt Lake (USA). Oecologia 89:168-175.