2009 Project Report

Limnological Analyses of Cutler Reservoir and Dingle Marsh with Respect to Eutrophication

Aquatic Ecology Practicum (WATS 4510) Class Report
Watershed Sciences Department
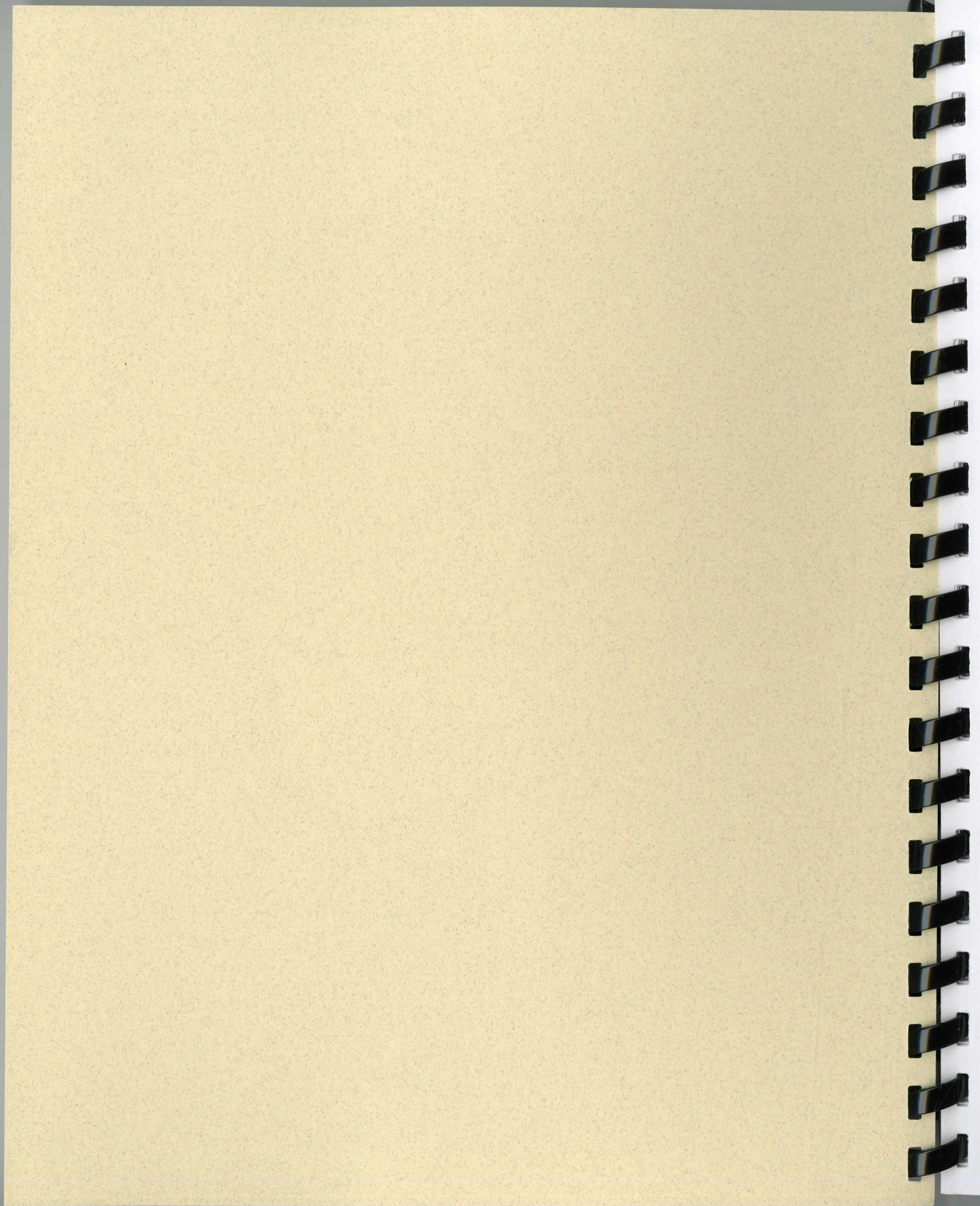College of Natural Resources
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Acknowledgements

Executive Summary

Cutler Reservoir is located in Cache county, Utah and was created for the purposes of irrigation, water storage and flood control. High nutrient loading to Cutler has raised concerns about the health of this system and has resulted in it being listed on the state's 303(d) list of impaired waters. The TMDL plan being drafted for Cutler lists dissolved oxygen and phosphorous as the key issues of concern. The underlying problem created by nutrient loading is eutrophication. If Cutler is to remain as a valuable source of recreation, wildlife habitat, and water for the Cache Valley we must understand the underlying processes that control the system.

In September, 2009, Dr. Wayne Wurtsbaugh's Aquatic Ecology course began a series of student projects to better understand certain ecological and limnological characteristics of Cutler reservoir. Dr. Wurtsbaugh's classes from previous years have also analyzed limnological characteristics of Cutler Reservoir and Dingle Marsh. Dingle Marsh (Idaho) is a less impacted reference site situated higher in the watershed. The student projects were quite diverse this year and touched on a variety subjects.

Gil Rowley's study compared dissolved oxygen, total phosphorous and chlorophyll *a* levels at 17 sites throughout Cutler Reservoir, and additional samples were taken in Dingle Marsh. The mean value for chlorophyll *a* in Cutler was 32 µg/L indicating that the reservoir was eutrophic. However, concentrations ranged from 4 µg/L in the southern area, to 107 µg/L near the discharge of the Logan Wastewater Treatment Plant. Dissolved oxygen levels were measured at dawn to capture the lowest diel concentrations. Nevertheless, mean oxygen levels were high, with a mean of 8.9 mg/L, but there was significant spatial variability. The mean total phosphorous concentration was 0.22 mg/L, again indicating a eutrophic state. Mean respective concentrations of the limiting nutrient in Cutler and Dingle (nitrogen) were 1.08 and 0.45 mg TN/L. Gil demonstrated that the minimum oxygen concentrations in Cutler were negatively correlated with total phosphorous and chlorophyll *a* concentrations.

Justin Stout used recording oxygen sondes to asses the effects of two emergent bulrush beds on dissolved oxygen levels near the Logan & Little Bear River inflows to Cutler Reservoir. Both the minimum and maximum oxygen concentrations were generally lower at sites in or below the bulrush beds, than at up-flow sites, but the results were not statistically significant. Justin also found very high oxygen concentrations (diel range 9.6 to 20.7 mg/L) during the September study. Justin's project raises interesting questions regarding the interaction of these aquatic plants and oxygen levels. Peak saturation levels were >250% indicating that high oxygen levels may be toxic to fish in Cutler at this time of the year.

Dan Lamarra analyzed stable isotopes of nitrogen and carbon from cores and surficial sediments to understand patterns in nutrient deposition over time and space between Cutler Reservoir and Dingle Marsh. Cutler Reservoir and other sites within Cache Valley were significantly more enriched with the $\delta^{15}N$ stable isotope than were the sediments in Dingle Marsh. This suggests that sites with higher anthropogenic inputs are steadily enriched along and within the watershed as one moves from areas of low to high anthropogenic impacts.

Continuing with stable isotope analysis, Collin Cook's project focused on aquatic food webs in the Cutler and Dingle wetlands. Collin's project showed that Cutler was more enriched with the stable isotope of nitrogen across multiple trophic levels compared to Dingle Marsh. The elevated levels of $\delta^{15}N$ values again suggest higher level of anthropogenic nutrient inputs to Cutler than to Dingle Marsh.

Ryan Leonard and Ben Marett used aquatic invertebrate biotic indices and density estimates to compare Dingle and Cutler. The benthic invertebrate samples collected in their project are some of the largest taken to date and will allow for future studies to compare changes over time. Dingle Marsh higher had a higher number of pollution intolerant taxa than in Cutler. Hilsenhoff Biotic Indices indicted that Cutler had lower water quality than Cutler, but the differences were not large. Overall benthic invertebrate densities were very low in both systems, but densities were twice as high in Dingle as in Cutler. Although these differences were not statistically significant, taken together, they suggest that Cutler Reservoir is impaired relative to the reference site.

The final report was created by J.D. Abbott and Deb Collins to look at paleolimnogical characteristics of Cutler Reservoir. A 19.5-cm long core was collected from the northern part of the reservoir and 25 sections were extracted to analyze sediment phosphorous, nitrogen and phaeophytin pigments (the product of chlorophyll degradation) over time. Portions of each section were sent out for lead-210 analysis and dated. The lead 210 data suggest that the sediments at the collection site had been disturbed, making it impossible to give an accurate chronology for the different depth strata. Nevertheless, the best estimate for the sediment core dating suggests that the core spanned from 2009-1933. Phosphorous levels increased from 0.6 mg P/g in ~1933 to approximately 0.75 mg P/g in 2009. Nitrogen and carbon concentrations also increased 25-30% over the span of the core, but phaeophytin pigment levels displayed no pattern with time.

Overall, the chemical and biological results indicate that Cutler Reservoir is affected by nutrient loading and that it is moderately eutrophic. However, during the September study, oxygen levels were relatively high; suggesting that impairment at this time of the year was not severe.
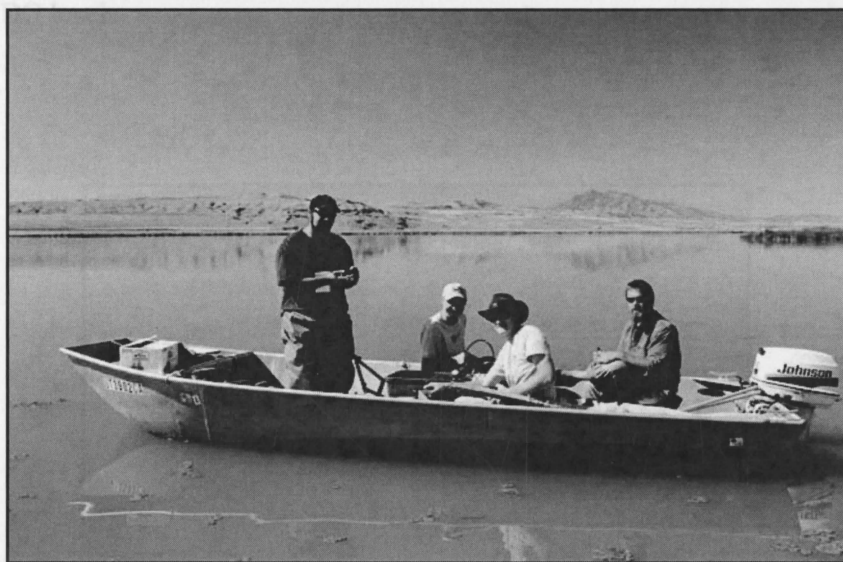
Sampling in Cutler Reservoir

Spatial Comparisons of Dissolved Oxygen, Total Phosphorus, and Chlorophyll-a Concentrations in Cutler Reservoir

Gilbert Rowley

Abstract

Cutler Reservoir–located in Cache County, Utah–has been listed as impaired under the Clean Water Act because of low dissolved oxygen (DO) concentrations and high levels of phosphorus. In this study I compared changes in DO concentrations on a large spatial scale to see how DO levels are affected spatially in Cutler Reservoir. Total phosphorus (TP), chlorophyll-a (Chl-a), depth, temperature, specific conductivity, distance to vegetation, and DO were all measured at 17 sites spread throughout Cutler Reservoir. The samples were all collected before dawn to capture the minimum DO levels at the sites. During the September study the respective mean levels for temperature, Chl-a, phosphorus and minimum DO were 15.5°C, 32 μg/L, 0.22 mg P/L and 8.9 mg O_2/L. Dissolved oxygen concentrations were negatively correlated with TP and chlorophyll concentrations, while depth, temperature, specific conductivity, and distance to vegetation did not show a strong relationship with DO. DO concentrations varied across the reservoir, with the lowest levels near the outflow of the Logan Wastewater Treatment Plant, but there was not a consistent relationship with distance from this point source. Additionally, TP and Chl-a concentrations varied greatly with spatial location as did DO concentrations. This may be caused by different tributaries and sources of inflow that bring excess nutrients into the reservoir. Therefore, understanding spatial variation in nutrient loading in Cutler reservoir is important for predicting DO levels.
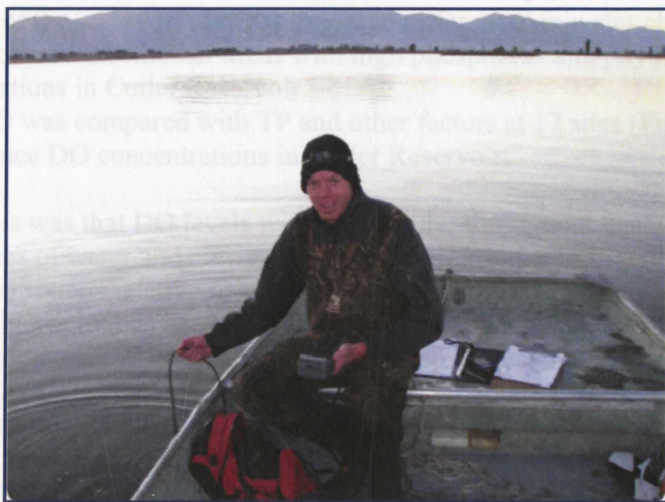
Gil Rowley —Early morning O_2 measurements in Cutler

Introduction

Low dissolved oxygen (DO) levels are commonly found in wetlands (Dodds 2002). Low concentrations of DO can be harmful to fish, invertebrates, and other aquatic life (Budy et al. 2007, Spieles and Mitsch 2000, SWCA 2009). Because of its importance to aquatic life, DO is often used as an indicator of water quality. Cutler Reservoir, located in Cache County, Utah, has been identified under Section 303(d) of the Clean Water Act as impaired because of low DO concentrations and excess phosphorus loading (SWCA 2009).

Within Cutler Reservoir, DO levels fluctuate on a diel cycle (Mason and Elsner 2009, Baillie 2007, SWCA 2009). This is a common characteristic found in many wetlands (Dierberg et al. 2002, Ford et al. 2002, and McCormick Laing 2003). The diel cycle is directly affected by photosynthesis, respiration of aquatic plants and animals, and daily changes in temperature (SWCA 2009). My study focused on the daily minimums associated with diel fluctuation since DO minimums are among the top concerns of managers working with Cutler Reservoir.

Among other factors, total phosphorus (TP) from both point and non-point source pollution is causing DO concentrations in Cutler Reservoir to decrease (SWCA 2009). Phosphorus is often associated with algal growth, which often regulates DO concentrations. Excessive nutrient loading increases the algal content, which in turn, increases primary production. Algal respiration and decomposition of the sedimenting algae decreases DO concentrations.

There are many contributing sources to the pollution problems that Cutler Reservoir is currently facing. Phosphorus is receiving the majority of attention at this time because it is currently found in excess amounts in Cutler Reservoir (SWCA 2009). Nonpoint sources that are adding phosphorus to the Cutler system include; storm water runoff from impervious surfaces, runoff from land engaged in agricultural practices, and naturally occurring processes (SWCA 2009). Other sources of pollution in and around the Cutler Reservoir area include regulated point sources from wastewater treatment facilities and industrial activities (Figure 1) (SWCA 2009.

Since Cutler Reservoir receives water and nutrients from many different sources, it is possible that certain areas of the reservoir may exhibit large differences in water chemistry and aquatic life). My study focused on whether areas with high phosphorus and phytoplankton resulted in low DO concentrations in Cutler Reservoir.
Consequently, DO was compared with TP and other factors at 17 sites (Figure 2) to see what parameters influence DO concentrations in Cutler Reservoir.

My null hypothesis was that DO levels would be similar throughout Cutler Reservoir. I predicted that various sources of water and nutrient inflow to Cutler Reservoir would not significantly change DO concentrations relative to site location. An additional null hypothesis was that total phosphorus and chlorophyll would be similar throughout the reservoir, and would not be correlated with minimum DO concentrations. My project was, in part, a collaborative study with Justin Stout's project (also included in this report), which looked at fine-scale spatial variation in diel oxygen concentrations relative to macrophyte beds. Therefore, an additional null hypothesis for my study was that the distance to emergent macrophytes would not play an important role in DO levels over a large spatial scale.

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Methods

Sampling took place on three different mornings between 5:00 and 8:15 a.m., just before and just after sunrise. Data collected in 2007 and 2008 from Cutler Reservoir show that this is the time of day when DO levels are at their lowest (Mason and Elsner 2009, Baillie 2007). On 22 September 2009 the southern reaches of the reservoir were sampled (Sixth South, Valley View, and Cache Junction sites) by wading from shore. The following morning, 23 September 2009, samples were taken by boat at various sites located south of Benson Marina (Sites 1-9). On 28 September 2009 sampling was conducted at five sites north of Benson Marina which concluded sampling (Sites 10-14). Two field replicates were taken from sites 1 and 3 (sampled earlier on 22 September 2009). Replicate samples were taken to account for possible changes in weather patterns, boat traffic, or any other outside influences that may have affected previously collected data. The week prior to the collections mean daily air temperatures were 18.6°C (65° F), on the 22nd they had fallen to 12.2° C (54 °F). By the collections on the 23rd and 28th they had warmed somewhat to 15.6 °C (Weather Underground website archived data).

Sites were selected the morning of sampling. Sites both near and far from shore were selected in the main channel. These sites were spread out to try and cover a large area of the reservoir so that large spatial analyses could be made. Seventeen sites were sampled between the hours of 5am and 8:15am.. All of the sites chosen were relatively shallow (<2.7 m), and no thermal stratification was observed.

Parameters Measured—At each sample site, global position system (GPS) coordinates were taken. DO concentrations (milligrams per liter; mg/L), percent saturation of DO, water temperature (°C), and specific conductivity (microSiemen/cms; μs) were measured using a calibrated Model 85 YSI oxygen-temperature-conductivity probe. The station depth and time were also recorded (Appendix 1). Measurements were taken at 0.5 meter from the water's surface, and 0.2 meters from the bottom of the reservoir at those sties that exceeded 0.75 m in depth. For sites that were less than 0.75 m deep, a single reading at mid-depth was taken. Distance from vegetation was approximated by sight and recorded. Grab water samples for Chl-a were taken at each site ~ 40 cm below the water's surface with dark-colored 125-ml bottles. TP grab samples were taken with 125-ml HD polyethylene bottle 40 cm below the water's surface. Both the Chl-a and the TP samples were then immediately placed in a cooler on ice until analyzed.

Laboratory Analyses—Ten milliliter samples for Chl-a analysis were filtered the same day on 25-mm Gelman A/E, wrapped in aluminum foil and frozen. Subsequently, the filters were placed in 95% ethanol and stored in the dark for 24 hours to extract the Chl-a. The solution was then analyzed with a Turner 10 AU fluorometer using the non-acidification method of Welschmeyer (1994) to determine the amount of Chl-a present in each sample.

Water samples collected for TP analysis were frozen. The TP samples were sent to the Utah State University Aquatic Biogeochemistry Lab for further analyses. The samples were digested with persulfate (Valderrama 1981), and the resulting phosphate was analyzed by the method of Murphy and Riley (1962). The digested samples from 4 sites in Dingle and 11 sites in Cutler Reservoir were also analyzed for total nitrogen (TN) utilizing the cadmium reduction method.

Statistical Analysis—Each of the parameters measured was plotted against DO and regression analyses were conducted comparing the different parameters to DO concentrations. Chl-a and TP were compared and a regression analysis was done in Excel to determine their relationship in Cutler Reservoir.

GIS Analysis—To visualize possible external influences that may be contributing to nutrient loading in Cutler Reservoir, ArcMap was used to create maps of Cutler Reservoir and its surrounding area. The maps offer a visual representation of some of the influencing factors contributing to low DO concentrations and high phosphorus levels.

Results

DO concentrations varied markedly between the 17 stations. The mean DO concentration was 9.01 (mg/L) with a standard deviation of 1.43 mg/L (Figure 3). DO concentrations varied between 5.9 mg/L (site 6) and 11.9 mg/L (site 7) (Figure 4 and Table 1). At the northern most sample site, near Cache Junction, the DO was 7.4 mg/L. At the southern most site, near the Valley View Highway, the DO concentration was 8.7 mg/L.

Total phosphorus concentrations followed a pattern similar to DO concentrations (Figure 5; Table 1). The mean TP concentration in Cutler Reservoir was 219 µg/L (0.219 mg/L) with a standard deviation of 192.60 µg/L (Figure 3). There was a wide degree of variation in TP concentrations, which accounts for the high standard deviation. The highest phosphorus level was 850 µg/L (site 6) near the Wastewater Treatment Plant (WWTP) discharge. The lowest was 43 µg/L at site 14, which is located between Clay Slough and Newton Creek on the northern end of the reservoir. Mean respective TN concentrations in Cutler and Dingle were 1.08 ($\pm$ 0.14 s.e.) and 0.45 $\pm$ (0.05) mg/L (Table 2). The TN/TP (by weight) ratios for Cutler and Dingle at these sites were 4.7:1 (Cutler) and 12.9 (Dingle), indicating that phytoplankton growth in Cutler reservoir is N-limited, and growth in Dingle is P-limited.

The Chl-a concentrations also varied markedly between sample sites (Figure 6 and Table 1). The minimum Chl-a concentration was 3.3 µg/L (site 9) and the maximum was 105 µg/L (site 6) near the WWTP. The mean Chl-a concentration at Cuter Reservoir was 32.4 µg/L $\pm$ 25.0 µg/L s.d. (Figure 3). Similar to TP samples, there were large variations in Chl-a values between sites, which accounts for the high standard deviation.

There was a marginally-significant positive relationship between TP and Chl-a ($p = 0.065$; R^2 0.0186; Figure 7). Minimum dissolved oxygen concentrations in Cutler were negatively correlated with TP concentrations ($p = 0.0065$, $R^2 = 0.350$; Figure 8). DO and Chl-a were also correlated, which yielded a p-value of 0.036 and an R^2-value of 0.266, showing a significant relationship between these two parameters (Figure 9). These data indicate that DO is significantly influenced by the presence of TP and Chl-a.

Distance to the nearest vegetation was not correlated with DO levels ($p = 0.095$; $R^2 = 0.155$; Figure 10). This indicates that distance from vegetation was not a strong contributor to DO concentrations in the reservoir.

The DO concentrations were also regressed against depth, temperature, and specific conductivity. None of these yielded strong or significant relationships with DO concentrations (Appendix 2). Temperatures were relatively consistent across the reservoir (13.4-17.4°C). DO was not strongly affected by temperature differences because significant differences in temperature were not present. With fairly consistent depth from one end of the reservoir to the other, depth was also not a contributing factor.

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Discussion

There are many possible factors that potentially affect DO and TP concentrations in a system like Cutler Reservoir. In Cache County, agricultural and developed lands prevail; these contribute to nonpoint source pollution. Commercial and residential areas contribute to point source pollution. A map of the area around Cutler Reservoir (Figure 11), shows the watershed and sub-watersheds that deliver water, and therefore nutrients, to Cutler Reservoir. Other possible factors that can cause changes in DO concentrations include: wind, cloudbursts, disturbances caused by water craft, anglers, and trains. Algal blooms, and subsequent die-offs, can increase the sediment oxygen demand (SOD) and decrease DO rapidly (SWCA 2009). Consequently, it is difficult to locate the exact source(s) that are the major influencing factors acting on DO concentrations in the Cutler system. Regardless of the sources, the mean total phosphorus of 215 ug/L and the mean chlorophyll a level of 32 ug/L indicate that the system falls within the category of being eutrophic.

It is interesting that both the minimum and the maximum DO concentrations were found in close proximity to each other. Sites 6 and 7 are both located near Swift Slough, which carries the effluent from the Logan Regional Waste Water Treatment Plant (WWTP). Site 6 is located near the shore very close to where Swift Slough enters Cutler Reservoir (Figure 2). It is also in a small cove where nutrients can settle and accumulate because it is away from the main current of the reservoir. On the other hand, site 7 is located approximately 480 m out into the main water body from site 6, which allows for more water movement and a lower retention time for both water and nutrients. This potentially accounts for the higher DO concentration. Baillie (2007) found that TP concentrations were particularly high near the WWTP and suggested that concentrations along an east-to-west gradient moving away from the WWTP effluent might be expected to dissipate due to dilution from the main current. If this same principle is implemented for the entire reservoir, the farther the sample site is from tributaries and inflow water/nutrient sources, the higher the DO concentrations should be. An evaluation of my data supports this hypothesis. The Valley View site is situated close to where the Little Bear River and the Logan River enter Cutler Reservoir. To the north are both sites 8 and 9, which are located closer to the main channel where dilution is likely to occur. Valley View showed a DO concentration of 8.74 mg/L, while site 8 was 10.5 mg/L. Further in the main channel site 9 was higher at 11.45 mg/L (Table 1 and Figure 12). Site 13 is located close to where Clay Slough enters Cutler Reservoir. Both site 12 (located to the south) and site 14 (located to the north) are located more in the main channel and show higher DO concentrations (9.5 mg/L and 8.9 mg/L) than site 13 (8.8 mg/L) (Figure 12). This does not explain all the differences in DO concentrations between sites but may account for a portion of the variability observed.

Although minimum dissolved oxygen levels were negatively correlated with total phosphorus concentrations in Cutler, this may or may not be a causal relationship. The TN/TP ratios measured during this study, as well as previous measurements and bioassays (Abbott 2008) strongly suggest that phytoplankton in Cutler Reservoir are currently controlled by nitrogen rather than phosphorus. TN and TP levels were highly correlated ($r^2 = 0.87$; Valley View outlier removed), so it is likely that it is either TN concentrations or the combination of high TN and TP levels at particular sites that are fueling algal growth and leading to low dissolved oxygen levels. The significance of co-limitation by these nutrients has recently been highlighted (Conley et al. 2008; Lewis and Wurtsbaugh 2008) and the importance of joint control in management decisions is being debated (Schindler et al. 2008; Scott and McCarthy, in press). Because different strategies for removing nutrients from the Logan Wastewater Treatment Plant are currently being discussed, the true nature of algal nutrient limitation in Cutler is highly relevant.

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In this study, DO levels were higher than expected. The DO concentrations fluctuate on a diel cycle with highs usually occurring during early afternoon. The study was designed to capture the lowest DO concentrations of the day. Dissolved oxygen concentrations at dawn were expected to be low inc Cutler Reservoir since it has been listed as impaired based on oxygen levels. However, the DO levels for all of the sites sampled were higher than the minimums required for warm water fishes to survive (Budy et al. 2007). The 1-day minimum State criterion is that DO cannot sag below 3 mg/L for all life stages, and for early life stages there is a 5 mg/L minimum criterion. The 7-day average DO concentration cannot fall below 6 mg/L (SWCA 2009). The high values found in my study may be partially explained by the season in which sampling took place. During the fall season water temperature drops and water is able to retain more DO. Consequently, DO levels are potentially higher at this time of year. Nevertheless, saturation values were high at most sites (Appendix 1). Undoubtedly, there are other contributing factors influencing DO concentrations that may cause them to be higher this time of year. Future studies can re-evaluate DO concentrations over a large spatial scale at Cutler Reservoir by extending the temporal scale during which data is collected. Larger data sets and more replicates would help to better characterize the variability present in DO and TP concentrations throughout Cutler Reservoir.

Conclusions

Variability among DO concentrations, TP and Chl-a, existed over a large spatial scale at Cutler Reservoir during my fall, 2009 analysis. This appears to be caused by different tributaries and sources of inflow that bring excess nutrients into the reservoir. Also, DO concentration was significantly affected by TP and Chl-a concentrations, meaning that low oxygen levels occurred where TP and Chl-a levels were high. DO did not have a significant relationship with temperature, depth, or specific conductivity. Distance to vegetation may also be a contributing factor to low DO levels in Cutler Reservoir. The findings of this project pose interesting questions for future studies of individual components.

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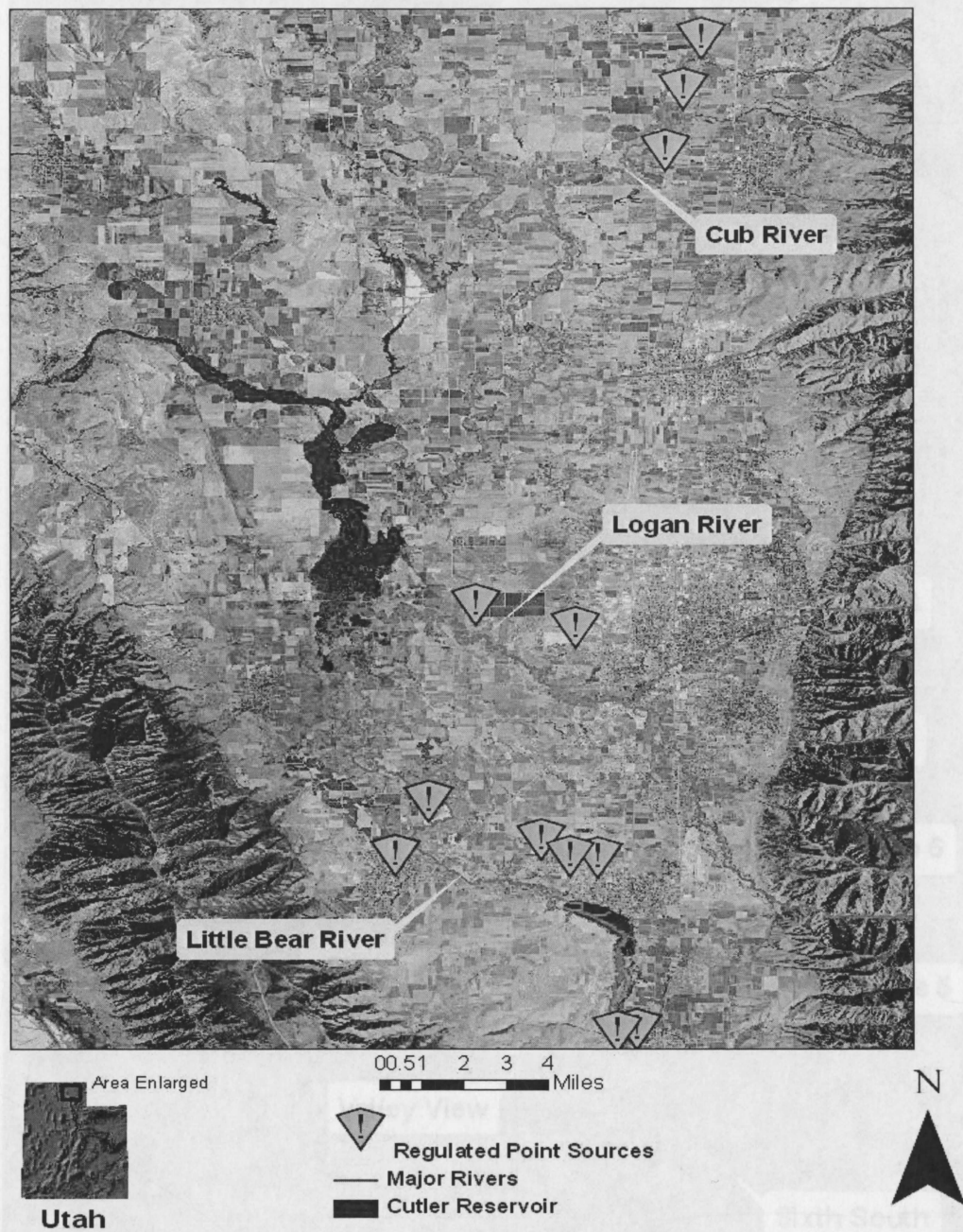
Figure 1. Regulated point sources located in Cache County near Cutler Reservoir (SWCA 2009). Notice that these point sources are located near some of the major tributaries that deliver water and nutrients into Cutler Reservoir. Other regulated sources outside of Cache County identified to be contributing nutrients into Cutler Reservoir include Montpelier, ID Waste Water Treatment Plant (WWTP); Soda Springs, ID WWTP; Grace, ID WWTP; Preston, ID WWTP; Franklin, ID WWTP; Clear Springs Foods, ID; Grace Fish Hatchery, ID; and Bear River Trout Farm, ID (SWCA 2009). Map derived from a 1-meter aerial image taken in 2006, acquired from ftp://ftp.agrc.utah.gov/. Location map, major rivers, and water sources acquired at: http://gis.utah.gov.
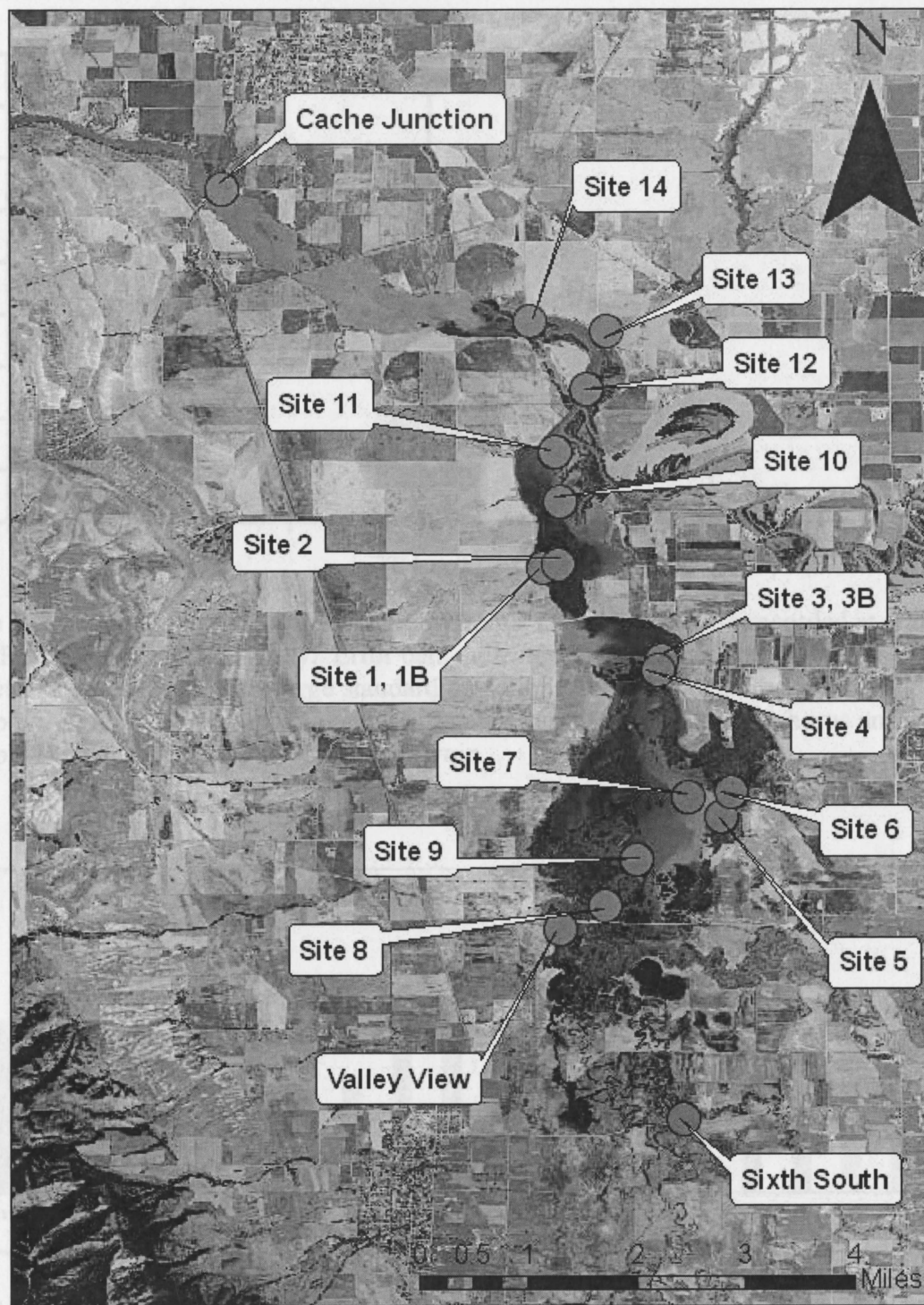
Figure 2. The 17 sample sites in Cutler Reservoir used in this study. Field replicates were taken at Sites 1 and 3 on different days. Map derived from a 1 meter aerial image taken in 2006, and acquired at ftp://ftp.agrc.utah.gov/.
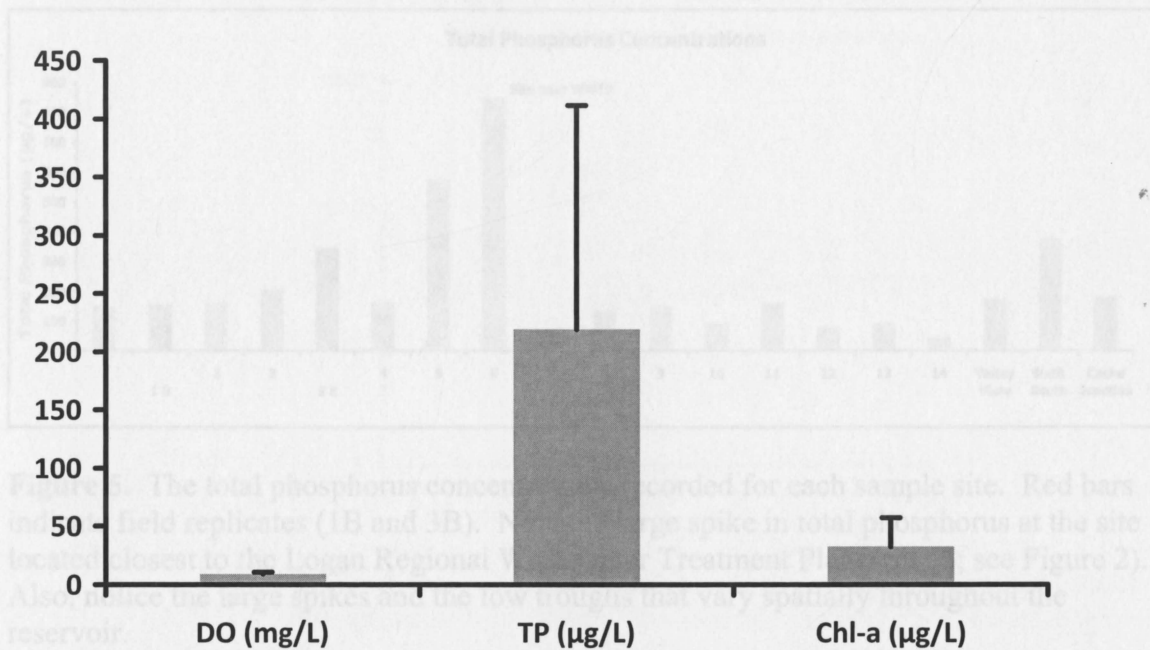
Figure 3. Mean dissolved oxygen, total phosphorus, and chlorophyll-a recorded at Cutler Reservoir during this study. Error bars indicate standard deviation for each parameter. Note the relatively large standard deviations in total phosphorus and chlorophyll-a. This is partially explained by spatial variations in total phophorus and chlorophyll-a concentrations at Cutler Reservoir.
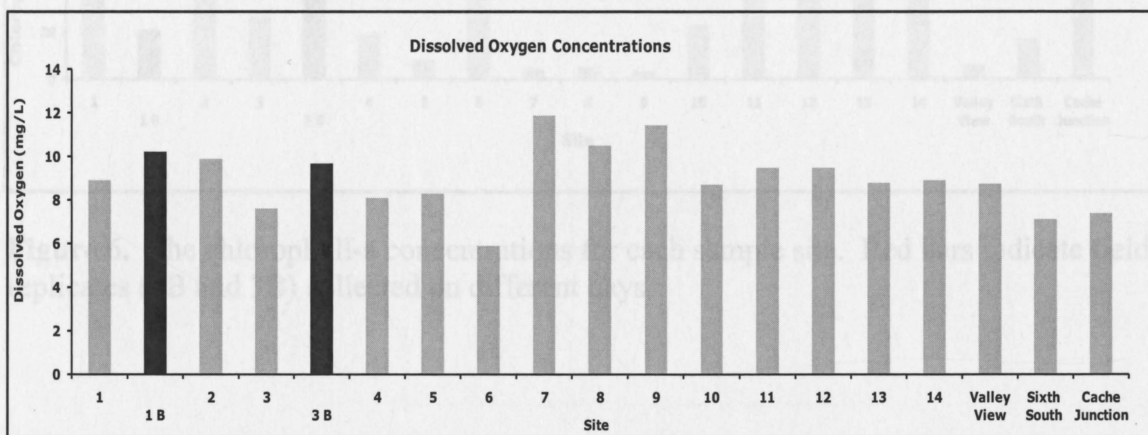
Figure 4. The dissolved oxygen concentrations measured at each site sampled. Red bars indicate field replicates (1B and 3B) collected on different days than the "A" sample.
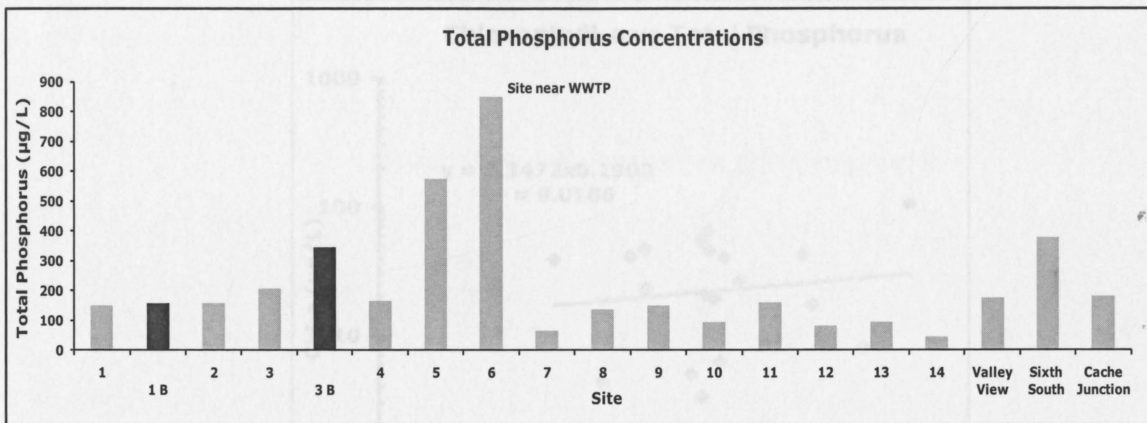
Figure 5. The total phosphorus concentrations recorded for each sample site. Red bars indicate field replicates (1B and 3B). Note the large spike in total phosphorus at the site located closest to the Logan Regional Wastewater Treatment Plant (Site 6; see Figure 2). Also, notice the large spikes and the low troughs that vary spatially throughout the reservoir.
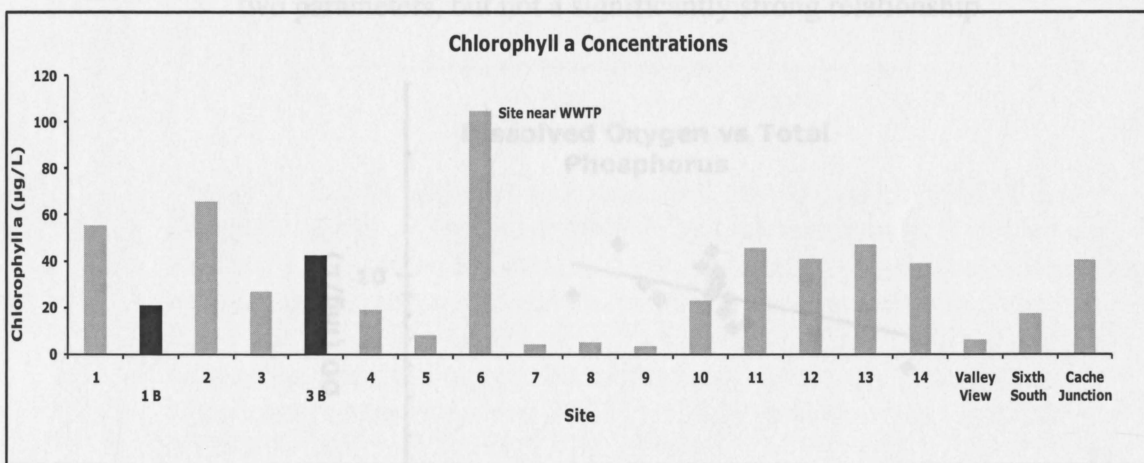
Figure 6. The chlorophyll-a concentrations for each sample site. Red bars indicate field replicates (1B and 3B) collected on different days.
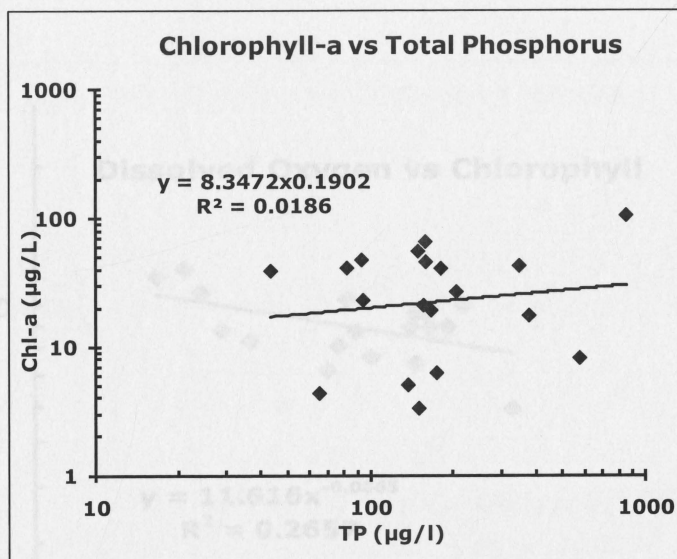
Figure 7. Relationship between chlorophyll-a and total phosphorus at 17 sites in Cutler Reservoir. This analysis yielded a p-value of 0.065 and an R^2-value of 0.019, indicating a possible correlation between the two parameters, but not a significantly strong relationship.
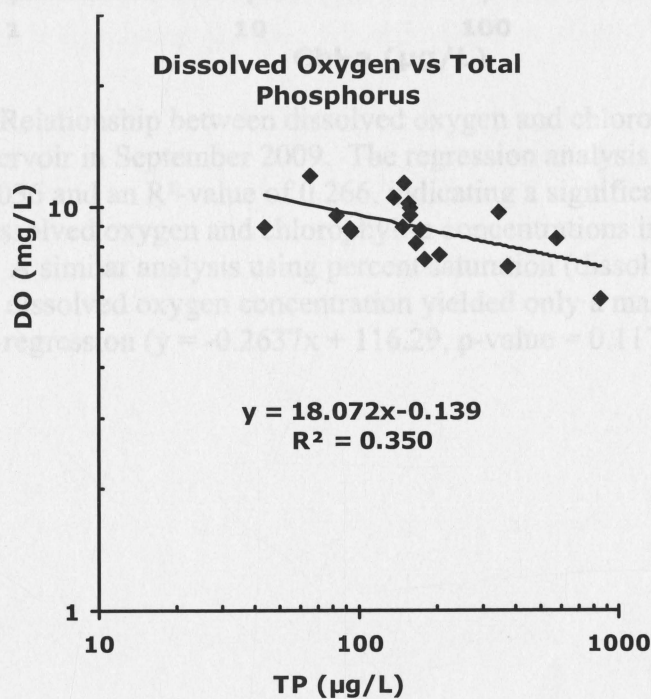
Figure 8. Linear regression comparing dissolved oxygen to total phosphorus for 17 sites in Cutler Reservoir. There was a significant correlation between the two parameters (p-value of 0.006 and R^2-value of 0.350), suggesting that dissolved oxygen levels are influenced by total phosphorus concentrations in Cutler Reservoir.
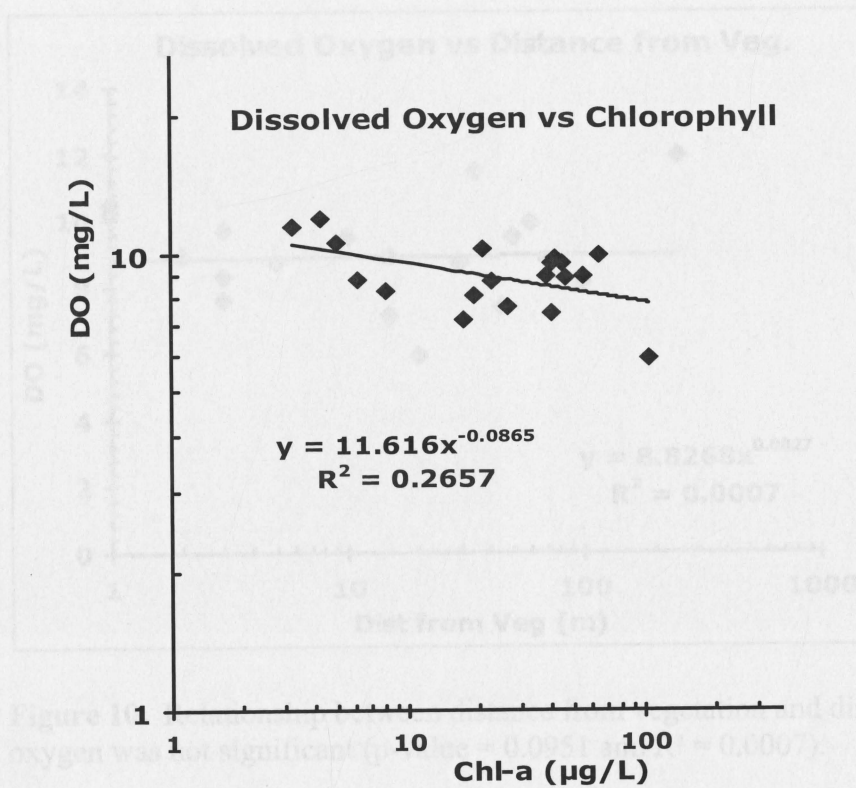
Figure 9. Relationship between dissolved oxygen and chlorophyll-a in Cutler Reservoir in September 2009. The regression analysis yielded a p-value of 0.036 and an R²-value of 0.266, indicating a significant relationship between dissolved oxygen and chlorophyll-a concentrations in Cutler Reservoir. A similar analysis using percent saturation (dissolved oxygen) rather than dissolved oxygen concentration yielded only a marginally significant regression (y = -0.2637x + 116.29, p-value = 0.117, and R² = 0.138).

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Dissolved Oxygen vs Distance from Veg.

$$y = 8.8268x^{0.0027}$$
$$R^2 = 0.0007$$

Figure 10. Relationship between distance from vegetation and dissolved oxygen was not significant (p-value = 0.0951 and R^2 = 0.0007).

Dissolved Oxygen Concentrations   Total Phosphorus Concentrations

Dissolved Oxygen Concentrations (mg/L)
- 0- 7.0
- 7.1 - 8.0
- 8.1 - 9.0
- 9.1 - 10.0
- 10.1 - 12

Area Enlarged

Utah

Total Phosphorus Concentrations (ug/L)
- 0 - 43
- 44 - 94
- 95 - 206
- 207- 574
- 575 - 850

0 0.5 1   2   3   4   Miles

N
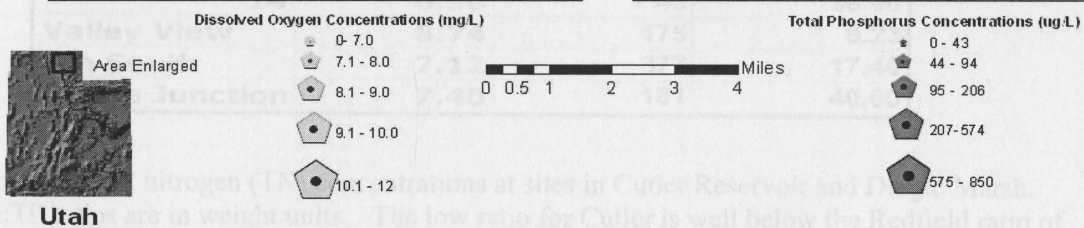
Figure 11. Dissolved oxygen concentrations for each site are represented on the left (yellow) and total phosphorus concentrations at each site are represented on the right (orange). This allows for a visual comparison between dissolved oxygen and total phosphorus concentrations on the large spatial scale used for this study. The size of each pentagon represents the corresponding concentrations for each site. At some sites, when total phosphorus pentagons (orange) are large, dissolved oxygen pentagons (yellow) are small (Sites 6, Cache Junction, and Sixth South). At other sites, total phosphorus pentagons are small and dissolved oxygen pentagons are large (Sites 14, 13, 12, 10, 7, 8, and 9). Findings confirm that when total phosphorus concentrations are high, dissolved oxygen levels are low, and vice-versa. This map was derived from a 1-meter aerial image taken in 2006, acquired at ftp://ftp.agrc.utah.gov/. The location map is a hillshade of Utah derived from a raster digital elevation model acquired from http://gis.utah.gov.

Table 1. Concentrations of dissolved oxygen, total phosphorus, and chlorophyll-a for each site sampled during this study.

Site	DO (mg/L)	TP (µg/l)	Chl a (µg/L)
1	8.90	149	55.30
1 B	10.20	157	20.90
2	9.90	159	65.60
3	7.60	206	26.70
3 B	9.70	347	42.50
4	8.10	167	19.20
5	8.30	575	8.12
6	5.90	850	105.00
7	11.90	65	4.33
8	10.50	137	5.02
9	11.45	150	3.30
10	8.70	94	22.90
11	9.50	159	45.70
12	9.50	82	41.00
13	8.80	93	47.30
14	8.90	43	38.90
Valley View	8.74	175	6.23
6th South	7.12	377	17.40
Cache Junction	7.40	181	40.60

Table 2. Total nitrogen (TN) concentrations at sites in Cutler Reservoir and Dingle Marsh. TN:TP ratios are in weight units. The low ratio for Cutler is well below the Redfield ratio of 6.8:1 (15:1 molar units), indicating that it is presently limited by nitrogen.

Location	Station	TN (ug/L)	
Cutler	1	966	
Cutler	2	831	
Cutler	3	732	
Cutler	3	914	
Cutler	4	803	
Cutler	5	1325	
Cutler	6	1757	
Cutler	11	753	
Cutler	14	843	
Cutler	Cache Junction	698	
Cutler	Valley View	2219	
Dingle	1	488	
Dingle	3	476	
Dingle	5	522	
Dingle	7	309	
			TN/TP
Cutler Mean		1076	3.7
Dingle Mean		449	12.9

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Appendix 1. Database containing results from the different parameters measured during this study, as well as date, time, and location (decimal degrees) of field work efforts at each site. Field and laboratory replicates are also recorded in this database. Sample collection times are Mountain daylight time (AM).
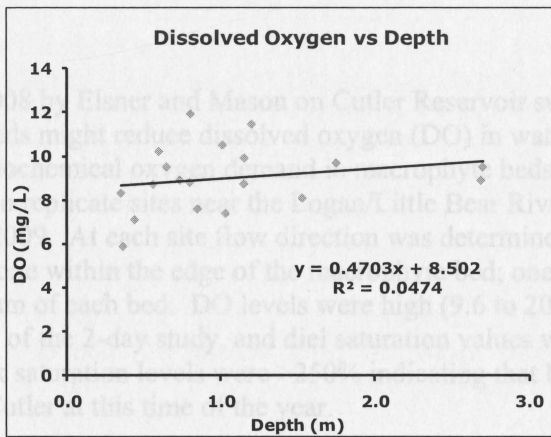
Site	Field Reps.	Labrotory Reps	Date Sampled	Lat.	Long.	Time	Max Depth (m)	TP (µg/L)	Chl-a (µg/L)	Distance from Veg.	Depth Measured (m)	DO (mg/L)	% saturated (DO) (Sealevel)	Temp. (C)	Specific Conductivity
1		a	23-Sep-09	N 41.79392	W 111.95872	5:23	0.73	149	55.300	2	0.20	8.90	89.2	15.97	499.2
1		b	23-Sep-09	N 41.79392	W 111.95872						0.50	8.80	88.4	15.60	498.6
1	B	a	28-Sep-09	N 41.79381	W 111.95879	7:30	0.60	157	20.900	1	0.40	10.20	103.2	15.60	534.0
1	B	b	28-Sep-09	N 41.79381	W 111.95879										
2		a	23-Sep-09	N 41.79429	W 111.95679	5:43	1.15	159	65.600	60	0.50	9.90	101.6	16.30	504.0
2		b	23-Sep-09	N 41.79429	W 111.95679			218			0.95	9.90	98.6	16.30	504.0
3		a	23-Sep-09	N 41. 78145	W 111.94292	5:57	0.85	206	26.700	3	0.50	7.60	71.2	16.10	506.0
3		b	23-Sep-09	N 41. 78145	W 111.94292						0.65	7.60	73.4	16.20	506.0
3	B	a	28-Sep-09	N 41.78149	W 111.94312	7:45	1.75	347	42.500	3	0.50	9.70	100.9	16.90	617.0
3	B	b	28-Sep-09	N 41.78149	W 111.94312						1.50	9.40	97.9	17.10	617.0
4		a	23-Sep-09	N 41.78005	W 111.94363	6:04	1.53	167	19.200	100	0.50	8.10	80.2	16.20	509.0
4		b	23-Sep-09	N 41.78005	W 111.94363						1.30	8.00	77.6	16.30	509.0
5		a	23-Sep-09	N 41.76044	W 111.93506	6:18	0.35	575	8.120	3	0.20	8.30	79.8	14.30	402.8
5		b	23-Sep-09	N 41.76044	W 111.93506										
6		a	23-Sep-09	N 41.76377	W 111.93391	6:26	0.36	850	105.000	20	0.20	5.90	56.5	13.00	682.0
6		b	23-Sep-09	N 41.76377	W 111.93391										
7		a	23-Sep-09	N 41.76310	W 111.93961	6:36	0.80	65	4.330	250	0.50	11.90	115.0	14.80	437.9
7		b	23-Sep-09	N 41.76310	W 111.93961							11.90	115.4	14.80	436.0
8		a	23-Sep-09	N 41.74843	W 111.95074	7:37	1.01	137	5.020	1	0.50	10.50	100.2	13.40	537.0
8		b	23-Sep-09	N 41.74843	W 111.95074						0.80	10.30	98.5	13.60	536.0
9		a	23-Sep-09	N 41.75455	W 111.94646	8:22	1.20	150	3.300	35	0.50	11.45	110.7	13.90	532.0
9		b	23-Sep-09	N 41.75455	W 111.94646						1.00	11.29	109.3	14.00	532.0
10		a	28-Sep-09	N 41.80267	W 111.95660	5:31	0.56	94	22.900	30	0.25	8.70	89.5	16.40	746.0
10		b	28-Sep-09	N 41.80267	W 111.95660										
11		a	28-Sep-09	N 41.80930	W 111.95727	5:58	0.65	159	45.700	50	0.25	9.50	98.2	16.80	643.0
11		b	28-Sep-09	N 41.80930	W 111.95727										
12		a	28-Sep-09	N 41.81762	W 111.95302	6:15	2.70	82	41.000	10	0.50	9.50	99.6	17.40	760.0
12		b	28-Sep-09	N 41.81762	W 111.95302						2.50	8.30	87.9	17.50	863.0
13		a	28-Sep-09	N 41.82541	W 111.95039	6:41	0.80	93	47.300	70	0.60	8.80	91.6	17.10	765.0
13		b	28-Sep-09	N 41.82541	W 111.95039										
14		a	28-Sep-09	N 41.82684	W 111.96070	6:54	2.69	43	38.900	15	0.50	8.90	92.7	17.30	795.0
14		b	28-Sep-09	N 41.82684	W 111.96070						2.50	8.70	91.1	17.30	800.0
Valley View		a	22-Sep-09	N 41.74529	W 111.95665	7:06	1.15	175	6.230	5	0.50	8.74	83.9	13.60	551.0
Valley View		b	22-Sep-09	N 41.74529	W 111.95665						0.95	8.62	83.4	13.60	547.0
6th South		a	22-Sep-09	N 41.71987	W 111.94033	5:11	0.44	377	17.400	15	0.25	7.12	68.7	13.60	
6th South		b	22-Sep-09	N 41.71987	W 111.94033			121							
Cache Junction		a	22-Sep-09	N 41.84448	W 112.00159	6:22	1.03	181	40.600	45	0.50	7.40	74.5	16.00	738.0
Cache Junction		a	22-Sep-09	N 41.84448	W 112.00159						0.80	7.31	74.7	16.10	737.0

Appendix 2. Linear regressions for depth, temperature, and specific conductivity plotted against dissolved oxygen concentrations at Cutler Reservoir. The p-values for each of these parameters are also listed below each graph. The dissolved oxygen concentrations were not significantly influenced by these factors.

Dissolved Oxygen vs Depth

$y = 0.4703x + 8.502$
$R^2 = 0.0474$

p-value = 0.3705

Dissolved Oxygen vs Temperature

$y = 0.1018x + 7.4277$
$R^2 = 0.0101$

p-value = 0.6819

Dissolved Oxygen vs Specific Conductivity

$y = 0.0003x + 8.8618$
$R^2 = 0.001$

p-value = 0.8985

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Effects of Emergent Macrophyte Beds on Diel Cycles of Dissolved Oxygen in Cutler Reservoir
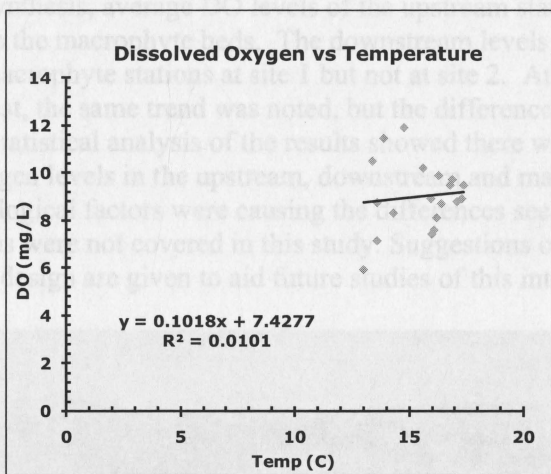
Justin Stout

ABSTRACT

A study performed in 2008 by Elsner and Mason on Cutler Reservoir suggested that emergent macrophyte beds might reduce dissolved oxygen (DO) in water close to the beds. To determine if biochemical oxygen demand in macrophyte beds influenced nearby oxygen levels two replicate sites near the Logan/Little Bear River inflow were analyzed in September 2009. At each site flow direction was determined and three sondes were deployed: one within the edge of the macrophyte bed; one~ 5 m upstream; and one ~5-m downstream of each bed. DO levels were high (9.6 to 20.7 mg/L) both during the night and day of the 2-day study, and diel saturation values were always greater than 100%. Peak saturation levels were >250% indicating that high oxygen levels may be toxic to fish in Cutler at this time of the year.

The data suggested that macrophyte beds reduced DO levels, but the effects were small. At times of peak photosynthesis, average DO levels of the upstream stations were 13-14% higher than those in the macrophyte beds. The downstream levels were lower than both the upstream and macrophyte stations at site 1 but not at site 2. At dawn, when oxygen levels were lowest, the same trend was noted, but the differences between sites were less than 1 mg/L. Statistical analysis of the results showed there were no significant differences between oxygen levels in the upstream, downstream and macrophyte beds. It is possible that other biological factors were causing the differences seen at peak DO levels. These interactions were not covered in this study. Suggestions of possible changes in the sampling design are given to aid future studies of this interaction.

Gil Rowley and Justin Stout at macrophyte bed study site

Introduction

The composition of habitat in Cutler Reservoir (hereafter Cutler) makes it an interesting area to study. Habitats range from shallow wetlands, to slow moving river. Each area creates a multitude of habitats for wildlife. The presence of emergent macrophytes in the system may have undesirable feedbacks by causing low levels of DO (TMDL, 2009). The accumulation of senesced macrophyte material creates an abundant supply of carbon for bacterial growth and respiration (Crumpton and Rose, 1996). This continuous respiration and the absence of photosynthesis at night, account for the diel fluctuations seen in most eutrophic systems. Such diel cycles of DO have been noted in Cutler by studies done for the 2009 TMDL (TMDL 2009) and previous studies. While these diel cycles are natural and to be expected, low levels of DO can have adverse affects on aquatic life, in particular fish species. Effects can range from increased physical stress to mortality. Each species of fish has different DO tolerances, but generally concentrations less than 3 to 5 mg/L for greater than 24 hours is detrimental to fish health.

In addition to the known effects of low DO, high concentrations of DO (>100%) can also have negative effects on fish and other aquatic life. For example; extremely high concentrations of oxygen (>300%) in a lake in California is thought to have been the cause of a trout and sunfish die-off (McKee and Wolf 1963). The EPA created guidelines in regards to dissolved gas supersaturation, recommending a total gas pressure of <110% of local atmospheric pressure (EPA 1986). This recommendation is recognized by most states, but fish deaths caused by supersaturation are more often attributed to excess nitrogen. Supersaturated levels of oxygen have been recorded in Cutler reservoir (TMDL 2009, Elsner and Mason 2008). Possible causes were thought to be caused by wind disturbances, precipitation events, boating activity near the sensors, and temperature change (TMDL 2009, Nurhayat et al 2006), but it is more like that high levels of photosynthesis in eutrophic Cutler Reservoir explain the high levels of DO .

Cutler Reservoir has large stands of emergent macrophytes (*Scirpus* spp. and *Typha* spp.). These stands slow and divert flow to entrap senesced plant material and sediments. This adds to the available carbon for bacteria to use. This increase in bacterial decomposition can cause striking differences between DO levels found in macrophyte beds, and the surrounding water (Price et al 1994). Crumpton and Rose (1996) found that emergent macrophyte beds in prairie potholes (Goose Lake Marsh) were almost continuously anoxic during the study period. They study also found that there were three zones around the macrophyte beds: 1) the zone of emergent macrophytes, providing significant underwater structure for the trapping of sediments and plant material which was consistently anoxic, 2) the transition zone, an area of sparse vegetation moving in to the open water which showed a slight increase in the amounts of DO, and 3) open water, with no structure to trap sediments or plant materials which was rarely anoxic and consistent high levels of DO. Low levels of DO in the macrophyte beds were attributed to shading by the plants, as well as an increase in the amount of decomposition that occurred at the site.

The presence of the macrophyte beds in Cutler coupled with shallow depths and high primary production, creates an ideal environment for oxygen consumption (TMDL 2009; Dodds 2002). To assess whether macrophyte beds might contribute to the low oxygen levels periodically observed in Cutler, I analyzed diel patterns of DO in macrophyte beds in both 'upstream' and 'downstream' positions. Although managers have focused on how nutrients have stimulated algal

production and their subsequent oxygen demand, the oxygen demands of macrophyte beds may also be an important factor in future management decisions for Cutler Reservoir.

METHODS

Study Sites

Several study sites were chosen utilizing Google Earth based on visual comparisons. Once in the field, comparisons were made and two sites were chosen based upon similar vegetation densities and flow patterns as suggested by Crumpton and Rose (1996). The sites were located 200-400 m north of the Valley View Bridge (Highway 30) where there was moderate flow. Water velocities and flow patterns at the sites were determined by using rhodamine dye. The dye was placed five meters upstream and we recorded the time needed to reach the macrophyte bed. Flow velocity at Site 1 was 0.40 m/sec; at Site 2 it was 0.52 m/sec. For both sites water flow in the macrophyte bed was stagnant and little dye reached the main flow over the time period of 30 minutes. Water depths at the edges of the beds were ~ 0.5 meters, so sondes could be placed in at similar depth. Contrary to expectations, a major portion of each macrophyte bed was a solid ground. This was likely a consequence of the macrophytes trapping sediments. Rather than water passing directly through the macrophytes, it passed along the margin of each bed where the emergent macrophyte (*Scirpus* spp.) was also present. The lack of direct flow through the majority of the bed may have limited the impact on oxygen concentrations.

Field Methods

Six In-Situ Troll 9500 sondes were deployed to measure DO for 48 hours. Prior to deployment, the RDO optical dissolved oxygen sensor of each sonde was calibrated using a two point calibration method following the In-Situ Incorporated methods. Each sonde was calibrated first at 100% oxygen saturation, using an aerated bath, and then calibrated again at 0% saturation using sodium sulfide (Na_2SO_3). After calibration the sondes were set to sample every 15 minutes, and placed in a single aerated tub for 2 hours for to allow me to determine if there were small discrepancies remaining after the calibration. The sondes were then placed in a protective box and transported to the sites. Each sonde was zip-tied inside a six inch diameter- 24 inch long HDPE pipe with $2^{5/8}$inch holes drilled every six inches on center. This allowed for flow to reach the sensors of the sondes. Pipes were suspended 20 cm below the surface of the water from a buoy that was attached to cinderblock. Sondes were deployed at each site in the following manner: A site was chosen near the middle (MD) of the emergent macrophyte bed where the depth was sufficient to keep the sonde submerged at all times. The same procedure was followed for both the up-stream (US) and downstream (DS) stations. At each sonde station a water depth was taken to ensure upstream and downstream sondes were placed in similar depths with an average depth of 1.8 meters. GPS coordinates (Table 1) at each site were taken using a Garmin 60 CSX handheld device. The coordinates were associated with the station name as being the middle sensor at the site (MD).

Table 1. Table giving Easting and Northing coordinates of the site and station locations on Cutler Reservoir. Coordinates taken *in UTM zone 12.*

SITE	STATION	EASTING	NORTHING
Site 1	Upstream (US)	420944.8	4622278
Site 1	Middle (MD)	420957.6	4622284
Site 1	Downstream (DS)	420972.8	4622283
Site 2	Upstream (US)	420840.6	4622250
Site 2	Middle (MD)	420856.6	4622252
Site 2	Downstream (DS)	420877.1	4622260

The sondes were deployed the morning of September 23rd, 2009 between the hours of 6:00 am and 8:00 am, and removed the morning of September 25th, 2009 during the hours of 7:00 am to 9:00 am. Upon removal the sondes were placed again in an aerated bath, to measure instrument drift.

Data Analysis

Data downloaded from each sonde was compiled into a master database. A graph was created for each site consisting of the three stations at each site (Appendix 2.) Data from the sondes was normalized from post field calibrations. These corrections were done by taking the average concentration for each sonde during the time the sondes were in the aerated bath. The deviation of each sonde from the grand average of all the sondes was then subtracted from the field values. The average of one sonde was -0.32 mg/L different than the grand average, thus by adding -.32 to the whole data set from that sonde, the error in sonde one was corrected. The high and low levels of DO were then selected for comparison. To avoid errors due to choosing a single high or low value, it was necessary to create an average maximum and minimum DO for each sonde. This was accomplished by taking approximately 5 data points to the left and right sides of the local maximum and minimum values and averaging the DO levels for approximately 2.5 hours, thus giving one value as the maximum level of DO seen each day and the one minimum near dawn. These values were then analyzed by ANOVA for significant differences.

Temperature data from each site was also graphed to show comparisons between each station and site. No statistical analysis of temperature and DO was preformed as it was used solely for visual comparison between stations.

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RESULTS

There were differences in the maximum values of DO at each site (Figure 1). At Site 1 a 5 mg/L difference in oxygen concentration was measured at the upstream and downstream sondes during periods of maximum photosynthesis. This was not as prominent at Site 2, but the upstream and middle sondes did show higher concentrations than the downstream sonde. At night, when respiration was the dominant process, all values at both sites decreased and become essentially equal. Temperature displayed a diel cycle and there was little difference in temperatures between stations (Figure 1).

A 2-way ANOVA of maximum DO levels utilizing the two sites and 3 positions at each site, indicated that neither site nor position were significant factors influencing DO (F1,2, p = 0.54, p = 0.333). One way ANOVAs of the maximum and minimum values yielded similar results (Table 2 and Appendix 3).

Table 2. Showing the results of the three single factor ANOVAS run to compare sites and stations. Results were not significant.

Values Analyzed	Result of ANOVA
First high levels of DO, contrasting sites 1 and 2	$F_{2,3}$ of 9.55, and p value of .24
Low levels of DO, contrasting sites 1 and 2	$F_{2,3}$ of 9.55, and p value of .19
Second high levels of DO, contrasting sites 1 and 2	$F_{2,3}$ of 9.55 and p value of .19

It is possible that using an ANOVA to analyze the data was not appropriate for this data set. When the data used to calculate the ANOVA were plotted showing the variance between sites, it appeared that the variances were not equal. This may cause the resulting p-values to be inaccurate. Figure 2 shows the three variance plots for each of the ANOVA tests. Using the results from the ANOVA, there is no statistical power to state that there is a significant difference between DO levels in the macrophyte beds and the open water. Regardless of this result, my opinion is that there was a difference in the DO levels during the peak levels of DO. The exact cause of these differences and the cause for the DO levels to become near equal at expected low level times are issues for future study.

DISCUSSION

Results of this study provided only limited support from previous work on this subject that suggests that emergent macrophyte beds can reduce nighttime oxygen levels. An earlier study done on a wetland found a distinct difference in DO between the habitats of open water and emergent macrophyte beds (Crumpton and Rose 1996). Even though my study shows that this relationship is not as prevalent at our two study sites, it cannot be negated as a possible factor influencing DO concentrations in Cutler Reservoir. Although the limited number of independent sites (2) did not yield significant differences, it should be noted that at Site 2 differences in DO of up to 5 mg/L were noted between sondes that were <35 m apart. Differences at Site 2 were less, but it is possible that there were different flow patterns that could have diverted water from the macrophyte beds. We had no way of assessing whether flow paths moved directly from the macrophyte beds towards the lower sonde. These mixing patterns could explain the differences between the two sites. If a second set of sites were chosen, replication of the processes would offer stronger statistical power and perhaps different results. The statistical analysis of the study was based upon weak assumptions on equality of variance between the sites. A more in-depth analysis of the data collected may show differences that were not detected in the ANOVA test. Although thousands of data points were collected, only three average values for each maximum and minimum were used to determine if there was a difference between the sites. Possible use of non-averaged data points would offer more power to evaluate true differences.

It was interesting to note that maximum DO levels were extremely high, reaching 250% (20.7 mg/L) of saturation. It is likely that high nutrient levels, combined with high solar radiation, allowed for increased photosynthetic rates. It was also notable that although the oxygen concentrations declined considerably at night, they never dropped below 9.5 mg/L. This suggests that during this time of year low oxygen levels would not be a problem. However, the high supersaturated oxygen levels may pose a danger to fish in Cutler Reservoir.

This study now sets the stage for subsequent studies on plant-water interactions. Due to the time constraints of this study, not all parameters affecting the DO in or near these macrophyte beds could be examined. In future studies it would be beneficial to understand the amount of photosynthesis that is occurring in and near the beds, as well how the macrophyte beds influence the production of oxygen through shading. It is possible that the differences seen in the DO levels are due to shading by the plants (Dodds 2002), and increased consumption in the beds. A study of sediment oxygen demand of the macrophyte beds versus open water would possibly demonstrate factors that were not taken into account in this study (Price et al 1994). This holds with the original hypothesis that the emergent macrophyte beds are depleting the amounts of DO in the system. However, we would have expected to see this trend throughout the entire process, but at night the oxygen levels at the different sites converged, suggesting that the differences in respiration are minimal. Additionally, because water was moving through the channels at about 0.5 m/sec, it is quite possible that the "upstream" water had been influenced by macrophytes further up the channel.

Other data that could be collected and used in analysis are: (adapted from Crumpton and Rose 1996): 1) Plant litter in each of the different sites. This can account for decomposition at each of the sites. 2) Secchi depths at each site. If light penetration was greater than the depths of the macrophyte beds this could explain the increased DO in the deeper water where more photosynthesis was occurring. Finally, subsequent studies should leave the sondes in place for a longer period of time. This would ensure that the differences seen during the day and those taken at night are truly representatives of the area. Without a replication differences in low DO could not be determined.

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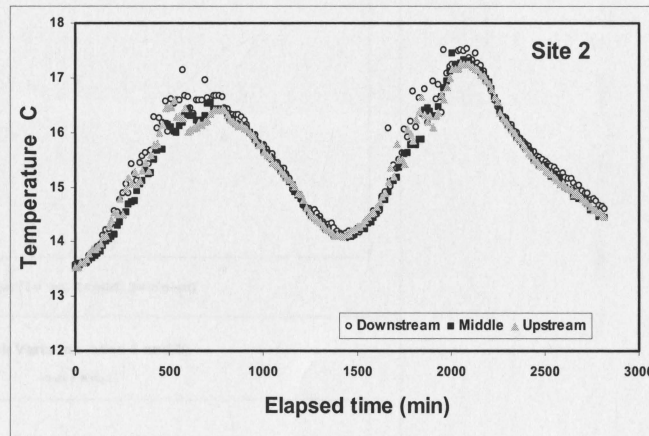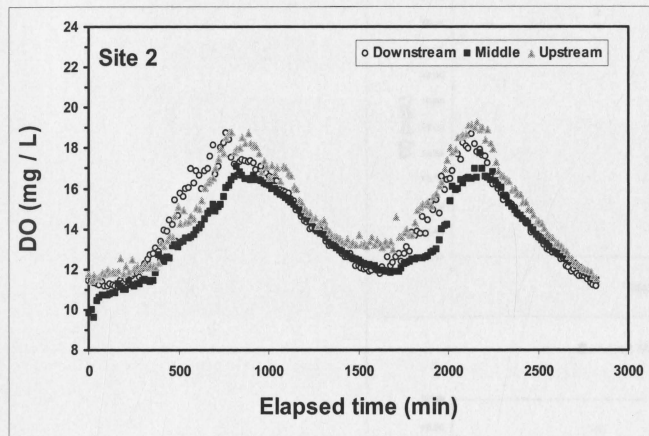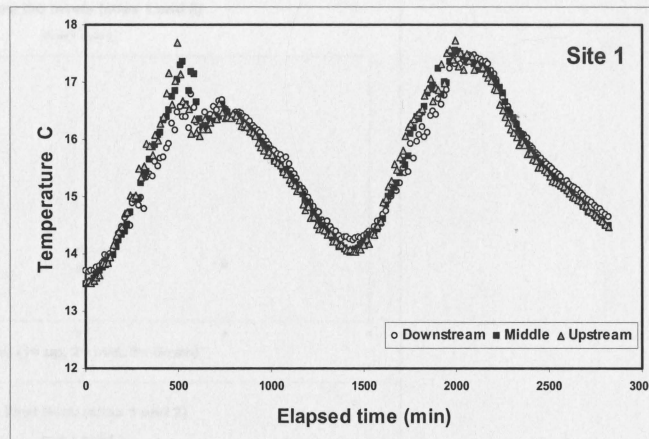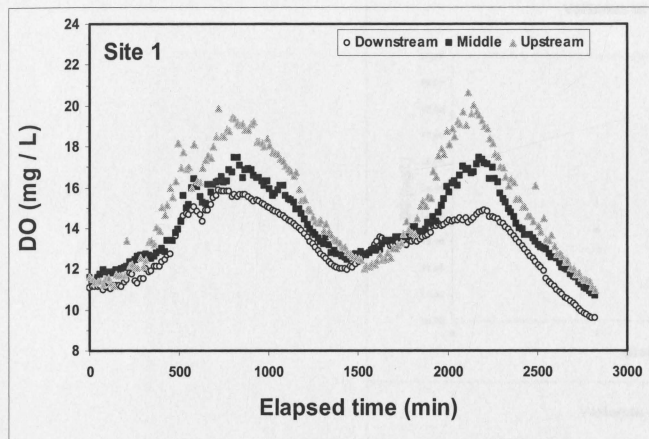
Figure 1. Oxygen (left) and temperatures (right) recorded at two sites in Cutler Reservoir from September 23-25, 2009. Note the differences in DO at sites upstream, downstream and in the middle of the macrophyte beds during the mid-day maxima, and the very large diel fluctuations. Also note the similar temperature values at each station for the entire time, and the relatively small fluctuations from approximately 12 to 18°C.

Figure 2. Graphs showing the dissolved oxygen concentrations between sites at each of the peaks and single low during the entire study period. Stations are equated with numbers. 1= upstream, 2= middle and 3= downstream. Unequal variances between sites makes it difficult to justify the usage of an ANOVA for analysis.

APPENDICES

Appendix 1:Map of Sites

Appendix 2: Map of sites with sampling stations.
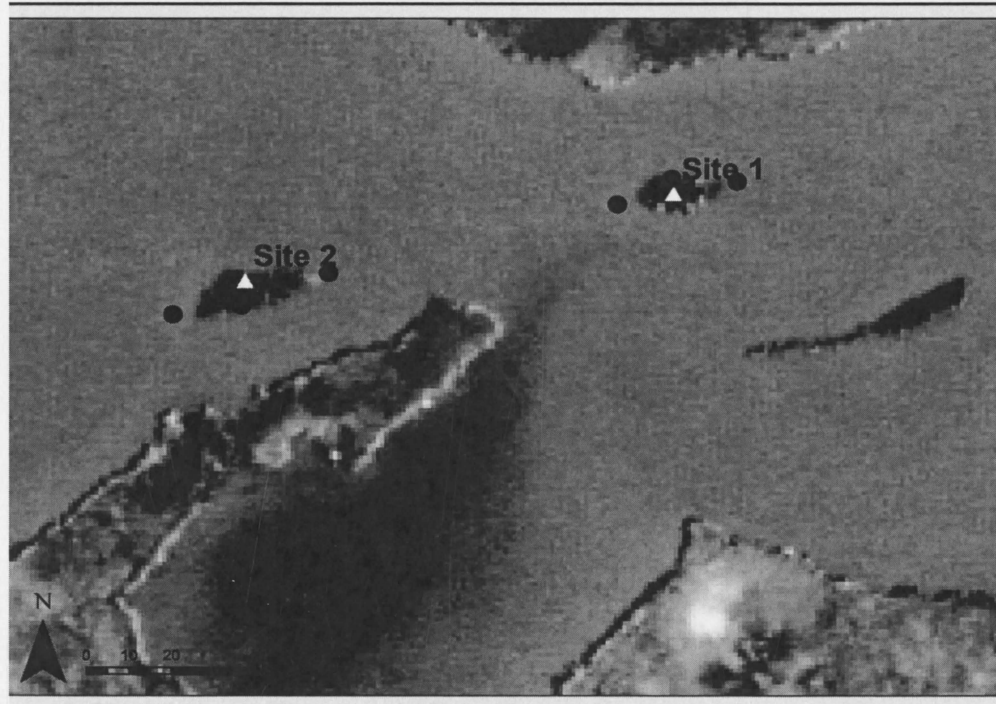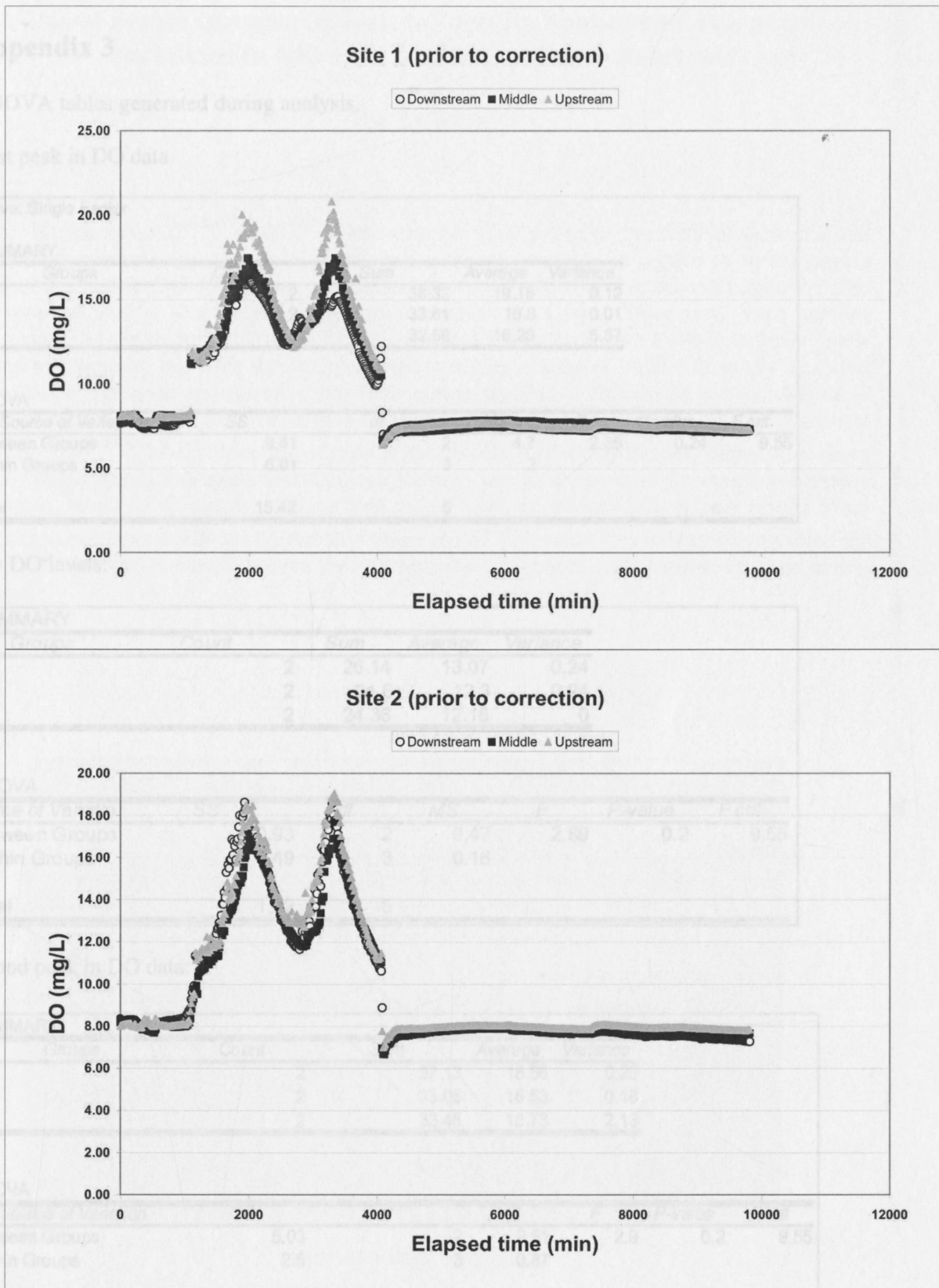
Figure 2. Plot of the raw data at each site.

Appendix 3

ANOVA tables generated during analysis.

First peak in DO data

Anova: Single Factor					
SUMMARY					
Groups	Count	Sum	Average	Variance	
US	2	38.32	19.16	0.12	
MD	2	33.61	16.8	0.01	
DS	2	32.56	16.28	5.87	
ANOVA					
Source of Variation	SS	df	MS	F	P-value	F crit
Between Groups	9.41	2	4.7	2.35	0.24	9.55
Within Groups	6.01	3	2			
Total	15.42	5				

Low DO levels:

SUMMARY					
Groups	Count	Sum	Average	Variance	
US	2	26.14	13.07	0.24	
MD	2	24.6	12.3	0.24	
DS	2	24.36	12.18	0	
ANOVA					
Source of Variation	SS	df	MS	F	P-value	F crit
Between Groups	0.93	2	0.47	2.89	0.2	9.55
Within Groups	0.49	3	0.16			
Total	1.42	5				

Second peak in DO data:

SUMMARY					
Groups	Count	Sum	Average	Variance	
US	2	37.13	18.56	0.29	
MD	2	33.06	16.53	0.18	
DS	2	33.45	16.73	2.13	
ANOVA					
Source of Variation	SS	df	MS	F	P-value	F crit
Between Groups	5.03	2	2.51	2.9	0.2	9.55
Within Groups	2.6	3	0.87			
Total	7.62	5				

Use of Stable Isotope Analysis to Identify Spatial and Temporal Variations in Nitrogen Loading to Cuter Reservoir

Dan Lamarra

Abstract

Stable isotope values ($\delta^{15}N$ and $\delta^{13}C$) were determined by collecting the surficial sediments and a paleolimnetic core for analysis of spatial and temporal variations in regards to anthropogenic nutrient loading to Cutler Reservoir. Samples were collected from five sites throughout Cutler Reservoir as well as four supplemental sites throughout the Cache Valley area. Four samples from higher in the watershed (Dingle Marsh) were collected to compare a site with lower levels of nutrient loading, but with similar hydrologic qualities. Samples were statistically analyzed using linear regression and student t-tests to determine significant differences and correlations to land use area and project sites within the Cutler Reservoir watershed. Results of the student t-test indicate that $\delta^{15}N$ levels in the lower parts of Cache valley were significantly higher than those from Dingle marsh indicating that Cutler Reservoir acts as a terminus for enriched nitrogen effluents. Analysis of upstream human land usage, however, yielded insignificant results when assessing correlations between human land usage and $\delta^{15}N$ levels. This indicates that a there are more complex factors contributing to the enrichment of sediments in Cutler than explored in this project.

Dingle Marsh – Mud Lake

Introduction

Increased nitrogen loading has been implicated in eutrophication (an increase in productivity) on a worldwide basis. Much of this loading is attributable to the growing human population along the world's aquatic corridors; a significant component of this nitrogen input is from human sewage effluent and livestock activities (Costanzo et al.. 2001). The identification and regulation of non-point and point source nutrient inputs is increasingly important to the health and aesthetic of aquatic resources. Understanding the distribution and fate of human-based nitrogen has become a priority for watershed management at local scales and a focus for water quality criteria at national scales (NRC 1994, as cited by Luecke & Mesner). The use of stable isotopes derived from the sediments of lakes and rivers can be a useful tool in identifying the spatial and temporal variations as well as sources of carbon and nitrogen in aquatic ecosystems.

The elements carbon and nitrogen each have two naturally occurring forms, the first of these forms which are called low masses, each have equal numbers of protons and neutrons. Low mass atoms are denoted as ^{12}C and ^{14}N. These forms make up approximately 99% of all naturally occurring carbon and nitrogen atoms (Fry 2006). The less abundant forms of nitrogen and carbon are referred to as the atom's high masses. These atoms contain an extra neutron and can be denoted as ^{13}C and ^{15}N, because of the extra neutron the atoms have a higher atomic weight. These isotopes do not degrade, and thus are referred to as stable isotopes. Isotope values are typically represented with the notation "δ" or the Greek lower-case delta. Delta signifies the difference of measurement in regards to a standard and can be found for individual isotopes in the following equation (Fry 2006).

$$\delta^H X = [(R_{sample} / R_{standard} - 1)]*1000$$

Where X is the atom in question, H is the heavy isotope mass, and R is the ratio of heavy isotopes to light isotopes in the element. The δ notation involves a final multiplication by 1000; this multiplication amplifies very small differences measured between samples and standards. (Fry 2006) The final result is a value of δ in the units of ‰ or parts *permil*, which has the ability to make very small (but important) differences between samples more understandable.

Isotopes function as naturally occurring markers that can be used to track the circulation of elements. Isotopes trace ecological connections at many levels, from individual microbes to a whole landscape (Fry 2006). Nitrogen is commonly used in ecological research as a tracer in food webs; for example, species tend to become more enriched with $\delta^{15}N$ the higher they are in the trophic structure of the ecosystem. Likewise, the lower in the trophic structure the organism resides the lower the $\delta^{15}N$ signatures will be. Human waste has often been connected with an

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elevated $\delta^{15}N$ signature which is different than that of the natural ecosystem (Costanzo *et al.* 2001) which makes $\delta^{15}N$ a useful tool in identifying sources of anthropogenic inputs.

$\delta^{13}C$ can be used as a tracer for the flow of carbon through a system. Sediment $\delta^{13}C$ generally reflects the $\delta^{13}C$ of the most abundant primary producers in the environment (Gu *et al.* 1996). Lake and river sediments signature will reflect the general source of carbon, whether it be terrestrial or aquatic in origin. The ratio of $^{13}C / ^{12}C$ in terrestrial carbon is often higher than that of aquatic carbon due to the greater structural carbon present in the source (Gu *et al.* 1996). High $\delta^{13}C$ values are typically found in oligotrophic (unproductive) lakes where primary production is low and systems rely on allochthonous (external) sourced carbon. Eutrophic systems are highly productive and $\delta^{13}C$ values should reflect a lower value indicating that carbon flow in the system is autochthonous (internal) in nature (Gu *et al.* 1996).

In this study we attempted to indentify the spatial and temporal variations of the stable isotopes $\delta^{13}C$ and $\delta^{15}N$ in the Cutler Reservoir watershed, in hopes of identifying nitrogen inputs to the reservoir.

Study Sites

In September 2009 benthic surface sediments were collected from 13 study sites in the Cutler Reservoir watershed for spatial analysis of the stable isotopes $\delta^{13}C$ and $\delta^{15}N$ (Map 1). Cutler Reservoir was the primary area of interest and was considered the central location for the study. Dingle Marsh (USFWS Bear Lake National Wildlife Refuge) served as a comparison site being hydrologically similar but spatially higher in the watershed, with expected lower nutrient loading.

Cutler Reservoir is located six miles west of Logan, Utah, at an elevation of 4,407 feet. Cutler impounds the waters of the Bear, Logan, and Little Bear Rivers as well as other small tributaries. Cutler Dam, built in 1927, is operated by Rocky Mountain Energy (formerly PacifiCorp) and is used to provide water for agricultural use in Box Elder County, as well as power generation (DEQ 2009).

Dingle Marsh is located on the north end of Bear Lake, 40 miles north east of Logan Utah and 12 miles south of Montpellier Idaho, at an elevation of 5,970 feet. Water flows from the Bear River to Dingle marsh and then into Bear Lake. Water moves into Bear Lake during spring runoff; subsequently, during the summer irrigation season, water from Bear Lake is drained in a canal along the margin of Dingle marsh, and back into the Bear River. The Bear River then flows through Idaho and eventually into Cutler Reservoir. Dingle Marsh samples were collected

in the region of Mud Lake, which is morphologically very similar to the central area of Cutler Reservoir.

Another water source for Cutler Reservoir is the Logan River. The head waters of the Logan River begin in the relatively pristine Bear River mountain range north east of Logan, Utah in an area known as Franklin Basin. The Logan River travels approximately 40 linear miles through Logan Utah to its terminus at Cutler Reservoir. The Logan River discharges approximately 255,930,021 m^3 of water per year to Cutler Reservoir, second only to the Bear River (DEQ 2009).

The Little Bear River is the third tributary of Cutler Reservoir and was also analyzed. Hyrum Reservoir is located on the Little Bear River just south of Hyrum City, at an elevation 4,664 feet. The overall surface area of the Reservoir is 438 acres^2, it is used as water storage for agriculture and recreation (DEQ Hyrum TMDL). The Little Bear River, which flows from the reservoir, travels approximately 10 linear miles to its terminus at Cutler Reservoir. With a discharge of approximately 84,835,007 m^2 of water per year into the reservoir (DEQ 2009)

Field and Laboratory Methods

Field methods—A 6.3-cm diameter Wildco gravity core was used at each site. In Cutler Reservoir two surface sediment cores were taken at each site for replicates to generate greater statistical power: In Dingle marsh a single core was taken at each site. Samples taken from areas less than one meter in depth were typically taken by pushing the corer into the sediments; samples taken deeper than one meter were extracted by gently lowering the gravity corer so as to minimize disturbance of the surficial sediments. Samples, whether prepared on land or boat, were taken by slicing the top centimeter from the length of the core. Samples were then placed into a plastic zip lock bag, marked with their GPS location, and subsequently frozen for lab analysis.
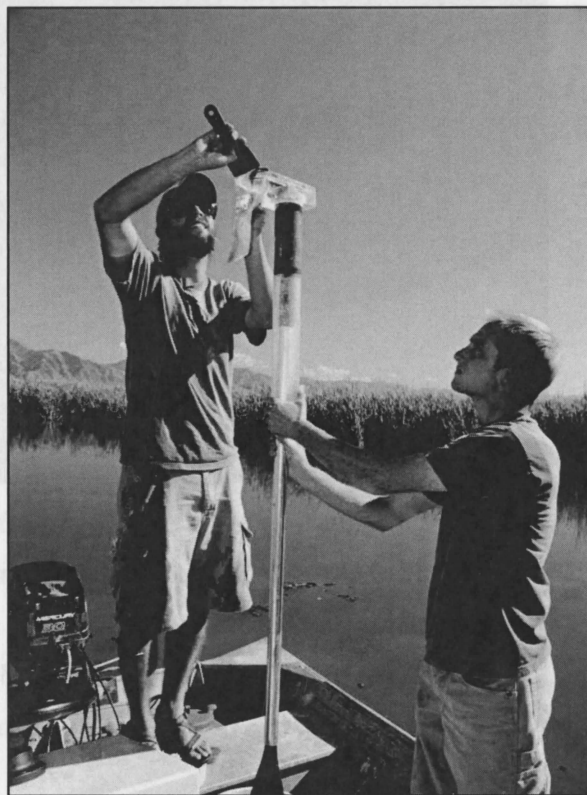
For analysis of temporal variation of stable isotopes $\delta^{13}C$ and $\delta^{15}N$ three 20 centimeter paleolimnetic cores were extracted from Cutler Reservoir. These cores were sealed, GPS marked,

and taken back to the lab for sub-sampling and packaging. Only one core was used for dating and analysis of isotopes.

Lab Methods— Sediment samples were dried at 70°C for approximately 24 hours to remove all moisture. Samples were homogenized to pass through a 350-μm mesh and placed into 5x9 mm silver capsules and hydrated; samples were then placed into a desiccator with a beaker of concentrated hydrochloric acid for 10 hours to remove inorganic carbonates which can skew the results of the organic carbon analysis (Harris et al. 2001). Samples were dried for a final time at 70°C for12 hours to remove any residual acid and moisture from the samples. Dried samples in the silver tins were then placed in 5x9 mm tin containers and re-incapsulated and taken to the Utah State University Stable Isotope laboratory where they were analyzed for δ^{13}C and δ^{15}N. The laboratory method utilizes a continuous-flow direct combustion and mass spectrometry with a Europa Scientific SL-2020 system

One of the three paleolimnetic cores was chosen based on visual interpretation of possible sedimentation lines and a lack of mixing. The core was then sliced into sections with a thickness of 0.75 centimeters. Each section was placed into a plastic bag, and frozen until further analysis could be conducted. Each section from the core was segmented into multiple aliquots; for this experiment we used two sub-samples from the whole. One sub-sample was weighted, dried at approximately 70°C for 24 hours then weighed again to acquire the wet to dry weight mass ratio. These samples were sent to Mycore Scientific Incorporated, for analysis of the [210]lead content, for dating. Aliquots designated for the δ^{13}C and δ^{15}N were subjected to the same protocol and methods outlined above in the previous paragraph.
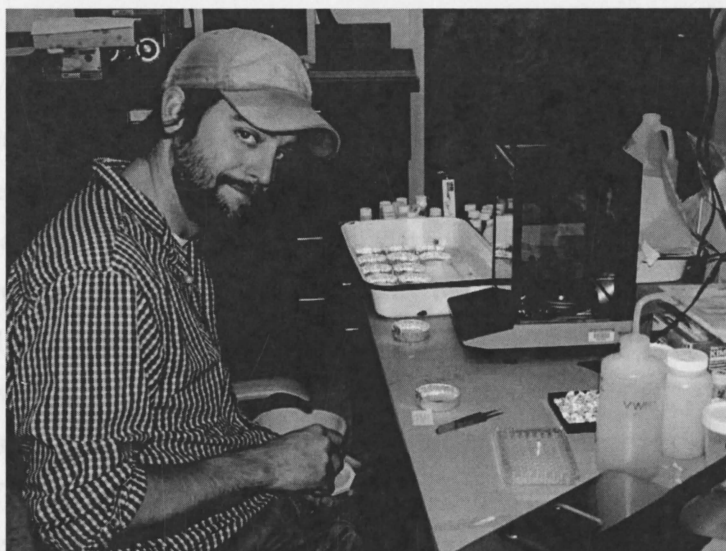
GIS Methods— Geospatial analysis was conducted using ESRI ArcMap 9.3, projected in the North American Datum 1983, UTM zone 12 north. Base, land-use, and hydrologic maps were created using public access Vector and Raster files. Sites were edited and assigned both δ^{13}C and

δ^{15}N values to reflect spatial magnitude difference between individual sites (see map 2 & 3). Sites were isolated individually and all upstream human land-use areas was clipped to a buffer width of 0.5 mile around all main stem inputs as well as major tributaries (see map 4). Upstream land uses were then segmented to three types; total human augmented land use area, urban land use area, and agricultural land use area. Areas were transformed to manageable dimensions (acres) and exported for analysis.

Analysis

Samples were statistically analyzed using Microsoft Excel 2007. Linear regression models were run on each of the stable isotope verse land-use plots to determine if significant correlations and possible causalities existed. Tertiary and Cutler reservoir sites were grouped together and compared with Dingle Marsh sites with a student t-test for the purposes of statistically determining if the values of δ^{13}C and δ^{15}N varied enough to support our conclusions.

Results & Discussion

There were large differences in δ^{15}N values, ranging from δ2.55 at the pristine 3rd Dam site, to δ 3.0–4.8 at the Dingle Marsh sites, to high values of 5.8–8.6 in Cutler and Hyrum Reservoirs and in the lower Logan River (Figure 1; Tables 1-2). Surprisingly, the highest δ^{15}N value was recorded in the Logan River at the10th West site, suggesting that sources in the city of Logan, or from the Blacksmith Fork River contribute substantial enriched ^{15}N. δ^{15}N values in Cutler Reservoir were significantly higher in Cutler Reservoir than in Dingle Marsh (P = 0.001; two-tailed t-test; Table 5). However, when all of the Cutler Reservoir δ^{15}N values were compared against all of the tertiary sites within the valley, there was no significant difference (P > 0.05). This is undoubtedly the result of grouping disparate sites in the Tertiary group.

Sediment carbon in Cutler Reservoir was significantly isotopically lighter than in Dingle Marsh (-27.6 vs. -24.4; P = 0.003). This suggests that there is a greater organic carbon contribution from phytoplankton in eutrophic Cutler Reservoir than in Dingle Marsh. However, when all the data from the sites were pooled, neither the tertiary sites nor the Dingle Marsh sites proved to be significantly different (t-tests, respective P-values of 0.25 and 0.12). This indicates that there are not large differences in the carbon source signatures between Cache Valley and sites located outside the valley.

Cutler δ^{13}C and δ^{15}N variations were also tested using linear regression analysis in regards to upstream human augmented land use area (Figure 2 & 3). Regression values showed no significant correlation between δ^{15}N and any upstream land usage, with a r^2 value of (.003).

The purpose of this analysis was to determine if upstream land use area could be used as an indicator of agricultural and urban influences on $\delta^{15}N$. The results of the linear regression indicate that no one type of source; either non-point or point source can be attributed to the elevated $\delta^{15}N$ values observed within Cutler Reservoir.

In contrast to the ^{15}N data the linear regression analysis for $\delta^{13}C$ versus human augmented land use area was highly significant (P = 0.024; Figure 3). Land use explained 49% of the variance of differing carbon inputs to the ecosystems of interest.

The results from the core dating were inconclusive. This is probably the effect of mixing in the upper layers of the sediments at the site where to core was taken, and the lead-210 dating was minimally useful. There were also no significant differences in isotopic values for $\delta^{13}C$ or $\delta^{15}N$ through the length of the core, but there were trends in the amount of nitrogen and carbon in different strata (see report of Abbott and Collins, page 90)

Conclusion

The surficial sediments of Cutler Reservoir are highly enriched with the $\delta^{15}N$ stable isotope. This can be an indication of a strong anthropogenic influence on the system. During this project I have found that this enrichment cannot be attributed to the total amount of upstream human land usage or any single inflow to the reservoir, but is most likely a combination of multiple different sources, point and non-point. Cutler Reservoir and the tertiary sites within Cache Valley are significantly more enriched than Dingle Marsh, the comparison site higher in the watershed. This suggests that sites with higher anthropogenic inputs are steadily enriched along and within the watershed as one moves from areas of low to high anthropogenic impacts.

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Figure 1: Carbon and nitrogen isotopic ratios in the Cutler Reservoir drainage. Red squares represent all sites located at Dingle Marsh, blue diamonds denote Cutler Reservoir sites, and green triangles represent tertiary sites. Data for the individual sites in Dingle Marsh, as well as the average for that area are shown. Tertiary sites are 3rd Dam on the Logan River, Hyrum Reservoir, and Logan River at 1000 W bridge crossing.

Figure 1: Carbon and nitrogen isotopic ratios in the Cutler Reservoir drainage. Red squares represent all sites located at Dingle Marsh, blue diamonds denote Cutler Reservoir sites, and green triangles represent tertiary sites. Data for the individual sites in Dingle Marsh, as well as the average for that area are shown. . Tertiary sites are 3rd Dam on the Logan River, Hyrum Reservoir, and Logan River at 1000 W bridge crossing.

Figure 2: Relationship between $\delta^{15}N$ values of surficial sediments at the study sites and human-augmented land use area.
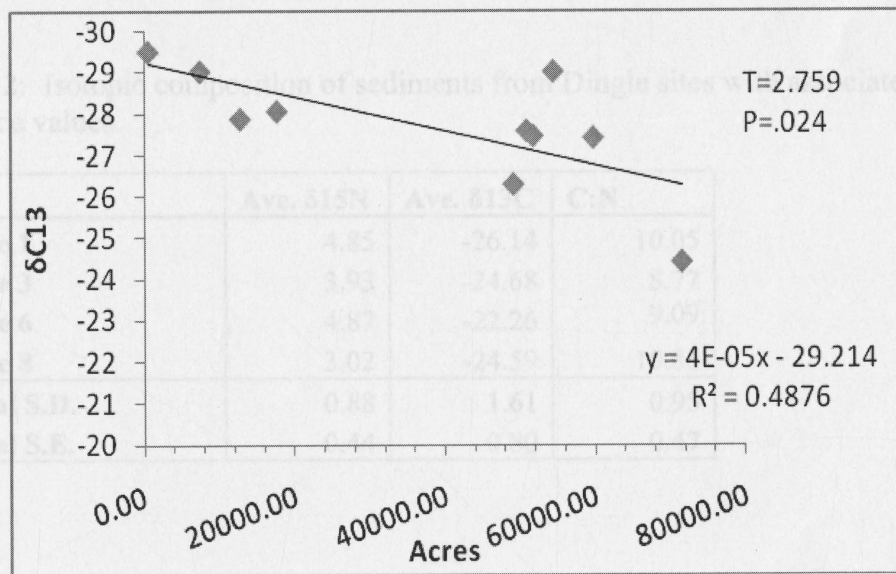
Figure 3: Relationship between $\delta^{13}C$ values in the surficial sediments and human-augmented land use area.

Table 1. **Isotopic composition of surface sediments in** Cache Valley sites with associated variance values.

Site	Ave. δ15N	S.D. δ15N	S.E. δ15N	Ave. δ13C	S.D. δ13C	S.E. δ13C	Ave. C:N	S.D. C:N	S.E. C:N
6th South	6.47	0.45	0.32	-28.06	0.17	0.12	9.76	0.24	0.17
Benson Marina	7.27	0.06	0.05	-29.02	0.14	0.10	8.37	0.15	0.10
Cache Junction	6.75	1.39	0.98	-27.41	0.08	0.05	8.93	0.06	0.04
Swift Slough	7.92	0.00	0.00	-27.46	0.83	0.59	9.76	0.25	0.18
Valley View	8.24	0.41	0.29	-26.29	0.28	0.20	10.63	0.76	0.54
Core	5.82			-27.58			9.42		
Hyrum Res.	7.75	0.06	0.04	-29.03	0.20	0.14	8.33	0.02	0.01
Logan river 10 west	8.62	1.91	1.35	-27.87	0.13	0.09	12.42	1.42	1.00
Logan River 3rd Dam	2.55	0.17	0.12	-29.51	0.01	0.01	14.92	0.47	0.33

Table 2: Isotopic composition of sediments from Dingle sites with associated spatial variance values.

Site	Ave. δ15N	Ave. δ13C	C:N
Dingle 1	4.85	-26.14	10.05
Dingle 3	3.93	-24.68	8.77
Dingle 6	4.87	-22.26	9.09
Dingle 8	3.02	-24.59	10.86
Spatial S.D.	0.88	1.61	0.95
Spatial S.E.	0.44	0.80	0.47

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Table 3: Project sites displayed with correlated upstream land-usage & values. HALUA=human augmented land use area

Site	H.A.L.U.A. (acres)	Urban (acres)	Ag (Acres)	δ15N	δ13C
3rd dam	575.71	575.71	0.00	2.55	-29.51
6th south	17807.86	15069.62	2738.24	6.47	-28.06
10th west	12882.23	12126.64	755.59	8.62	-27.87
Benson	54539.21	9.13	54530.08	7.27	-29.02
Swift Slough	51842.07	0.00	51842.07	7.92	-27.46
Valley View	49232.51	43931.30	5301.21	8.24	-26.29
Core	50980.82	45679.61	5301.21	5.82	-27.58
Hyrum Res.	7522.97	6350.24	1172.73	7.75	-29.03
Cache Junction	59860.11	58866.29	993.82	6.75	-27.41
Dingle	71699.03	50735.36	20963.67	4.17	-24.41

Table 4: *Output of two sample unequal variance t-test for the nitrogen isotopic composition Cache Valley sites verses Dingle Marsh.*

	Cache	Dingle
Mean	6.820	4.168
Variance	3.359	0.778
Observations	9	4
Hypothesized Mean Difference	0	
Df	11	
t Stat	3.519	
P(T<=t) one-tail	0.002	
t Critical one-tail	1.796	
P(T<=t) two-tail	0.005	
t Critical two-tail	2.201	

Table 5: *Output of two sample equal variance t-test for nitrogen isotopes in Cutler Reservoir Sites verses Dingle Marsh.*

	Cutler	Dingle
Mean	7.08	4.17
Variance	0.83	0.78
Observations	6.00	4.00
Pooled Variance	0.81	
Hypothesized Mean Difference	0.00	
df	8.00	
t Stat	5.00	
P(T<=t) two-tail	0.001	
t Critical two-tail	2.31	

Site Locations

0 2 4   8   12   16
Miles

Map 1: *Dots indicate project site locations throughout the Cutler Reservoir watershed.*

Spatial Variation δ15N

Legend

Delta_n15

- ○ 2.55
- ○ 3.02
- ○ 3.93
- ○ 4.85 - 4.87
- ○ 5.82
- ○ 6.47 - 6.75
- ○ 7.27
- ○ 7.75 - 7.92
- ○ 8.24 - 8.62

0 1.25 2.5   5   7.5   10
Miles
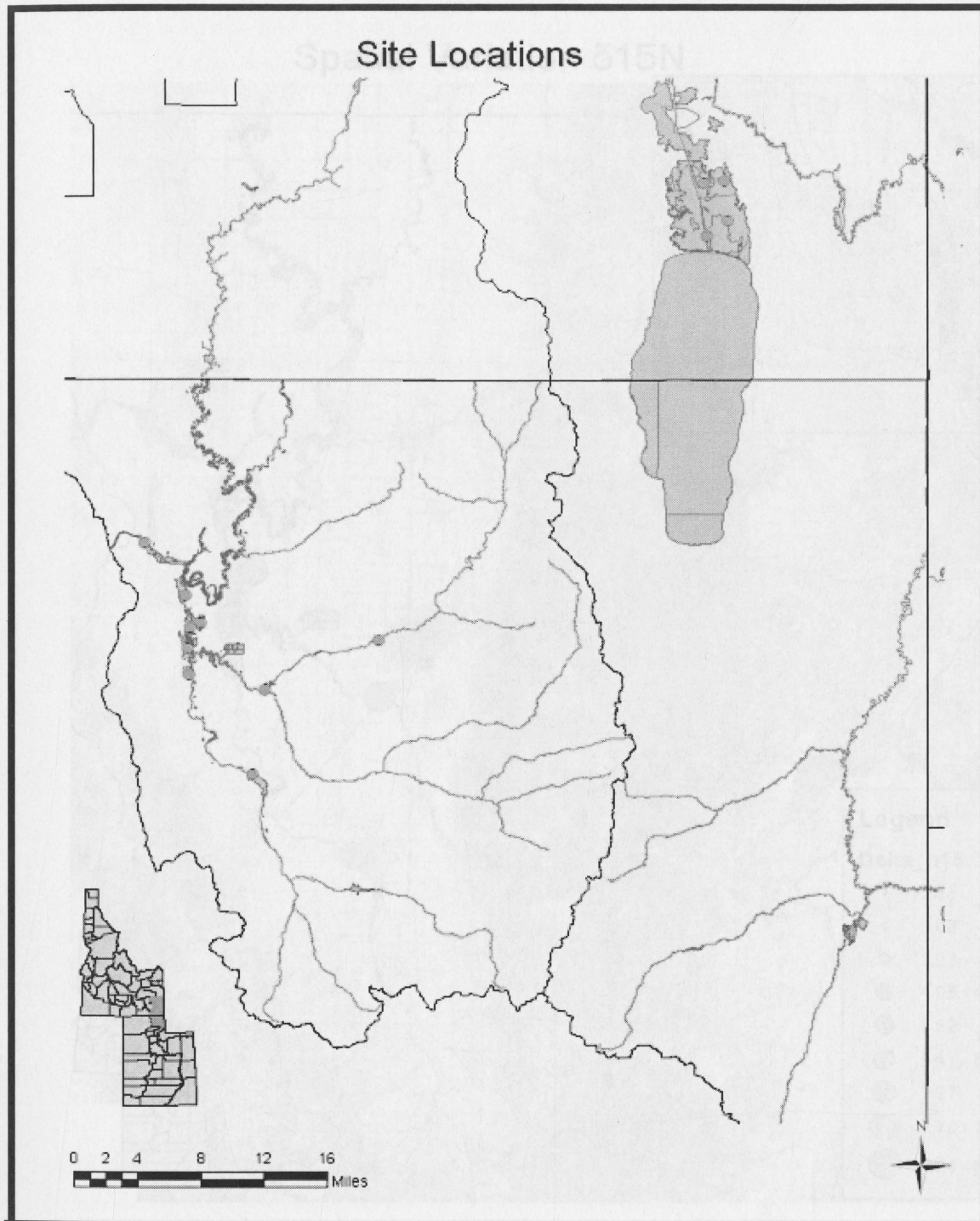
Map 2: *Dots denote project sites and associated δ15N values.*

Spatial Variation δ13C

Legend

Delta_c13

- ∘ -29.51
- ◇ -29.03 - -29.02
- ○ -28.06
- ○ -27.87
- ○ -27.58
- ○ -27.46 - -27.41
- ○ -26.29 - -26.14
- ○ -24.68 - -24.59
- ○ -22.26

0 1.25 2.5   5   7.5   10
Miles
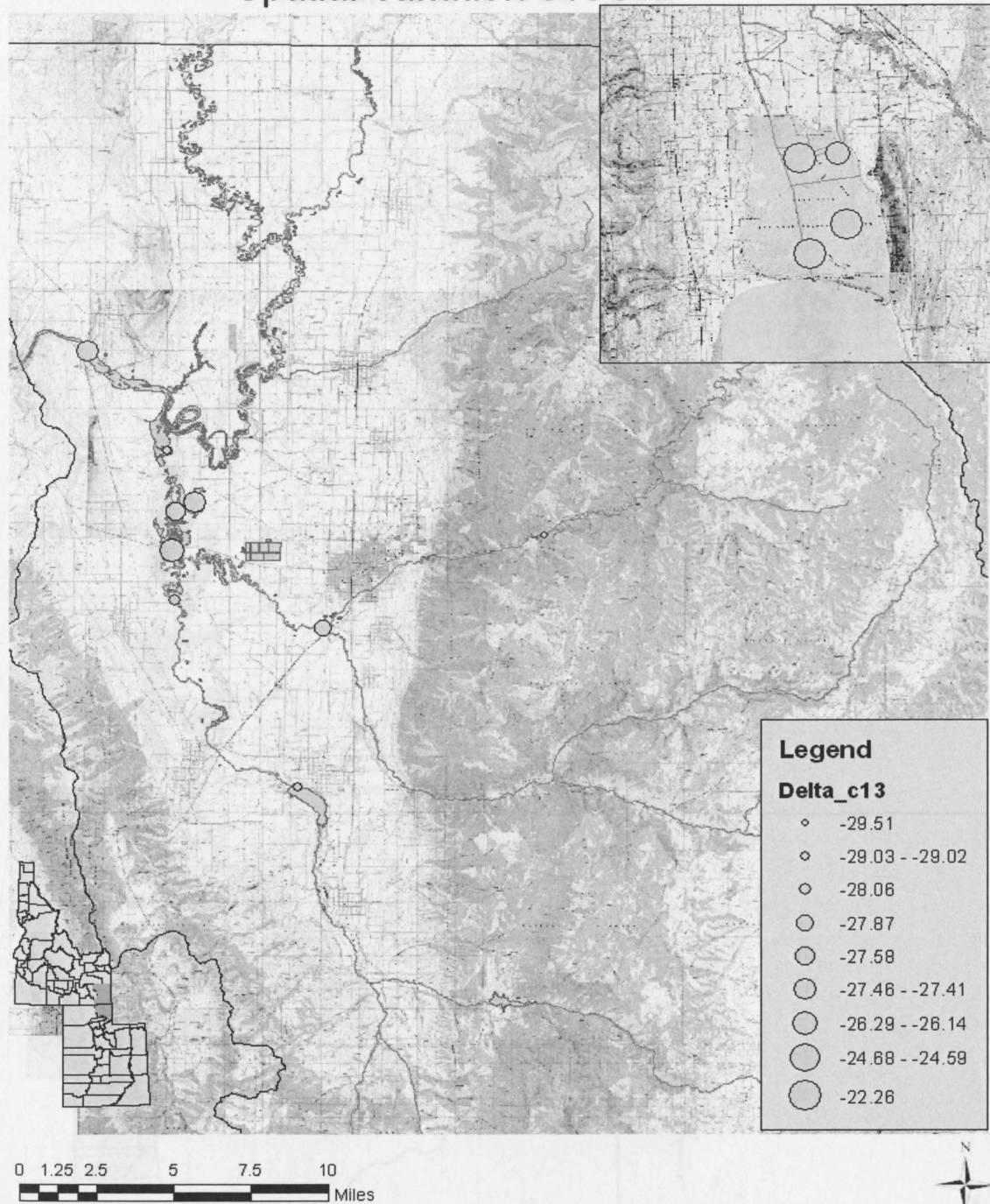
Map 3: *Dots denote project sites and associated* $\delta^{13}C$ *values*

Human Land-Use Area

Abstract

Stable isotope analysis can be used to gain a greater understanding of the effects of anthropogenic nitrogen in wetland food webs. Samples from multiple trophic levels were collected at both Cutler Reservoir and Dingle marsh, a less-polluted reference site, in September, 2009. All samples were processed and sent to Utah State University Department of Biology Stable Isotope Laboratory for analysis of $\delta^{15}N$ to $\delta^{13}C$. Analysis indicated significantly higher levels of $\delta^{15}N$ in Cutler Reservoir than in Dingle marsh. This was likely due to several direct sources of anthropogenic nitrogen which include the Logan City sewage treatment plant, dairies and other industrial and agricultural sources. $\delta^{13}C$ levels in the data were significantly higher in Cutler than in Dingle (ANOVA, p=0.02). Comparison of $\delta^{15}N$ levels measured this year with other published results for Cutler indicated that there has been no significant change over the past four years. Future studies may require a larger number of samples to make results statistically stronger and relevant. A longer sampling period would also be necessary to determine significant changes in anthropogenic nitrogen in Cutler Reservoir.

0   3.5   7   14   21   28
Miles

N

Map 4: *Dots denote project sites throughout the Cuter Reservoir watershed. Green shaded areas show ½ mile buffer area of total human land use area.*

Using stable isotopes to assess effects of anthropogenic nitrogen in Cutler Reservoir

Colin Cook

Abstract

Stable isotope analysis can be used to gain a greater understanding of the effects of anthropogenic nitrogen in wetland food webs. Samples from multiple trophic levels were collected at both Cutler Reservoir and Dingle marsh, a less-polluted reference site, in September, 2009. All samples were processed and sent to Utah State University Department of Biology Stable Isotope Laboratory for analysis of $\delta^{15}N$ to $\delta^{13}C$. Analyses revealed significantly higher levels of $\delta^{15}N$ in Cutler Reservoir than in Dingle marsh. This was expected due to several direct sources of anthropogenic nitrogen which include the Logan Wastewater Treatment plant, dairies and other industrial and agricultural sources. $\delta^{15}N$ levels in the biota were significantly higher in Cutler than in Dingle (ANOVA; p=0.02). Comparison of $\delta^{15}N$ levels measured this year with other published results for Cutler indicated that there has been no significant change over the past four years. Future studies should utilize a larger number of samples to make results statistically stronger and relevant. A longer sampling period would also be necessary to determine significant changes in anthropogenic nitrogen in Cutler Reservoir.
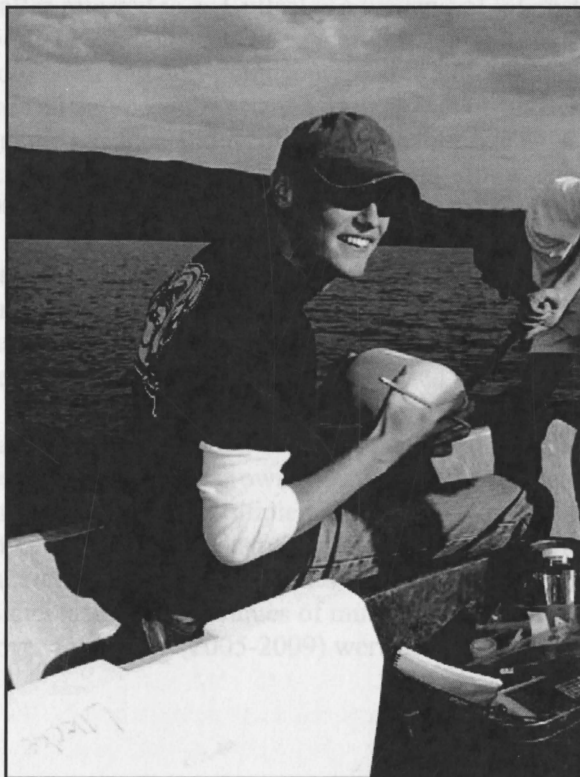
Colin Cook Processing Samples at Dingle Marsh

Introduction

Wetlands often have very dynamic and complex food-webs. Using stable isotope analyses to better understand varying trophic interactions is a common technique. Heavy isotopic ratios such as $^{13}C{:}^{12}C$ and $^{15}N{:}^{14}N$ are a useful way to determine food sources in food-webs (Dodds, 2002). Comparing the ratios of $\delta^{15}N$ ($^{15}N{:}^{14}N$) to $\delta^{13}C$ ($^{13}C{:}^{12}C$) helps to clarify and separate different food-web components (Dodds, 2002). These different ratios occur in the food webs because of a process known as fractionation. Fractionation occurs when isotopes become enriched or depleted as they travel through different organisms in the food web (Peterson and Fry 1987). Different organisms within a food web may also react differently to anthropogenic inputs. For example, Cole *et al.* (2004) found that macrophytes had significantly higher $\delta^{15}N$ values when wastewater input increased.

While these isotopic ratios are helpful in determining food web interactions, they can also serve to identify potential anthropogenic sources. McCelland and Valiela (1998) demonstrated the impacts that anthropogenic nitrogen can have on food webs within shallow coastal estuaries. Luecke and Mesner (unpublished) hypothesized that anthropogenic nitrogen may have similar impacts to freshwater ecosystems, and they showed increasing concentrations of the heavy isotope of nitrogen in lower reaches of the Little Bear River than in upper sections. Teranes and Bernasconi (2000) found that $\delta^{15}N$ values will increase in sediment over time when nitrogen is added to the system continuously. Because anthropogenic nitrogen from sources such as agriculture and wastewater effluent tends to have higher $\delta^{15}N$ signatures, these changes can be tracked over time and space.

Cutler Reservoir (hereafter referred to as Cutler) is a wetland of interest because it is considered eutrophic (TMDL Report, 2008). Nutrients of concern include nitrogen and phosphorus which enter Cutler through both point and non-point sources. Agriculture, industry, and human wastewater treatment are all sources of nutrient enrichment. The nutrient of concern in my research was anthropogenic nitrogen and its effect on the food-web. To assess potential changes of $\delta^{15}N$ in Cutler Reservoir I compared data collected in my study with results provided by past studies (Budy *et al.* 2005, Randall, 2008).

Dingle Marsh (hereafter referred to as Dingle) is another wetland system which serves as a transport unit for irrigation water into Bear Lake. Dingle has limited sources of anthropogenic nitrogen and is considered a non-eutrophic wetland. Despite being non-eutrophic, Dingle Marsh shares many similarities with Cutler Reservoir and was used as a reference location for this study.

Objectives of this research were to: (1) look at $\delta^{15}N$ values from several trophic levels in both Cutler and Dingle including: periphyton growing on macrophytes, seston, zooplankton, macrophytes, macro-invertebrates, and multiple fish species and (2) determine any differences that exist between the two wetland ecosystems. Due to significant anthropogenic nitrogen sources present at Cutler, but not at Dingle, the $\delta^{15}N$ values were predicted to be higher than those of Dingle. Any observed increases in $\delta^{15}N$ values of multiple trophic levels (seston, zooplankton, and largemouth bass) over four years (2005-2009) were also analyzed.

Methods

Study sites and Field Methods

Several sites were selected at both Cutler and Dingle for sampling. Periphyton, seston, zooplankton, and macrophytes were collected in Cutler on September 24th 2009 between Benson Marina and the Valley View highway. GPS coordinates are given in Appendix 1. Macro-invertebrates were also collected from four separate sites on the same day as part of a qualitative study on macro-invertebrates present in Cutler (Leonard and Marrett, this report). Multiple fish species were collected from a single site at Benson Marina on the same day. Replicate samples for periphyton, seston, and macrophytes samples were collected from three sites at Cutler in October 2009. Dingle samples (periphyton, seston, and zooplankton) were collected from four different sites on September 29th, 2009. Sample sites for macro-invertebrates included the following habitats at both Cutler and Dingle; emergent-macrophyte bed, submerged-macrophyte bed, rocky bank, and partially submerged log. Due to administrative restrictions at Dingle, fish could not be collected.

Similar field methods were used to collect all samples at Cutler and Dingle to limit differences in sampling error. Seston (phytoplankton + detritus + bacteria) was collected from surface water at each site with a 500-ml sample jar and stored in a refrigerator on return to the laboratory. Periphyton was collected by cutting emergent macrophytes at their base and placing them in a large sample jar filled with surface water. The jar was shaken vigorously for approximately two minutes to remove epiphytes. Macrophytes were then removed and stored in separate containers. A 0.5-m diameter, 153-μm mesh zooplankton net was used to collect zooplankton samples. Macro-invertebrates were collected with a kick net according to methods described by Leonard and Marrett (this report). Fish samples were collected from one sweep with a 10-m seine net and placed in a sample jar filled with 90% alcohol. All samples were placed in a cooler until dried.

Lab processing

Seston and periphyton samples were filtered onto 25-mm diameter Gelman AE filters approximately 24-96 hours after collection. This deviated slightly from common procedure which requires processing within 24 hours. Due to time constraints, not all samples met this requirement. Zooplankton samples were poured onto a 250-μm sieve and then rinsed into a sample jar using distilled water. Any large non-zooplankton organic particles present were removed with tweezers to avoid errors in the isotopic signature. These samples were then filtered onto 25-mm diameter Gelman AE filters. All samples were then dried for 48-96 hours at 50-70°C (Voss et al. 2000). Samples were then fumigated in a desiccator with 100 ml of 12 M HCL acid for approximately one hour to remove any potential contamination from inorganic carbonates. The samples were then removed from the desiccator and dried for another 24 hours. Excess filter not containing any sample was cut away and removed before weighing and

encapsulating. All filter samples were placed into 12x5 mm sized tin capsules and compressed as recommended by the UC Davis Stable Isotope Facility.

Macrophyte, macro-invertebrate, and fish samples were first identified and labeled. Macrophyte samples were identified as hardstem bulrush (*Schoenoplectus acutus*) and were cut into small pieces. Fish samples contained the following juveniles; largemouth bass (*Micropterus salmoides*), speckled dace (*Rhinichthys osculus*), and yellow perch (*Perca flavescens*). In the case of the largemouth bass, muscle tissue samples were taken from below the dorsal fin. All perch and dace samples were left whole. All fish, macrophytes, and macro-invertebrate samples were then dried for 48-96 hours at 50-70°C. Small macro-invertebrate samples were encapsulated whole, while fish and macrophytes samples were ground into a homogeneous powder using a mortar and pestle. All samples were then weighed and encapsulated in 4x6-mm sized tin capsules (Grey, 2006). Encapsulated samples were placed in separated wells of a cell plate.

All samples were sent to the Stable Isotope Lab at the Utah State University Department of Biology. Encapsulated samples were analyzed for $\delta^{15}N$ and $\delta^{13}C$ by a qualified technician with a continuous-flow direct combustion mass spectrometer using a Europa Scientific SL-2020 system.

Results

Comparison of $\delta^{15}N$ values between Cutler and Dingle:

When compared together on separate scatter plots, there are noticeable differences in the isotopic signatures between the samples taken at Cutler and Dingle (Figure 1.). The $\delta^{15}N$ values in Cutler Reservoir were considerably higher than those in Dingle (Figure 1). Due to a limited number of samples, a 2-way ANOVA was run using only the samples with sufficient replicates collected at both Cutler and Dingle (seston, periphyton, and zooplankton). The ANOVA for $\delta^{15}N$ indicated that location was highly significant ($p < 0.05$), that taxa were nearly significant ($p = 0.07$) and the interaction between location and taxa was highly significant ($p < 0.05$). Sediment samples were also collected and run for $\delta^{15}N$ (see Lamarra, this report). Because the sediment data was comparable between Cutler and Dingle, it was also used in this report. The difference in $\delta^{13}C$ was also statistically significant (ANOVA; p-value=0.047).

The $\delta^{15}N$ values for fish were very high in Cutler Reservoir (near +16). These results suggest that the fish rely primarily on benthic invertebrates whose $\delta^{15}N$ values were +10 - +12, rather than on zooplankton (+5). Fractionization between trophic levels is typically +3-5, so it is unlikely that the fishes sampled were relying directly on zooplankton as a prey base.

In general, the samples taken from Dingle were not as depleted in $\delta^{13}C$ as those taken from Cutler (Figure 1). However, the 2-way ANOVA for $\delta^{13}C$ for the three abundant taxa types indicated that neither location nor taxa were significantly different (respective p values of 0.13 and 0.23). Sediment δ13C values were also less depleted in Dingle than in Cutler Reservoir (see Lamarra, this report).

Changes in $\delta^{15}N$ values in Cutler over the past four years

Unfortunately, only values for seston, zooplankton, and largemouth bass were available from all three years (2006, 2008, and 2009) of data. ANOVA tests showed no significant differences

between the three sets of data (all p-values >>0.05). Linear regressions were also run between $\delta^{15}N$ values and time. In all instances, no significant relationship was found (Figure 3).

Budy, P., Dahle, K., and Thiede, G. P. 2007. An evaluation of the fish community of Cutler
        Reservoir and the Bear River above the reservoir with consideration of the potential for
        future fisheries enhancement. 2006 Annual Report to the Utah   Department of
        Environmental Quality, Division of Water Quality. 75 pages.

Discussion

Comparison of $\delta^{15}N$ values between Cutler and Dingle

Cole, M.L., Valiela, I., Kroeger, K. D., Tomasky, G. L., Cebrian J.,Wigand, C., McKinney, R. A.,

It is apparent that there are significant differences between the $\delta^{15}N$ values in Cutler and Dingle. The difference appears throughout the food web in its entirety and on individual levels. There are several methods that could be improved in this study that may show even stronger differences. The number of samples and replicates taken at each study should be increased in order to have stronger statistical power. Due to lack of samples processed from Dingle, the food web from that area cannot be adequately examined.

Again, the greatest limitation in the study was the lack of samples and replicates taken. There are multiple levels of the food web which are not represented because a lack of available data. It is recommended that a more data should be taken in order to give truly significant results. Another possible factor influencing the relationship between time and $\delta^{15}N$ is simply the protocol followed for obtaining and processing samples. Budy *et al.* (2007) did not fumigate seston or zooplankton samples to remove inorganic carbonates while this study and the Randall (2008) study did.

McClelland, J. W., and Valiela, I. 1998. Changes in food-web structure under the influence of
        increased anthropogenic nitrogen inputs to estuaries. Marine Ecology. 168:259-271

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        Ecology and Systematics 18: 293 320.

Randall, J. 2008. Comparative limnological analysis of Cutler Reservoir and Dingle Marsh with
        respect to eutrophication. In 2008 Project Report Comparative Limnological Analysis of
        Cutler Reservoir and Dingle Marsh with Respect to Eutrophication. pp. 57-70.

Teranes, J.L., and Bernasconi, S.M. 2000. The record of nitrate utilization and productivity
        limitation provided by δ15N Values in lake organic matter-a study of sediment trap and
        core sediments from Baldeggersee, Switzerland. Limnology and Oceanography. 45:801-
        813.

Utah Department of Environmental Quality UDEQ. 2008. Middle Bear River and Cutler
        Reservoir TMDLs.

Voss, M., B. Larsen, M. Leivuori and h. Vilias. 2000. Stable isotope signals or   eutrophication
        in Baltic Sea sediments. J. of Mar. Sys. 25:287-298.

References

Budy, P., Dahle, K., and Thiede, G. P. 2007. An evaluation of the fish community of Cutler Reservoir and the Bear River above the reservoir with consideration of the potential for future fisheries enhancement. 2006 Annual Report to the Utah Department of Environmental Quality, Division of Water Quality. 75 pages.

Cole, M.L., Valiela, I., Kroeger, K. D.,Tomasky, G. L., Cebrian,J.,Wigand, C., McKinney, R. A., Grady, S. P., and da Silva., M. H. C. 2004. Assessment of a delta N☐15 isotopic method to indicate anthropogenic eutrophication in aquatic ecosystems. Journal of Environmental Quality. 33:124☐132.

Dodds, W. 2002. *Freshwater ecology: Concepts and environmental applications*. Academic Press. Pg 396-398.

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Peterson, B.J., and B. Fry. 1987. Stable Isotopes in Ecosystem Studies. Ann. Rev. Ecology and Systematics 18: 293☐320.

Randall, J. 2008. Comparative limnological analysis of Cutler Reservoir and Dingle Marsh with respect to eutrophication. In 2008 Project Report Comparative Limnological Analysis of Cutler Reservoir and Dingle Marsh with Respect to Eutrophication. pp. 57-70.

Teranes, J.L., and Bernasconi, S.M. 2000. The record of nitrate utilization and productivity limitation provided by δ15N Values in lake organic matter-a study of sediment trap and core sediments from Baldeggersee, Switzerland. Limnology and Oceanography. 45:801-813.

Utah Department of Environmental Quality UDEQ. 2008: Middle Bear River and Cutler Reservoir TMDLs.

Voss, M., B. Larsen, M. Leivuori and h. Vllius. 2000. Stable isotope signals or eutrophication in Baltic Sea sediments. J. of Mar. Sys. 25: 287-298.

Figure 1. Mean carbon and nitrogen isotopic values for different trophic levels in Cutler Reservoir and Dingle marsh. Individual values and standard deviations are included in Appendix 2.

Figure 2. Comparison of ^{13}C and ^{15}N isotopic signatures in Cutler and Dingle on four separate trophic levels; periphyton, seston, zooplankton, and sediments. The sediment data is from Dan Lamarra (this report).
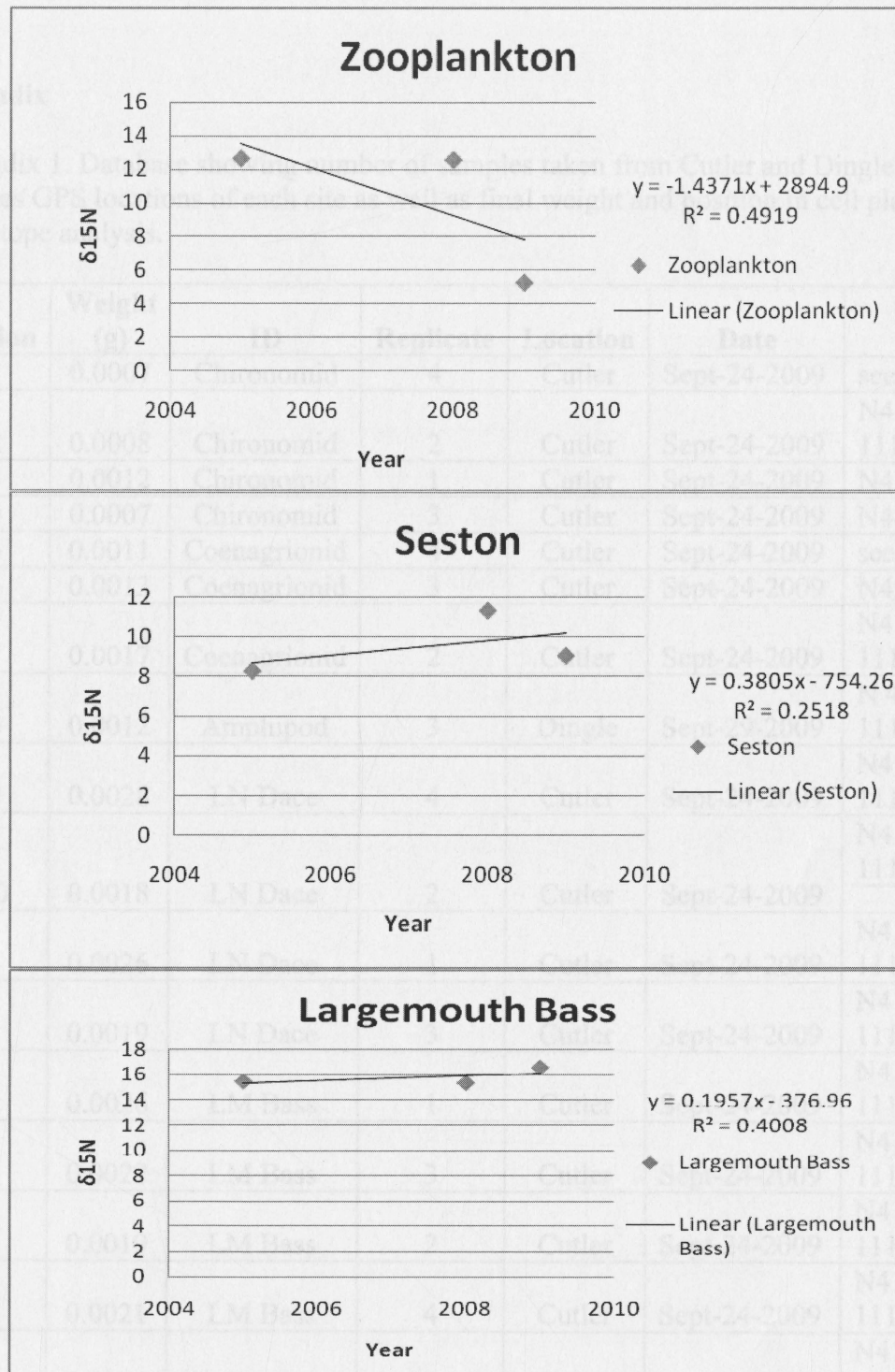
Figure 3. Relationship between time (x-axis) and mean $\delta^{15}N$ (y-axis) levels in Cutler Reservoir. A linear regression was run for each trophic level, R^2 values and line equations are included.

Appendix

Appendix 1. Database showing number of samples taken from Cutler and Dingle.
Includes GPS locations of each site as well as final weight and position in cell plate sent
for isotope analysis.

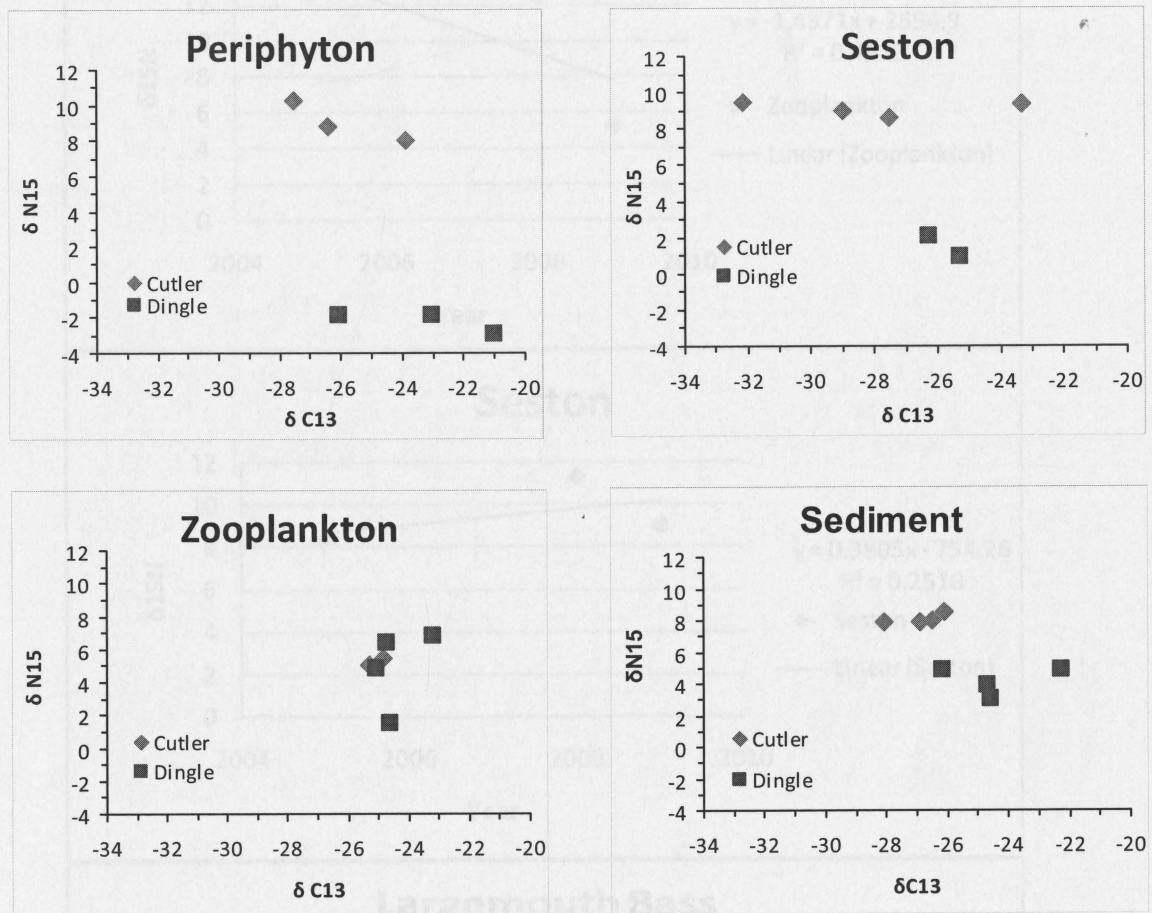| Position	Weight (g)	ID	Replicate	Location	Date	GPS
A1	0.0007	Chironomid	4	Cutler	Sept-24-2009	see Leonard and Marrett
A2	0.0008	Chironomid	2	Cutler	Sept-24-2009	N41.78175 W-111.943017
A3	0.0012	Chironomid	1	Cutler	Sept-24-2009	N41.77526 W-111.9476
A4	0.0007	Chironomid	3	Cutler	Sept-24-2009	N41.7831 W-111.94591
A5	0.0011	Coenagrionid	4	Cutler	Sept-24-2009	see Leonard and Marrett
A6	0.0013	Coenagrionid	3	Cutler	Sept-24-2009	N41.7831 W-111.94591
A7	0.0017	Coenagrionid	2	Cutler	Sept-24-2009	N41.78175 W-111.943017
A8	0.0012	Amphipod	3	Dingle	Sept-29-2009	N 42.14462 W -111.32320
A9	0.0028	LN Dace	4	Cutler	Sept-24-2009	N41.78785 W-111.952957
A10	0.0018	LN Dace	2	Cutler	Sept-24-2009	N41.78785 W-111.952957
B1	0.0026	LN Dace	1	Cutler	Sept-24-2009	N41.78785 W-111.952957
B2	0.0019	LN Dace	3	Cutler	Sept-24-2009	N41.78785 W-111.952957
B3	0.0026	LM Bass	1	Cutler	Sept-24-2009	N41.78785 W-111.952957
B4	0.0022	LM Bass	3	Cutler	Sept-24-2009	N41.78785 W-111.952957
B5	0.0019	LM Bass	2	Cutler	Sept-24-2009	N41.78785 W-111.952957
B6	0.0021	LM Bass	4	Cutler	Sept-24-2009	N41.78785 W-111.952957
B7	0.0016	Perch	3	Cutler	Sept-24-2009	N41.78785 W-111.952957
B8	0.0026	Perch	2	Cutler	Sept-24-2009	N41.78785 W-111.952957
B9	0.0018	Perch	4	Cutler	Sept-24-2009	N41.78785 W-111.952957
B10	0.0027	Perch	1	Cutler	Sept-24-2009	N41.78785 W-111.952957

C1	0.0031	E. Macrophyte	1	Cutler	Oct-23-2009	N 41.77738 W - 111.94492
C2	0.0036	E. Macrophyte	3	Cutler	Oct-23-2009	N 41.77122 W - 111.94358
C3	0.0029	E. Macrophyte	4	Cutler	Sept-24-2009	N41.77526 W-111.9476
C4	0.003	E. Macrophyte	2	Cutler	Oct-23-2009	N 41.77488 W - 111.94351
C5	0.0255	Seston	1	Cutler	Oct-23-2009	N 41.77738 W - 111.94492
C6	0.0364	Seston	2	Cutler	Oct-23-2009	N 41.77488 W - 111.94351
C7	0.0355	Seston	3	Cutler	Oct-23-2009	N 41.77122 W - 111.94358
C8	0.0291	Seston	4	Cutler	Sept-24-2009	N41.78175 W- 111.943017
C9	0.0271	Seston	1	Dingle	Sept-29-2009	N 42.13345 W - 111.30936
C10	0.0282	Seston	2	Dingle	Sept-29-2009	N 42.13638 W - 111.29151
D1	0.0328	Periphyton	1	Dingle	Sept-29-2009	N 42.13345 W - 111.30936
D2	0.0313	Periphyton	2	Dingle	Sept-29-2009	N 42.14059 W - 111.29249
D3	0.0335	Periphyton	3	Dingle	Sept-29-2009	N 42.14854 W - 111.29295
D4	0.0344	Periphyton	1	Cutler	Oct-23-2009	N 41.77738 W - 111.94492
D5	0.0278	Periphyton	2	Cutler	Oct-23-2009	N 41.77488 W - 111.94351
D6	0.0337	Periphyton	3	Cutler	Oct-23-2009	N 41.77122 W - 111.94358
D7	0.0252	Zooplankton	1	Dingle	Sept-29-2009	N 42.13345 W - 111.30936
D8	0.0288	Zooplankton	2	Dingle	Sept-29-2009	N 42.13638 W - 111.29151
D9	0.0312	Zooplankton	3	Dingle	Sept-29-2009	N 42.14854 W - 111.29295
D10	0.027	Zooplankton	4	Dingle	Sept-29-2009	N 42.15278 W - 111.31091
E1	0.0307	Zooplankton	1	Cutler	Oct-23-2009	N 41.77738 W - 111.94492
E2	0.0282	Zooplankton	2	Cutler	Oct-23-2009	N 41.77488 W - 111.94351

Appendix 2. Average $\delta^{15}N$ and $\delta^{13}C$ values as well as standard deviation.

ID	Location	mean N15	St. dev.	Mean C13	St. dev.	Year
Chironomid	Cutler	10.10	3.48	-26.53	2.67	2009
Coenagrionid	Cutler	13.33	0.50	-28.30	0.83	2009
Amphipod	Dingle	4.89		-25.50		2009
LN Dace	Cutler	15.85	0.36	-29.58	0.48	2009
LM Bass	Cutler	16.56	0.23	-29.04	0.59	2009
Perch	Cutler	16.34	0.72	-29.78	1.08	2009
E. Macrophyte	Cutler	9.10	1.67	-28.52	0.48	2009
Periphyton	Dingle	-2.23	0.63	-23.38	2.54	2009
Periphyton	Cutler	9.02	1.16	-25.93	1.88	2009
Zooplankton	Dingle	4.89	2.40	-24.39	0.85	2009
Zooplankton	Cutler	5.27	0.30	-25.08	0.31	2009
Seston	Cutler	9.06	0.40	-28.00	3.65	2009
Seston	Dingle	1.50	0.82	-25.77	0.69	2009

Appendix 3. Isotopic information obtained on individual samples in the 2009 study of the Cutler and Dingle.

Location	ID	Site #	d15N	d13C	ug N/mg	ug C/mg	C:N
Cutler	Periphyton	1	8.0	-23.9	8.62	56.88	6.6
Cutler	Periphyton	2	8.8	-26.4	5.31	34.73	6.5
Cutler	Periphyton	3	10.3	-27.5	11.27	59.86	5.3
Cutler	LM Bass	1	16.7	-28.6	155.41	502.18	3.2
Cutler	LM Bass	3	16.8	-29.2	153.31	508.95	3.3
Cutler	LM Bass		16.2	-28.5	157.82	519.55	3.3
Cutler	LM Bass	4	16.5	-29.8	130.97	435.74	3.3
Cutler	LN Dace	4	15.5	-30.2	98.98	505.06	5.1
Cutler	LN Dace	2	15.6	-29.0	87.82	295.81	3.4
Cutler	LN Dace	1	15.9	-29.5	146.97	488.05	3.3
Cutler	LN Dace	3	16.3	-29.6	140.44	465.29	3.3
Cutler	Perch	3	16.6	-30.7	117.66	399.34	3.4
Cutler	Perch	2	17.0	-29.6	151.57	517.76	3.4
Cutler	Perch	4	16.4	-30.5	148.54	498.01	3.4
Cutler	Perch	1	15.3	-28.3	120.96	443.58	3.7
Cutler	Chironomid	4	11.7	-25.5	111.78	461.74	4.1
Cutler	Chironomid	2	4.9	-24.3	82.96	363.70	4.4
Cutler	Chironomid	1	12.2	-30.4	113.74	434.60	3.8
Cutler	Chironomid	3	11.5	-25.9	96.42	413.87	4.3
Cutler	Coenagrionid	4	13.1	-27.3	98.63	354.68	3.6
Cutler	Coenagrionid	3	12.9	-28.8	142.41	479.16	3.4
Cutler	Coenagrionid	2	13.9	-28.8	91.35	309.53	3.4
Cutler	Seston	1	9.4	-32.1	2.83	15.12	5.3
Cutler	Seston	2	8.6	-27.5	2.12	13.91	6.6
Cutler	Seston	3	9.3	-23.3	2.53	17.43	6.9
Cutler	Seston	4	8.9	-29.0	2.97	18.43	6.2
Cutler	E. Macrophyte	1	10.5	-28.4	4.10	443.56	108.1
Cutler	E. Macrophyte	3	7.2	-28.1	4.10	433.29	105.7
Cutler	E. Macrophyte	4	8.2	-28.4	2.81	388.43	138.2
Cutler	E. Macrophyte	2	10.5	-29.2	7.28	426.68	58.6
Cutler	Zooplankton	1	5.5	-24.9	12.68	55.40	4.4
Cutler	Zooplankton	2	5.1	-25.3	12.13	51.97	4.3
Dingle	Periphyton	1	-1.8	-26.1	2.91	21.66	7.4
Dingle	Periphyton	2	-3.0	-21.0	2.94	26.85	9.1
Dingle	Periphyton	3	-1.9	-23.0	3.41	25.76	7.6
Dingle	Amphipod	3	4.9	-25.5	58.50	296.82	5.1
Dingle	Seston	1	2.1	-26.3	1.20	8.81	7.4
Dingle	Seston	2	0.9	-25.3	1.27	10.20	8.1
Dingle	Zooplankton	1	6.4	-24.7	4.69	26.95	5.7
Dingle	Zooplankton	2	6.8	-23.2	3.22	20.60	6.4
Dingle	Zooplankton	3	4.9	-25.1	6.05	30.99	5.1
Dingle	Zooplankton	4	1.5	-24.6	3.62	20.09	5.5

A Qualitative and Quantitative Analysis of Aquatic Invertebrates in the Context of Nutrient Pollution: A Comparison of Eutrophic Cutler Reservoir and Dingle Marsh

Ryan Leonard and Ben Marett

Abstract

A quantitative and qualitative comparison was performed on the aquatic benthic invertebrates of two wetland systems, eutrophic Cutler Reservoir (UT) and less- impacted reference system, Dingle Marsh (ID). Quantitative samples were taken in the open water areas of each system using a. Qualitative samples were taken with kick nets at four different habitat types in each system. Overall benthic invertebrate densities were very low in both systems, but densities were twice as high in Dingle as in Cutler; with Caenidae mayflies being the dominate taxa in Dingle and Chironomidae (midge) larva the dominate taxa in Cutler Reservoir. The Hilsenhoff Biotic Index (HBI) was used both qualitatively and quantitatively as a metric to provide insight into the effect of eutrophication on invertebrate density and diversity. HBI indices from the qualitative kick net samples were lower (indicating better water quality) in Dingle than in Cutler, but the differences were not significant. HBI numbers for the quantitative samples were higher in Cutler Reservoir then in Dingle Marsh, but again, there was no significant difference between the sites. Finally, there were more pollution-intolerant taxa in Dingle and fewer pollution tolerant taxa than in Cutler Reservoir. These results suggest that Dingle Marsh is slightly less polluted than Cutler Reservoir.

Ryan and Ben sieving benthic samples in Cutler Reservoir

Introduction

Invertebrates are an essential part of any aquatic ecosystem. They provide food for other organisms, such as ducks and fish, (Kaufman 1996) and are good indicators of the overall health of the system they live in (Hilsenhoff 1977). Certain taxa such as Plecoptera (Stonefly) and Ephemeroptera (Mayfly) are known to be very intolerant of pollution, whereas taxa such as worms and certain species of Chironomidae (midge) larvae have been known to survive conditions on the verge of anoxia (Hokinson and Jackson, 2005). As nutrient levels in lakes or wetlands increase the rate of decomposition can also rise. This increase in decomposition decreases the concentrations of dissolved oxygen and allows for more tolerant invertebrates to out compete less tolerant species.

There are many metrics that can work together in assessing water quality. Relative abundance can be used to determine if there is one dominating taxa in a system (Miller 2009). The Hilsenhoff biotic index uses invertebrate density to assess how impacted a system is in relation to a variety of factors, particularly organic pollution (Hilsenhoff 1988). The Shannon diversity index uses relative abundance to measure the diversity of a system (Miller 2009). Density provides a metric by which food abundance in a system can be compared. Research has made it easier and faster to use these four metrics as indicators of water quality (Lear 2009, Hilsenhoff 1988, Hokinson 2005). These metrics will be applied to quantitative and qualitative data to observe the density and diversity between Cutler and Dingle Marsh in the context of eutrophication.

Cutler Reservoir, located in Cache County, Utah, is classified as a eutrophic system. It receives an annual load of 381,000 kg of total phosphorous (TP) per year (DEQ 2008). Point sources of TP include the Logan waste water treatment plant (WWTP), JBS Swift and Company, Hyrum WWTP, and Wellsville lagoons. These point sources contribute a substantial portion of the annual phosphorous load to the reservoir (DEQ 2008). Additionally, many agricultural and residential non-point sources empty into Cutler Reservoir via the rivers and streams that feed Cutler (DEQ). The eutrophication of water caused by high nutrient levels can lead to low levels of dissolved oxygen (DO), and can create an environment that adversely affects aquatic life.

Dingle Marsh, located to the north of Bear Lake, is used for water conveyance from the Bear River into Bear Lake. It also serves to remove nutrients like nitrogen and phosphorus from water before entering Bear Lake (Lamarra et. al. 1980). However, Dingle Marsh is considered to be somewhat pristine. Mean total phosphorus and chlorophyll concentrations measured in Dinble Marsh by Gilbert Rowley as part of the class project were 35µg/L and 3.7 µg/L respectively. Phosphorus and chlorophyll concentrations in Cutler, by comparison, were 6 and 9 times higher than in Dingle (Appendix 1). Dingle Marsh therefore provides a standard by which pollution in Cutler can be compared.

The invertebrate diversity in Cutler may be limited by high nutrient levels in the reservoir (Gray 2009). Stoller (2007) showed that benthic invertebrate biomass in Cutler Reservoir is very low, thus affecting the diet of fish and the birds that utilize the system. The eutrophication and the resulting low dissolved oxygen in Cutler create an environment that may adversely affect density and diversity of the benthic community. The objective of our study was to assess if there were differences in invertebrate density and diversity between Cutler Reservoir and Dingle Marsh.

Methods

Quantitative Field Sampling for Benthic Invertebrates

Field sampling was conducted on September 24, 2009 at Cutler Reservoir and September 29, 2009 at Dingle Marsh. All sampling sites in Cutler were located in an open water area located south of Benson Marina (Figure 1). Sites from Dingle were located in the open water area known as Mud Lake (Figure 2). These two areas were similar in depth and both are bordered by emergent bull rush.

In order to obtain benthic invertebrates for the quantitative samples, an Eckman dredge (20 cm x 20 cm) was used at 16 different locations; eight in Cutler marsh and eight in Dingle marsh. Dredge samples were taken at depths between 0.75 m - 1.25 m. We used a 500 µm sieve to separate small organic and inorganic matter, as suggested by Lind (1985). Removing these small particles allowed for easier processing in the lab. If any sample had submerged macrophytes protruding from the dredge the excess foliage was cut off at the edge of the dredge and discarded prior to being sieved. Each sample was placed in a one quart mason jar or plastic bag and preserved with 95% ethanol. Samples with large amounts of organic matter after sieving were separated into multiple jars.

Qualitative field sampling of invertebrates

Qualitative invertebrate samples were taken in Cutler Reservoir and Dingle Marsh using 30 cm x 30 cm kick nets with 500 µm mesh. Four sampling locations were chosen in each system. Cutler Reservoir was sampled first. As a result, the four habitat types that were chosen for sampling in Dingle Marsh mirrored the habitat types sampled in Cutler where possible. For instance, a log, a rocky bank, water surrounding bulrushes and submerged macrophyte beds were sampled in each system. Because the number of taxa naturally varies from habitat to habitat, it was necessary to sample different habitat types to maximize the number of taxa captured. This was necessary to obtain more representative data for each of the systems. As sampling effort increases, estimates of the biological status of a site become more precise (NERC Open Research Archive, http://nora.nerc.ac.uk/4628/1/SCHO03 07BMHG-e-e.pdf). Therefore, it is necessary to implement equal sampling effort in each system to produce comparable data. To produce such data, each sampling location was sampled for two minutes and with two nets for three of the habitat types. The forth habitat type (wooden logs) were sampled for two minutes with only one
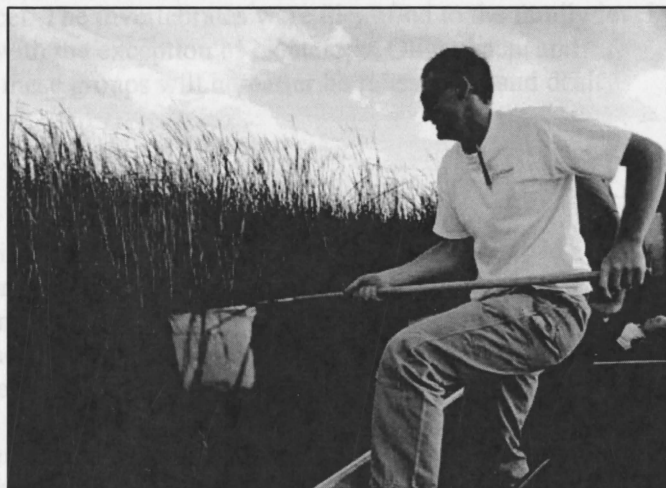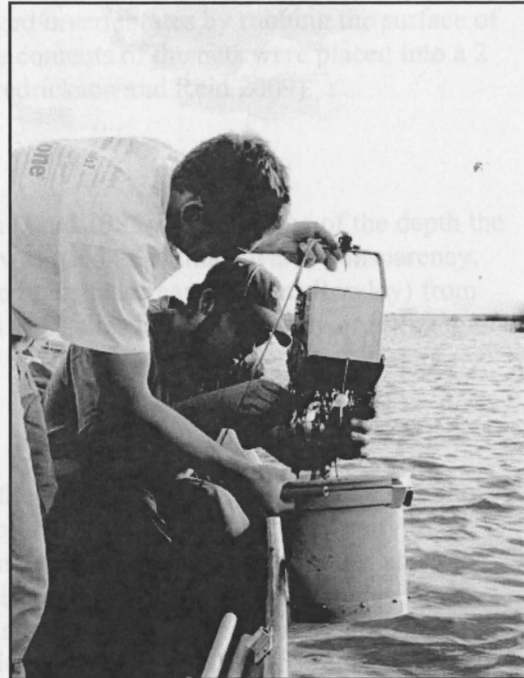
net. One person operated the net while another dislodged invertebrates by rubbing the surface of the log. After sampling for each site was complete, the contents of the nets were placed into a 2 gallon plastic bag and preserved with 95% alcohol (Fredrickson and Reid 2009).

Water Quality Measurements

To measure water clarity we used a 20 cm Secchi disk (Lind 1985). The average of the depth the disk disappeared and reappeared in the water column was used to estimate water transparency. Total phosphorus and chlorophyll levels were measured by a classmate (Gilbert Rowley) from water at four of our sample sites in Dingle Marsh, and at 17 sites in Cutler Reservoir utilizing the methods described in Chapter 1.

Lab Procedures for Benthic Invertebrate Samples

To assist in the sorting and identification time, ten of the quantitative samples were sent to the BLM's National Aquatic Monitoring Center (Bug Lab) located at Utah State University to be sorted and identified. We sorted and identified the remaining six quantitative samples using the same protocol as the Bug Lab (National Aquatic Monitoring Center 2009). All samples were again run through a 500 μm sieve to separate any remaining sediment from the sample. In the Bug Lab protocol, invertebrate samples are sub-sampled to a percent (50%, 25%, 12.5% and so on) of the original sample that is thought to contain 600 invertebrates. However, since none of our samples contained 600 invertebrates, the samples were analyzed completely for invertebrates. Small portions from each sample were placed into Petri dishes and viewed under a dissecting microscope, to sort through the organic matter. This process was repeated until the whole sample had been sorted. The invertebrates found were separated into their respective orders and preserved in 70% ethanol until they were identified. Invertebrates were keyed out to various taxonomic levels using *Aquatic Insects of North America* (Merritt and Cummins 2008). Finished samples were preserved in 70% ethanol. For the purpose of this study, most invertebrates were classified to the family level where possible, oligochaets (worms) were identified to class, copepods and cladocerans (zooplankton) were identified to order and nematodes (Nematodes) were identified to Phylum.

For the qualitative samples a 500μm sieve was used to wash as much sediment as possible from each sample. This clarified the water and made it easier to see the invertebrates. Organic material was separated from the invertebrates using a dissecting microscope. Each invertebrate found was placed in a glass vial with alcohol for later identification. After each organism was identified, a database was created using Microsoft Excel. The invertebrates were identified to the family level according to Merritt & Cummins (2008) with the exception of Nematoda, Oligochaeta and Isopoda. For the purposes of this project, these groups will hereafter be referred to, and dealt with, as families unless stated otherwise.

Data Analysis

Quantitative (dredge) samples— Density is a measure per unit area of the invertebrates in a sample. This metric provides an indication of the invertebrate biomass within the system. In systems of high nutrient pollution, the density of pollution-tolerant species can increase, where as the density of the intolerant species will normally decrease. However, depending on the severity of the pollution in a system, the overall density can either increase or decrease.

The Hilsenhoff biotic index (from here referred to as the HBI) uses assigned pollution tolerance numbers and density of the invertebrates to give an indication of water quality (Hilsenhoff 1988).

To calculate the HBI numbers for the quantitative data. The HBI value for all scoring taxa at one site was calculated then summed to give the HBI number for each sample. Pollution tolerant families like Chironomidae (midges) are assigned a high HBI value (e.g. 8), pollution intolerant families like Plecoptera (stoneflies) are assigned a low HBI value (e.g. 1) (Hilsenhoff 1988). The HBI uses a scale of 0-10 to rate the water quality. A score close to 0 signifies excellent water quality and a score close to 10 signifies poor water quality (Hilsenhoff 1988). Additional pollution tolerance numbers were obtained from (Arims, Handbook 2, 2002).

The Shannon diversity index uses the relative abundance and evenness of invertebrates in a sample to measure the uncertainty of choosing single taxa at random. The uncertainty of choosing certain taxa increases as the number of taxa present at that site increases. The Shannon diversity was calculated using the following equation:

$$H = -\sum_{i=1}^{s} \left[\left(\frac{n_i}{n} \right) \ln \left(\frac{n_i}{n} \right) \right]$$

The Shannon number is represented by H, n_i is the number of individuals in the sample from a specific family (s), and n is the total number of individuals in the sample (Ludwig 1988). High levels of taxa diversity are represented by a higher value of H. Sites with only one taxa score a value of zero which represents no diversity at that site.

Relative abundance measures the abundance of unique taxa as a percent of the taxa present at that site. For the purpose of this study, relative abundance was measured by summing the densities of each unique taxon at each site and dividing it by the total density of all eight samples. We calculated the index for both individual stations and for pooled data from all the stations within each wetland.

Qualitative (sweep net) samples—The HBI was also used to provide a means by which one system could be deemed more polluted than the other according to qualitative data. A pollution tolerance number was assigned to each taxonomic family according to (Hilsenhoff, 1988). The HBI value for each site was calculated. Because there were four sites or samples for each wetland, and therefore four HBI values for each wetland, an average of each of the four site HBI values was taken to be the total system HBI value. Because there were two variables that contributed to differences in the HBI values, namely habitat type and the wetland, it was necessary to run a 2-way ANOVA test to assess statistical differences.

It was also useful to plot the HBI values of one system against another in respect to their habitat type. For example the HBI values of the bulrushes in each system were compared with one another and so on. A graph of the number of families was plotted against the HBI values to illustrate how the number of high and low HBI values occurred in each system. For instance, if chironomidae, which has an HBI value of 8, appeared in three of the four Cutler samples, then it would count as three distinct 8's. In addition to a comparison of the total number of families per system, it was also necessary to plot the total number of families per habitat type. For instance, the macrophyte sample at Cutler had six families and the macrophyte sample at Dingle had 7 families. This graph relates back to the fact that a more polluted system will have fewer families which can be expanded to say that the number of families in analogous habitat types will also vary with pollution levels.

Results

Quantitative—Benthic invertebrate densities measured with the dredge samples were low and highly variable in both Cutler and Dingle (Figure 3). Mean invertebrate densities were only $179/m^2$ and $470/m^2$ in Cutler and Dingle, respectively. Despite the differences in the means, a t-test showed that there was no significant difference between the two systems in a site-by-site comparison (p = 0.17). Dingle site 5 had the largest density of any site at 4164 individuals/m^2 (Figure 3). There were two sites (Cutler 1 and Dingle 7) with densities of only 24 individuals/m^2 which was the lowest density recorded (Figure 3). Cutler site 7 was located closest to the Logan WWTP discharge area and had a density of 193 individuals per m^2 (Figure 3).

The mean HBI indices for the quantitative samples were similar in the two systems and there was no significant difference between the two (t-test; p = .39) The HBI for Dingle (7.48) was slightly lower than the HBI for Cutler (7.55) (Figure 4). The highest HBI value at any site was 8.00 and was seen in 5 different sites; Cutler sites 1, 3 and 8 and Dingle sites 2 and 7 (Figure 4). The lowest HBI number was 6.43 and is seen at Dingle site 4 (Figure 4).

In contrast to the HBI index, the Shannon diversity index for the combined station data in Dingle was considerably higher (1.09) than the index for the invertebrates in Cutler Reservoir (0.74) (Figure 5).

Relative abundance analyses at Cutler show that Chironomidae (midge) larva were the dominant taxa in that system comprising 69% of the total invertebrates sampled (Figure 6 a.). Oligochaetes (worms) were second, comprising 29% of the invertebrates; all other invertebrates comprise the remaining 2%; Sperchonidae (mite) Hyalellidae (scud) and Ceratopogonidae (no-see-ums) each contributed 0.66% (Figure 6 a.). Relative abundance data for Dingle showed Caenidae (mayfly) were the dominate taxa comprising 52% of all invertebrates sampled. Chironomidae larvae were the second most abundant, comprising 37% of the sampled invertebrates (Figure 6 b.) The remaining 12% were represented as follows: Pisidiidae (thumbnail clam) 5%, Oligochaeta and Sperchonidae 3%, and Hydroptilidae (micro caddis fly larva) 1% (Figure 6 b).

In Cutler Reservoir, the highest density of copepods was 896 per m^2. In Dingle Marsh, copepods only reached a density of 48 per m^2. The density of cladocera found is negligible.

Qualitative kick net samples— Overall diversity in the qualitative samples was relatively similar between Dingle and Cutler (Figure 7). Dingle was slightly more diverse with a total of fourteen families and Cutler had a total of twelve families. While a smaller number of taxa do indicate that there is more pollution in the system, a difference of only two families does not provide enough evidence to say that one system is more or less polluted than the other.

The four HBI values calculated for kick net samples for each of the habitat types in each system and then averaged to obtain the total HBI value(Table 1). The overall Cutler HBI value was 7.66, indicating that water quality is poor and that it has very significant organic pollution (Table 2). The overall Dingle HBI value was 7.09 indicating that it also has fairly poor water quality and that it had significant organic pollution (Table 2). Although both systems were significantly polluted, the lower HBI value for Dingle suggests that it is less polluted than Cutler Reservoir. A 2-way ANOVA (habitat type x wetland) yielded a p-value of 0.13 for habitat type and a p-value of 0.10 for the wetland, indicating that there was no statistically significant difference in the HBI values between the two systems or between the different habitat types.

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Despite the indifference of the ANOVA results, several graphs suggest that there may be a small difference in the pollution of levels of Cutler and Dingle. For instance, a graph of the HBI values for each habitat type shows that three of the four habitats in Dingle had lower HBI values than their respective habitat in Cutler (Figure 8). The fourth habitat type had equal HBI values for both systems. In Dingle Marsh, there are more families occurring with low HBI values and fewer families with high HBI values. Conversely, there are more families occurring with high HBI values in Cutler and fewer families with low HBI values (Figure 9). There were fewer families in three of the four habitat types in Cutler compared to Dingle which also suggests that Dingle may be slightly less polluted (Figure 10).

Discussion

Invertebrates are a major food source for the birds (Kaufman 1996) and fish that inhabit wetlands such as Cutler Reservoir and Dingle Marsh. Extremely low densities of invertebrates were seen in both wetlands. A low density of invertebrates, particularly cladocerans, suggests that there is a limited food supply for fish and birds that utilize them for food. Cladoceran species form a major part of the diet of some birds including the Northern Shoveler, Gadwall, and Mallard Hens (Eldridge 1990). Copepods account for a small portion of waterfowl diets (Eldridge 1990). Wurtsbaugh & Hawkins (1990) found that there were nearly 10,000 copepods per m^2 in Bear Lake. Copepod densities found in this study were negligible in comparison.

While the ANOVA (qualitative data) or t-test (quantitative data) did not render any statistically significant evidence to suggest that the HBI values varied between the two systems, the combined data suggests that Dingle Marsh may be slightly less polluted than Cutler Reservoir. This is illustrated by the generally lower HBI values in Dingle, higher number of families in Dingle for three of the four habitat types, and the occurrence of fewer taxa with high HBI values and more occurrences of taxa with low HBI values in Dingle. The overall Dingle HBI value was lower than it was for Cutler.

Total phosphorus and chlorophyll values are common measures of eutrophication. The differences between the two systems were obvious, with mean chlorophyll levels being approximately six times higher and total phosphorus about 9 times higher in Cutler than in Dingle (Appendix 2). The high concentration of phosphorous, which contributes to high chlorophyll levels, decreases water clarity. The Secchi depth is also a common measure by which eutrophication can be assessed. Shallow Secchi depths can be the result of high chlorophyll levels in the water. The Secchi depth measurement for Cutler was 0.38 m. The mean Secchi depth across all 8 sites for Dingle was 0.64m (at some sites Secchi depth could not be measured because it was greater than the station depth). While it is possible that high phytoplankton concentration (eutrophication) in Cutler is the culprit behind the low Secchi value, there may be other sources of material in the water. For instance, there are high numbers of carp in the system. Carp stir up the sediment, which will decrease water transparency. However, there are also carp in Dingle Marsh. Wind can also cause sediments to become suspended in the water column. This is especially true in shallow systems like Cutler and Dingle and where macrophytes are sparse, as is the case of Cutler Reservoir.

Submerged macrophytes, which provide a source of refuge for invertebrates and increase diversity (Momo *et. al*. 2005) were present in both systems. Several quantitative sample sites in Dingle had very dense patches of submerged macrophytes present whereas quantitative Cutler samples had none. The overall diversity between Cutler and Dingle was not consistent with the findings of Momo *et. al*. (2005). However, even when submerged macrophytes were present in Dingle, the dredge did not always return containing macrophytes. It is likely that the dredge

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passed through the macrophytes and collected sediments below the tangle mass of plants. These underlying areas are likely low-oxygen environments, as limited light would reach below the macrophytes. It is apparent that our method for sampling the macrophytes was somewhat faulty. Even if the macrophytes were collected by the dredge like we hoped, only parts of the plant would have been sampled. Ideally, the entire body of the macrophyte plants and all of the plants in a known area including the underlying sediment should have been sampled. This would have provided us a more accurate number of organisms per m^2 in the sites with macrophytes.

It is important to note that the stations selected in Cutler and Dingle were not representative of the entirety of each wetland. In Cutler Reservoir all of our quantitative samples were taken in the large central bay where the Logan WWTP effluent is discharged, so our data is not applicable to other parts of the reservoir. There were no submerged macrophytes near our sample sites west of the Logan WWTP. Similarly, we took quantitative samples only in Mud Lake at Dingle Marsh because this area was very similar to the open areas sampled in Cutler Reservoir. The sampled area in Cutler consisted only of bare sediments and bulrushes. Qualitative samples were taken in other areas of Cutler. It was apparent from visual inspection of the qualitative samples, particularly the submerged macrophyte habitat, that abundance and diversity were greatly increased in other areas of the reservoir. However, many of these taxa had high HBI values which would have driven up the overall Cutler HBI value.

Results from the data show that the density of invertebrates in Dingle was almost twice as high as Cutler. The high density in Dingle was due in large part to Site 5 which had a density of Caenidae (mayflies) that was almost higher than densities all of the Cutler samples combined. Based on visual examination of the single qualitative macrophyte sample from Cutler, it is likely that quantitative macrophyte samples from Cutler would have produced very high numbers as well.

Immediately west of Dingle Marsh there is a canal that flows from Bear Lake to the Bear River. This canal is connected to Dingle Marsh but definitely lies to the side. When fresh water is drawn from Bear Lake, it flows north through the canal thus offering less polluted water to the habitat in the canal when it is in use. This is significant to this study because the log and the rocky bank that were sampled in Dingle Marsh were in the canal. These habitats may be exposed to less polluted waters during parts of the year. However, HBI values taken at the log and rocky bank sites in Dingle are consistent with other HBI values in the system and with their respective habitat types in Cutler.

In conclusion, because of restrictions on funding and time in this study, it was only possible to take a limited number of samples. It is likely that further sampling of more habitat types, and replicates of those samples, would have increased the number of taxa found. Thus, the uneven distribution of invertebrates in the systems would have been sampled more representatively than they were. This, in turn, would have provided a more accurate HBI value for each system.. That being said, the data collected during our study is a good baseline for understanding representative taxa in each of the marshes and does increase our limited understanding of these systems. It also demonstrates the importance of applying appropriate indices.

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Figure 1. Map of the open water area of Cutler Reservoir where samples were taken. The Quantitative sites sampled are marked with the yellow markers. The Qualitative samples sites are marked with green.
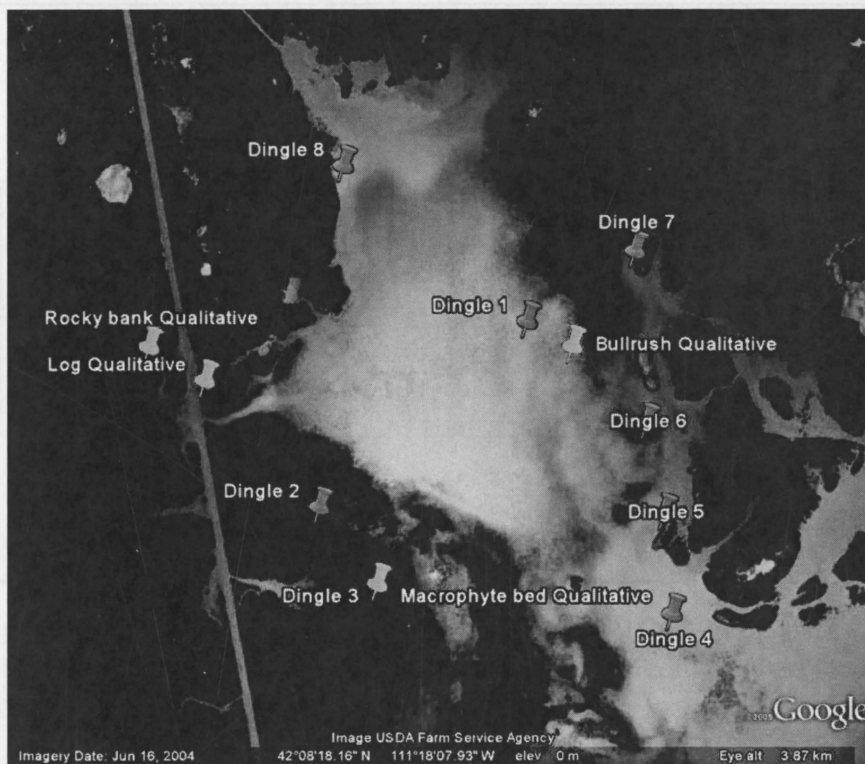

Figure 2. Map of open water at Dingle Marsh known as Mud Lake. The Quantitative samples are marked with the blue markers. The Qualitative sites are marked with green.
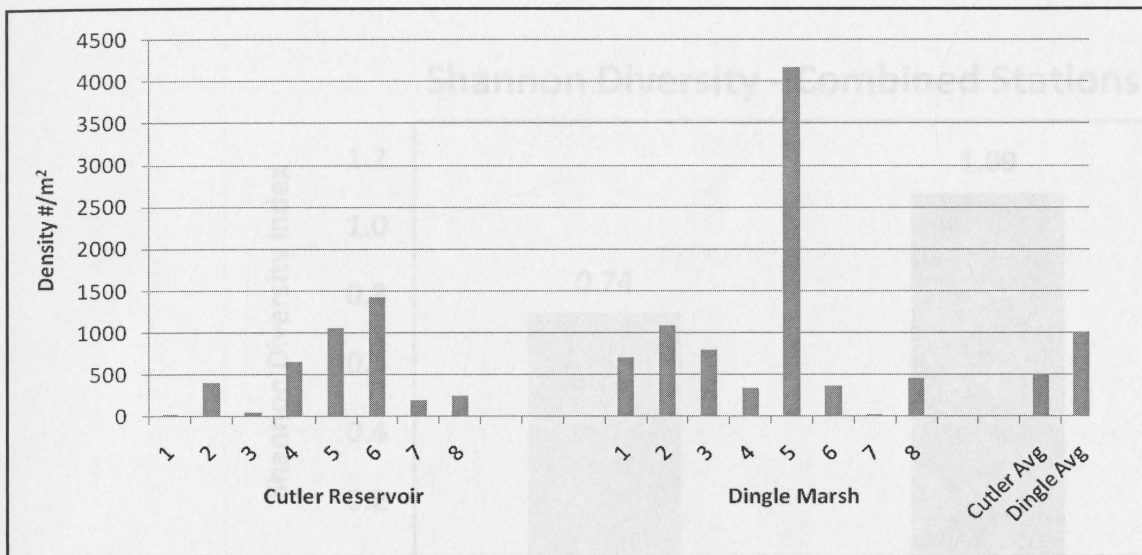
Figure 3. Densities at the 8 different sites in Cutler Reservoir (left) and Dingle marsh (right). The two bars on the end are the total of all eight sites for the two systems. Dingle Marsh sites 3 and 4 were the only samples that contained submerged macrophytes after the dredge was retrieved.
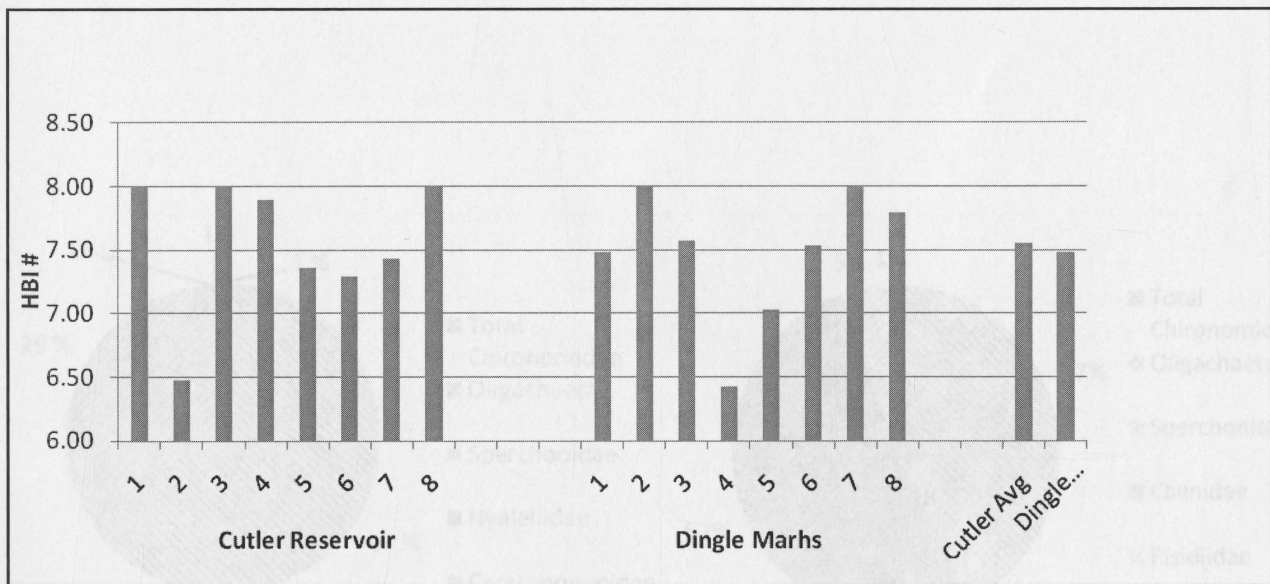
Figure 4. HBI numbers per site, Cutler Reservoir (left) and Dingle Marsh (right). The two bars at the end is the average HBI# for each system. The y axis starts at 6 because the all numbers lower than six are irrelevant for this study.

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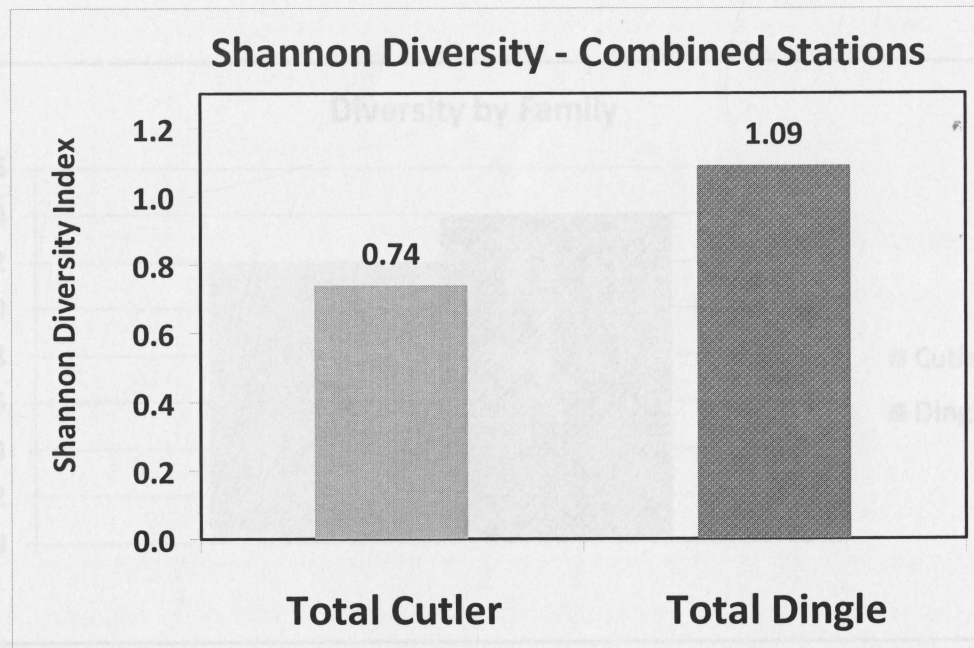
Figure 5. Shannon Diversity Index values for the eight Cutler Reservoir sites (blue) and Dingle Marsh (red).
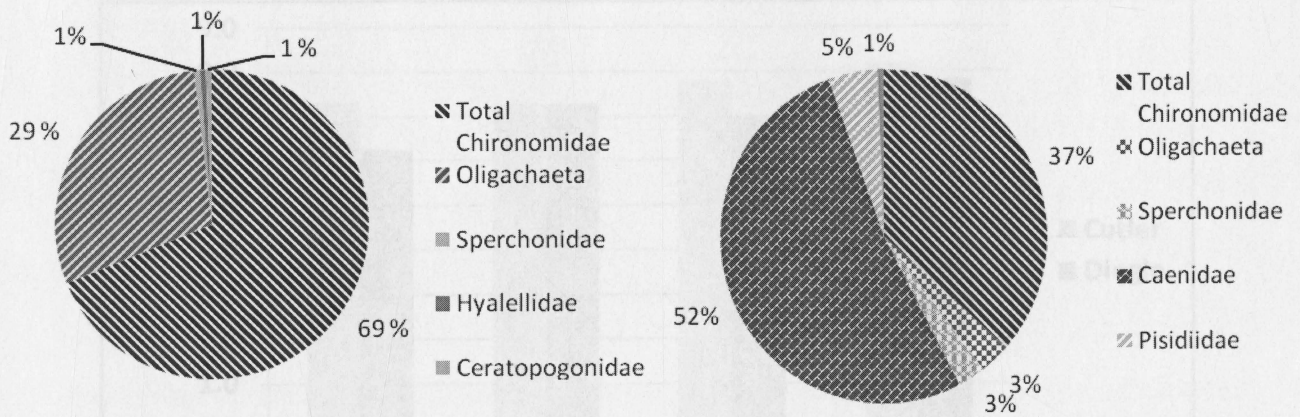
Figure 6. The relative abundances of benthic invertebrates in Cutler Reservoir (left) and Dingle Marsh (right). The very high contribution of Caenidae shown here is due to just one station. Because of the high abundances at that single station, it is very heavily weighted in the analysis. It is possible that the Cutler data could be biased this way too.
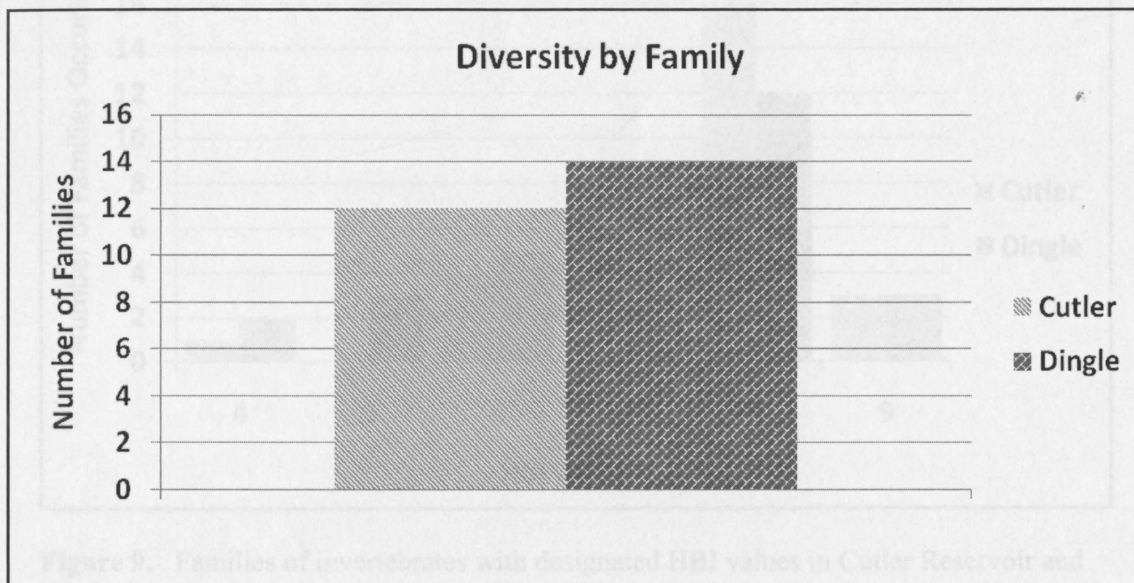
Figure 7. The total number of families found in the qualitative kick net samples in Cutler Reservoir and Dingle marsh. Fewer families in Cutler suggest that it is more polluted than Dingle. However, a difference of two families is not significant.
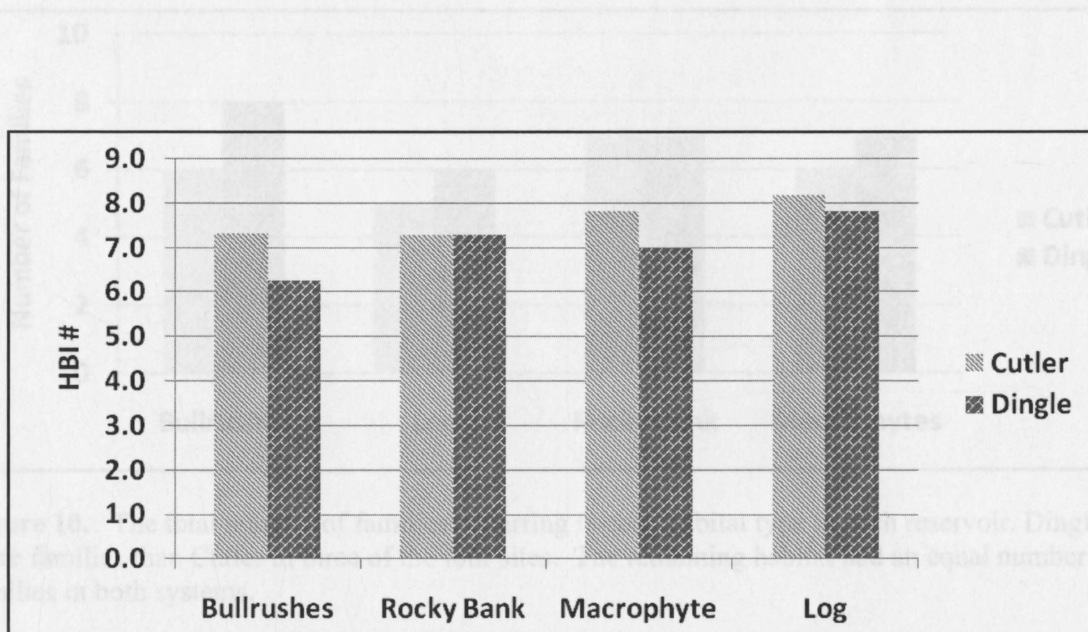
Figure 8. HBI values from each habitat type in each system. Notice that the HBI value is lower in three of the four Dingle Marsh sites. The remaining site has equal HBI values for Cutler Reservoir and Dingle Marsh.
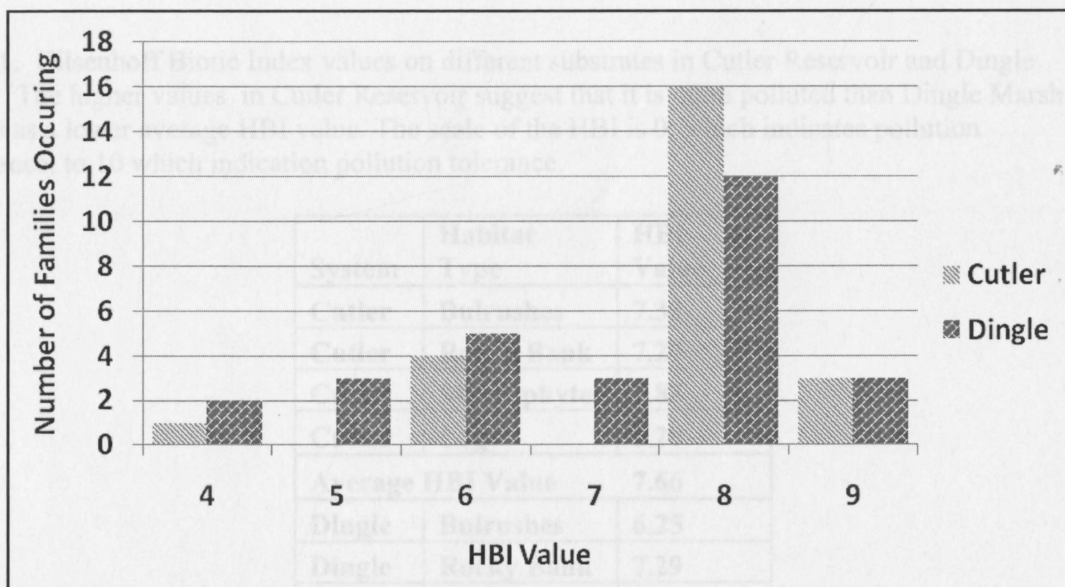
Figure 9. Families of invertebrates with designated HBI values in Cutler Reservoir and Dingle Marsh. Organisms with lower HBI values are less tolerant of organic pollution. Note that Dingle has more families in the lower HBI value range and fewer in the higher HBI value range than Cutler.
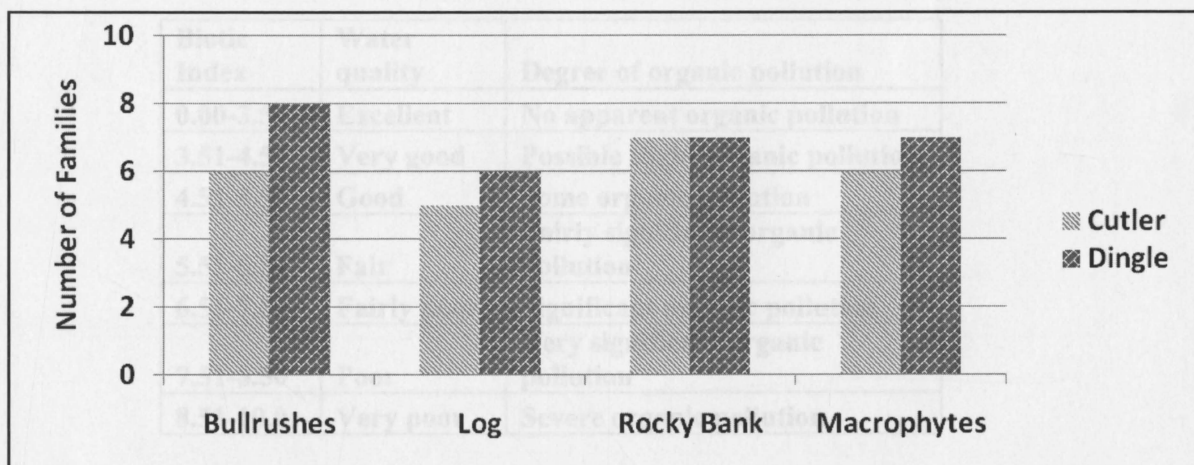
Figure 10. The total number of families occurring in each habitat type in each reservoir. Dingle had more families than Cutler in three of the four sites. The remaining habitat had an equal number of families in both systems.

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Table 1. Hilsenhoff Biotic Index values on different substrates in Cutler Reservoir and Dingle Marsh. The higher values in Cutler Reservoir suggest that it is more polluted than Dingle Marsh which has a lower average HBI value. The scale of the HBI is 0, which indicates pollution intolerance, to 10 which indication pollution tolerance.

System	Habitat Type	HBI Value
Cutler	Bulrushes	7.33
Cutler	Rocky Bank	7.29
Cutler	Macrophytes	7.83
Cutler	Log	8.20
Average HBI Value		7.66
Dingle	Bulrushes	6.25
Dingle	Rocky Bank	7.29
Dingle	Macrophytes	7.00
Dingle	Log	7.83
Average HBI Value		7.09

Table 2. Ranges of HBI values in relation to levels of organic pollution.

Biotic Index	Water quality	Degree of organic pollution
0.00-3.50	Excellent	No apparent organic pollution
3.51-4.50	Very good	Possible slight organic pollution
4.51-5.50	Good	Some organic pollution
5.51-6.50	Fair	Fairly significant organic pollution
6.51-7.50	Fairly poor	Significant organic pollution
7.51-8.50	Poor	Very significant organic pollution
8.51-10.0	Very poor	Severe organic pollution

Appendix 1. Total phosphorus (TP) and chlorophyll concentrations in Dingle Marsh and Cutler Reservoir during out study. Data from Gilbert Rowley.

Site	Site Reps.	Lab. Rep	Lat.	Long.	Time	Max Depth (m)	TP (ug/L)	Chl-a (ug/L)
				Dingle				
1	A	a	42.0139	-111.3156		0.9	58	5.8
1	B	b	42.0139	-111.3156		0.9		5.9
3	A	a	42.1335	-111.3094		1.1	38	5.1
3	B	b	42.1335	-111.3094		1.1		5.1
5	A	a	42.1364	-111.2915		1.2	24	1.8
5	B	b	42.1364	-111.2915		1.2		1.6
7	A	a	42.1485	-111.2930		1.0	19	1.9
7	B	b	42.1485	-111.2930		1.0		2.1
Mean						1.1	35	3.7
				Cutler				
1		a	41.7939	-111.9587	5:23	0.7	149	55.3
1	B	a	41.7938	-111.9588	7:30	0.6	157	20.9
2		a	41.7943	-111.9568	5:43	1.2	159	65.6
3		a	41.7814	-111.9429	5:57	0.9	206	26.7
3	B	a	41.7815	-111.9431	7:45	1.8	347	42.5
4		a	41.7801	-111.9436	6:04	1.5	167	19.2
5		a	41.7604	-111.9351	6:18	0.4	575	8.1
6		a	41.7638	-111.9339	6:26	0.4	850	105
7		a	41.7631	-111.9396	6:36	0.8	65	4.3
8		a	41.7484	-111.9507	7:37	1.0	137	5
9		a	41.7546	-111.9465	8:22	1.2	150	3.3
10		a	41.8027	-111.9566	5:31	0.6	94	22.9
11		a	41.8093	-111.9573	5:58	0.7	159	45.7
12		a	41.8176	-111.9530	6:15	2.7	82	41
13		a	41.8254	-111.9504	6:41	0.8	93	47.3
14		a	41.8268	-111.9607	6:54	2.7	43	38.9
Valley View		a	41.7453	-111.9567	7:06	1.2	175	6.2
6th South		a	41.7199	-111.9403	5:11	0.4	377	17.4
Cache Junction		a	41.8445	-112.0016	6:22	1.0	181	40.6
Mean						1.1	219	32.4
Ratio Cutler/Dingle							6.3	8.9

Appendix 2. Database for Cutler Reservoir and Dingle Marsh, data used for the qualitative section is found in this data set.

Site	Secchi (m)	Density #/m^2	HBI #	Average HBI	Shannon Diversity #	Shannon Diversity Index
Cutler 1	0.375	24.21	8.00	7.55	0.00	3.55
Cutler 2	0.375	411.62	6.47		0.79	
Cutler 3	0.375	48.43	8.00		0.00	
Cutler 4	0.375	653.75	7.89		0.32	
Cutler 5	0.375	1065.37	7.36		0.89	
Cutler 6	0.375	1428.57	7.29		0.65	
Cutler 7	0.375	193.70	7.43		0.90	
Cutler 8	0.375	242.13	8.00		0.00	
Dingle 1	0.450	702.18	7.48	7.48	0.81	5.37
Dingle 2	0.600	1089.59	8.00		0.60	
Dingle 3	>0.8	799.03	7.58		0.52	
Dingle 4	0.900	338.98	6.43		1.77	
Dingle 5	>0.95	4164.64	7.03		0.35	
Dingle 6	>.95	363.20	7.53		0.80	
Dingle 7	0.900	24.21	8.00		0.00	
Dingle 8	0.350	460.05	7.79		0.51	

Appendix 3. Database for Cutler Reservoir and Dingle Marsh for the quantitative dredge samples. Only one Secchi depth measurement was done in Cutler Reservoir.

Site	Lat	Long	Depth (m)	Average TEMP C	Secchi (m)	DO (mg/L)	Secchi (m)	Notes	Orthocladinae	Chironominae	Tanypodina	Total Chironomidae	Oligachaeta	Nemata	Sperchonidae	Caenidae	Pisidiidae	Hyalellidae	Copepoda	Cladoceran	Ceratopogonidae	Hydroptilidae	Total Density#/m²	HBI #	Average HBI	Shannon Diversity by site	Shannon Diveristy Index
Cutler 1	41.7694	-111.9455	0.80	19.83	0.375	11.30	0.375	No Submerged Aquatic Vegetation (SAV)	0	1	0	1	0	0	0	0	0	0	0	0	0	0	24.2	8.00	7.55	0.00	3.55
Cutler 2	41.7666	-111.9429	1.00	18.53	0.375	9.93	0.375	No Submerged Aquatic Vegetation (SAV)	0	4	0	4	13	0	0	0	0	0	37	0	0	0	411.6	6.47		0.79	
Cutler 3	41.7659	-111.9480	0.80	18.33	0.375	15.23	0.375	No Submerged Aquatic Vegetation (SAV)	0	2	0	2	0	0	0	0	0	0	0	0	0	0	48.4	8.00		0.00	
Cutler 4	41.7641	-111.9492	0.80	18.33	0.375	16.80	0.375	No Submerged Aquatic Vegetation (SAV)	0	22	3	25	0	0	1	0	0	1	0	0	0	0	653.8	7.89		0.32	
Cutler 5	41.7619	-111.9520	0.90	17.47	0.375	11.97	0.375	No Submerged Aquatic Vegetation (SAV)	1	29	0	30	13	0	0	0	0	0	1	1	1	0	1065.4	7.36		0.89	
Cutler 6	41.7640	-111.9543	0.95	19.07	0.375	15.10	0.375	No Submerged Aquatic Vegetation (SAV) Muddy bottom- very little material after	0	37	1	38	21	0	0	0	0	0	0	0	0	0	1428.6	7.29		0.65	
Cutler 7	41.7631	-111.9358	0.75	19.87	0.375	14.03	0.375	No Submerged Aquatic Vegetation (SAV) Sample taken by waste water treatment plant.	0	5	0	5	2	1	0	0	0	0	0	0	0	0	193.7	7.43		0.90	
Cutler 8	41.7589	-111.9398	0.85	17.67	0.375	16.23	0.375	No Submerged Aquatic Vegetation (SAV)	0	10	0	10	0	0	0	0	0	0	0	0	0	0	242.1	8.00		0.00	
Dingle 1	42.0139	-111.3156	0.90	17.30	0.450	7.10	0.450		0	21	1	22	2	0	3	2	0	0	0	0	0	0	702.2	7.48	7.48	0.81	5.37
Dingle 2	42.1371	-111.3129	0.90	16.93	0.600	8.53	0.600		3	29	0	32	0	0	0	0	13	0	0	0	0	0	1089.6	8.00		0.60	
Dingle 3	42.1335	-111.3094	1.10	16.53	>0.8	10.70	>0.8	Macrophyte Bed	2	23	1	26	7	0	0	0	0	0	0	0	0	0	799.0	7.58		0.52	
Dingle 4	42.1318	-111.2912	1.10	17.10	0.900	11.70	0.900	Macrophyte Bed	0	3	2	5	1	0	3	3	0	0	1	1	0	2	339.0	6.43		1.77	
Dingle 5	42.1364	-111.2915	1.20	17.13	>0.95	9.80	>0.95		8	0	1	9	0	0	2	158	3	0	0	0	0	0	4164.6	7.03		0.35	
Dingle 6	42.1406	-111.2925	0.90	17.17	>.95	12.00	>.95	Macrophyte bed but no macrophytes in Ekman Dredge sample.	4	6	0	10	0	0	1	4	0	0	0	0	0	0	363.2	7.53		0.80	
Dingle 7	42.1485	-111.2930	1.00	17.00	0.900	9.47	0.900		0	1	0	1	0	0	0	0	0	0	0	0	0	0	24.2	8.00		0.00	
Dingle 8	42.1528	-111.3109	0.75	16.50	0.350	8.67	0.350		2	13	0	15	0	0	0	4	0	0	0	0	0	0	460.0	7.79		0.51	

A Eutrophic History: A Paleolimnological Analysis
of Cutler Reservoir Sediments

J.D. Abbott and Deb Collins

Abstract

Cutler Reservoir is an impaired system according to criteria of the Water Quality Act. It is currently on Utah's 303(d) list of impaired bodies of water for having low dissolved oxygen and high levels of phosphorous. High levels of phosphorous (and nitrogen) are being released into Cutler Reservoir which is classified as eutrophic. Our study focuses on investigating the past to see how water quality has changed in the reservoir over time. Using a 19.5 cm-long sediment core taken from the reservoir, we measured lead-210, total phosphorous, chlorophyll, and pheophytin. Total phosphorous, chlorophyll, and pheophytin were extracted from 25 sections the sediments and analyzed individually. Nitrogen and carbon were also measured by Dan Lamarra as part of an isotopic assessment of eutrophication. The core did not yield a consistent lead-210 profile, making it difficult to date the material: It is likely that the core was from disturbed sediments. However, the best estimate was that the core recorded the reservoir's history from 2009 to 1933. Over the past ~80 years phosphorous levels increased from 0.6 mg P/g to approximately 0.75 mg P/g. Nitrogen and carbon content followed the same trend as phosphorus. Chlorophyll pigments were higher in the top 2-3 cm of the core, but the breakdown product of chlorophyll, phaeophytin, showed no trend with depth. The increase in phosphorus (if real) is likely human-based; coming from increased populations, urban run-off, agricultural run-off, and the Logan City Wastewater Treatment Plant. Although our interpretation is limited by the likelihood that the sediments where the core was taken had been disturbed, the analysis demonstrated the potential usefulness of this approach for understanding the time-course of anthropogenic impacts on the reservoir.
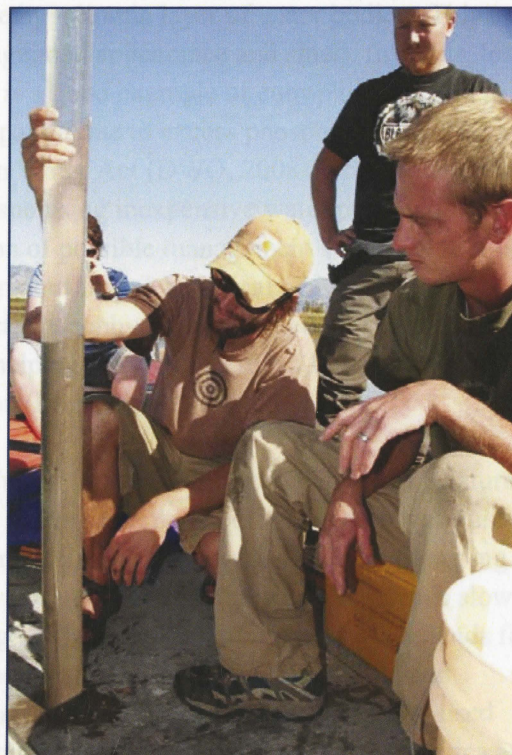
Introduction

Chlorophyll *a* and total phosphorous levels are key indicators of high nutrient levels in a water body. High concentrations of phosphorous and other nutrients often lead to eutrophication (Correll, 1999). As an ecosystem becomes more productive via this nutrient enrichment it stimulates primary producers and is referred to as eutrophication (Dodds, 2002). This stimulates algal blooms and can create anoxic conditions in the hypolimnia layer of water bodies. This causes several other problems that are hard to manage; bad appearance and smell, fish kills, toxic conditions, etc. Cutler Reservoir (hereafter Cutler) is a good example of eutrophication due to nutrient accumulation. Cutler has been listed as impaired due to excess phosphorous loading and other factors that violate section 303(d) of the Clean Water Act (DWQ, 2008). This violation of the Clean Water Act and difficulty in solving the issue using inexpensive water purification methods, has lead to serious debate and investigation of possible management solutions. In this paper we focus on the aforementioned key indicators, but from a historical perspective. Phosphorus, chlorophyll, and its breakdown product, phaeophytin are retained in the sediments, which in turn can be extracted and measured, thus helping us to reconstruct a limnological history of the nutrient loading and as a result, eutrophication (Engstrom et al. 1985).

There are two explanations for the impairment of Cutler. First, Cutler Reservoir may be naturally eutrophic by nature of being a wetland. Cutler was a natural wetland before the dam was built in 1927 (Ricks, 1953). Wetlands are very good at nutrient immobilization and sediment trapping. Wetlands are naturally shallow, therefore as floodwater moves through the area it spreads slowly, dropping sediments and nutrients as it passes through. The retention efficiencies of wetlands for various materials vary greatly; however, some of the greatest inputs include sediment and phosphorous (Dodds, 2002). As well as being natural nutrient sinks, wetlands are being used increasingly as a way to remove nutrients from sewage effluent, especially in North America and Europe (Dodds, 2002).

This leads to the second explanation for impairment–anthropogenic influence. Cutler is an example of the aforementioned sewage nutrient removal process (DWQ, 2008; Moore et al., 1991). The Logan City Wastewater Treatment Plant and other industries release their effluent either directly or indirectly into Cutler. Along with agricultural run-off from the surrounding farms and ranches; human caused effluent could be a major influence in the highly eutrophic conditions of the reservoir. In addition, the population of Cache County has also been steadily growing. In 1990 the population was 70,183, in 2000 91,391, and in 2008 112,610. Cache County grew by 21% from 2000 to 2008. The current growth rate, if sustained, would result in a population of 400,000 fifty years from now. This increase in population contributes to increased urban sources of run-off. As mentioned before, increased loading results in increased primary production. This in turn causes an oxygen demand increase because of algal decomposition. Further, hydrologic management of natural lakes can increase sedimentary nutrient flux under circumstances where lake volume and water levels become more variable (Hambright, et al. 2004).

We looked at the sedimentary record for answers to the possible causes of the high nutrient loads and resulting eutrophication. We hypothesized that the top layers of sediment contain higher levels of phosphorous, nitrogen and chlorophyll due to anthropogenic influences from the surrounding area. We measured extracted phosphorous, nitrogen chlorophyll *a*, and pheophytin *a* from a sediment core, which was also measured for lead 210 (^{210}Pb) to estimate the age of sediments.

Methods

Site Description—Cutler Reservoir, located on the border of Box Elder and Cache counties (41°77' N 111°94' W), covers 9.9 km^2 in Northern Utah. It is classified as eutrophic and has many characteristics of a classic wetland. It was built in 1927 to generate hydropower, but the reservoir also provides irrigation, recreation, wildlife, and aquatic habitat (Ricks, 1953). As an impoundment of the Bear River, Cutler has suffered from severe sediment accumulation, reducing its holding capacity and overall depth. The Logan, Little Bear, Bear, and Blacksmith Fork Rivers, Spring Creek and Swift Slough all discharge into Cutler.
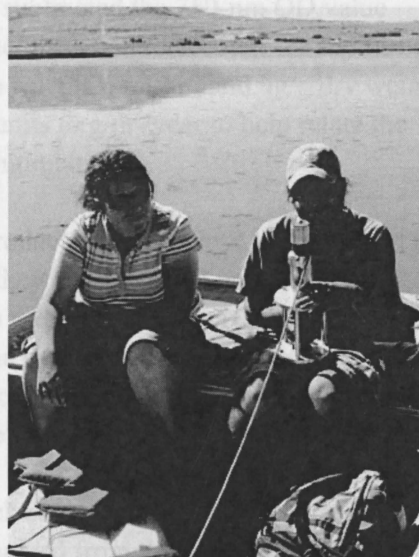
Sediment core—We used a Glew gravity corer with a 7.7-cm diameter to obtain three sediment cores from Cutler. We attempted coring at other locations but at many of these the upper layer of sediment was comprised of clay, which was almost impossible to penetrate. We then took the cores back to the university laboratory. The core selected for analysis was taken in the large open-water area between Benson Marina and Highway 30 (41.76638 N, -111.94276 W) where the depth was approximately 1 m. .

The sub-sampled core was 19.5 cm long. Upon inspection we could see a definite change in sediment characteristics at 4.5 cm from the surface layer of the core. The core was sliced into 25, 0.75 cm sub-sample sections. For percent dry weight analysis samples of 0.1-1.0 grams were taken from each section and put in plastic centrifuge tubes for drying to constant weight at 70°C. We sent in every other section, along with six other random sections to MyCore Scientific (Deep River, Ontario, Canada) for ^{210}Pb dating analysis utilizing alpha spectroscopy. A background (supported) lead 210 level of 0.076 Bq/g was assumed. This activity was measured in a Farmington Bay (Great Salt Lake) core (Naftz et al. 2000). Other samples were taken for chlorophyll *a*, pheophytin *a*, phosphorous, carbon and nitrogen isotope analyses.

Chlorophyll and pheophytin procedures—For the chlorophyll *a* and pheophytin *a* analysis we followed the protocol from *Standard Methods* [APHA]. Our protocol required at least 3 cm^3 of sediment for chlorophyll extraction. Samples were kept frozen until we were ready to extract the chlorophyll. A wet weight sediment weight equivalent to 3 ml was placed in 15 mL plastic

centrifuge tubes. We then added 10 mL of acetone to the tubes, and then sonicated the samples for 30 seconds to help break apart cells, which facilitates the extraction process. We wrapped the samples in tin foil and put them in the refrigerator for 24 hours complete the extraction. The following morning we analyzed the chlorophyll a and pheophytin a values from the extracts. We clarified the extracts by spinning them in centrifuge at high speed for 3 minutes. We transferred 3 ml of the clarified extract to the cuvette and read the optical density (OD) with a Cary Model 50 spectrophotometer at wavelengths of 750, 664, 647, and 630 nm. We then acidified the extract in the cuvette with 0.1 ml 0.1N HCl, waited approximately 90 seconds and re-read the extract. For our calculations, we used the OD at 750 and 664 nm. We subtracted the 750-nm OD value (turbidity reading) from the 664-nm OD before and after acidification to account for any turbidity in the extract. Using the corrected values, we were able to calculate mg/m^3 and ug/g dry weight of chlorophyll a and pheophytin a. We then adjusted the units to $g/m^2/year$ to help relate the chlorophyll and pheophytin to the amount of sediment accumulation.

We focused on the levels of pheophytin, the breakdown product of chlorophyll, in the rest of the study because pigments are often changed before or after they are deposited in the sediment. This is because of a combination of degradation mechanisms in the water column (Carpenter et al., 1986; Hurley & Armstrong, 1990: Leavitt, 1993) and in the surface sediments during deposition (Leavitt, 1993; Steenbergen et al., 1994). It is further broken down over time leaving pheophytin as one of the main residues of the original input of chlorophyll.

Total Phosphorous Lab Procedures—Sub-samples from selected sections were taken and dried. Between 0.015 and 0.02 grams were weighed and homogenized from the dried samples. We used the "standard" persulfate digestion method on the homogenized samples to measure total phosphorous. After these measurements were taken calculations were then made following the method used in Richardson et al. (2008).

^{15}N and ^{13}C isotopic content—Samples were taken from most core sections and analyzed for the stable isotopes of ^{15}N and ^{13}C, and for the nitrogen and carbon mass in dried samples by Dan Lamarra. Further details of the analysis are presented in his section of the report.

Results

Sediment dating and mass accumulation—The ^{210}Pb readings did not provide the hoped for continuous decline in activity with depth. Rather, the top 10-12 (cm) looked like it had been homogenized; however, below that level there was a moderate decrease in the amount of ^{210}Pb (Figure 1). Utilizing an assumption of continuous deposition, the sediment slices were estimated to range in age from 1933 (bottom) to late-2007 (top) (Appendix 1). Based on this dating, sediment deposition rates in Cutler may have increased from near 1000 $g/m^2/year$ to over 4000 $g/m^2/year$ (Figure 2).

However, the Mycore personnel strongly cautioned us about the results based on modeled sediment dates. They stated that they are not confident that the ^{210}Pb profile could be interpreted as indicative of sediment accumulation and that the uncertainty in the dates and accumulation

rates was very large and not due to an analytical issue. Rather, the absence of a monotonic decrease in ^{210}Pb activity was almost certainly due to natural processes (likely sediment disturbance) and the small amount of ^{210}Pb that was available for the analysis.

Phosphorous—There was an increasing trend of phosphorus throughout the sediment core, demonstrating that total phosphorus levels have been increasing over time (Figure 2; Appendix 1). The results from the MyCore lab show that the top 10-12 cm of the core had similar results indicating that these top layers have likely been homogenized. The top ten layers of sediment had a mixed range of values, which may be reflected from the mixing in the top sediments that have occurred. Below the 12-cm, there was a decrease in phosphorus.

Phaeophytin and Chlorophyll—The upper few centimeters of the core contained higher levels of chlorophyll, but there was not a consistent pattern down the core. Likewise, there was no pattern in phaeophytin pigment levels with depth when the results are expressed in ug/g dry sediment of core material (Figure 3). However, if the modeled dates and sediment deposition rates are accepted, the data suggest that the older sections of the core contained lower levels of pheophytin (Figure 4; Appendix 1). We focused on pheophytin because it is the natural degradation product of chlorophyll; therefore it more accurately shows the level of photosynthetic pigments over time. We wouldn't expect to find as much chlorophyll at greater depths because it is broken down rapidly. Between 1980 and 1990 there is a definite decrease in the level of pigments.

Phosphorous and phaeophytin relationship—We expected there to be a correlation between phosphorus content and phaeophytin in the core sediments because of the relationship between nutrients and algal production (Devesa-Rey *et al*, 2009). However, the data showed that there was no relationship between the two parameters (Figure 5; p = 0.58).

Nitrogen, carbon and ^{15}N and ^{13}C isotopes—There was a decrease in the amount of nitrogen per gram of sediment below a depth of about 12 cm (Fig. 6), thus paralleling the changes in lead-210 that occurred below that depth. The carbon content of the sediments also decreased deep in the core, although not as distinctly as the nitrogen content. Although organic matter (N and C) appeared to change somewhat with depth, the analysis of the ^{15}N isotope in the core showed that there was no relationship between levels of this pollution-indicator, and depth in the core (Appendix 1). However, the replicate analyses at some depths indicated quite high variability in the signals measured, so that it might have been difficult to detect a trend if one was present. Additional results of the isotopic analysis are presented in the section by Daniel Lamarra (this report).

Discussion

The core results suggest that total phosphorus levels in Cutler have been increasing over time. This is caused by muiltiple factors. The population of Cache Valley has steadily increased from 29,900 people in 1940 to currently 112,610 people (Swivel, no date). The wastewater treatment plant discharges into Cutler, therefore as the population increases the amount of wastewater that is being discharged increases. The rising population of Cache Valley also means an increase in

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development, which leads to higher levels of erosion and runoff resulting from paved surfaces. Higher levels of erosion will increase phosphorus and can promote algal growth (Ulen and Kalisky 2005). Agricultural practices in Cache Valley, including dairies, are still a common practice; this is also a large contributor of phosphorus and also may have an effect on sedimentation rates depending on the types of agricultural practices that are taking place. Other studies have shown very similar results. Some of these studies show that over time phosphorous and/or chlorophyll increased due to anthropogenic causes (Anderson, 1993; Eliers *et al*. 2004; Bunting *et al*, 2007; Guilizzoni *et al*, 2009).

Although the data suggest that phosphorus levels have risen in Cutler Reservoir, there was not a similar increase in the algal pigment phaeophytin, at least when the results are presented on an μg/g sediment basis. It is likely that the high variability in both the phosphorus and pigment data sets caused this result. However, two other factors could have contributed to a lack of correlation. First, nitrogen, not phosphorus is the nutrient that currently limits algal production in Cutler Reservoir (Abbott 2009), so phytoplankton growth may not increase with increases in phosphorus. Secondly, pigments that accumulate in the sediments are a result of production of both the phytoplankton and periphyton. It is possible that when Cutler Reservoir was clearer and lacked carp, there was high production of periphyton on the sediments, and these would have produced ample quantities of chlorophyll and subsequently phaeophytin.

Cutler was placed on the impaired list under the Clean Water Act because of high phosphorous inputs and low dissolved oxygen. Our results show that over time phosphorous as well as chlorophyll have been increasing. Once nutrient levels get high enough they start to have a negative impact on the different types of life that are dependent on the reservoir. High levels of phosphorus can drive down dissolved oxygen levels therefore making it less sustainable for fish, invertebrates, and as well as plants. With these high nutrient levels, Logan City has started investigating ways to help lower these levels by means of the wastewater treatment plant. The waste water treatment plant contributes about 16% if total phosphorus during the summer months and about 35% during the winter months (DWQ, 2008). If the wastewater treatment plant could implement management that would better filter the nutrients out of the water this would really help lower nutrient levels for the reservoir. However, that is not the only practice that needs to be changed. In order to help take Cutler off the impaired list more changes are needed. For instance, changes in fertilizers for agricultural land could help lower nutrients levels, also making sure that these fertilizers are applied after high runoff during spring would keep these fertilizers from being washed into Cutler. There are a lot of different practices that need to be changed to lower nutrient levels to help improve the water quality of Cutler.

Although our results suggest that nutrient levels in Cutler have been rising, there was a major analytical problems with our project. The Mycore lab stated that there was a high level of uncertainty in the ^{210}Pb analysis. This lowered the confidence for the dates of the sediment core. Because much of Cutler Reservoir is so shallow, it is quite possible that a major storm event, or some other disturbance homogenized much of the sediments. This is a problem in lakes that are not deep enough to strongly stratify (e.g. Eilers et al. 2004). To provide a better understanding of what has happened in the past it may be a good idea to repeat this project, but to do it with

multiple cores throughout the reservoir. This would increase the likelihood of obtaining a core with intact stratigraphy that could provide reliable dates and other information. We were unable to have multiple cores analyzed due to funding and time. We were also limited because we only had a gravity corer available that could penetrate 30-50 cm into the sediments. A piston corer that could take a longer core near the dam would be preferred.

In conclusion, over time phosphorous and chlorophyll levels have increased. Cutler has always had a fair amount of both of these key indicators, but changes in recent years have caused the reservoir to become more eutrophic. These causes appear to human-based, though we do not have enough data to pin-point a certain main source. Increased population is at the root of all the inputs that contribute to high levels of phosphorous. The most obvious input comes from the wastewater treatment plant; if the levels of phosphorous were to be better managed the concentration of phosphorous in the reservoir would go down significantly. In order for this study to be more valuable, further data should be collected and analyzed.

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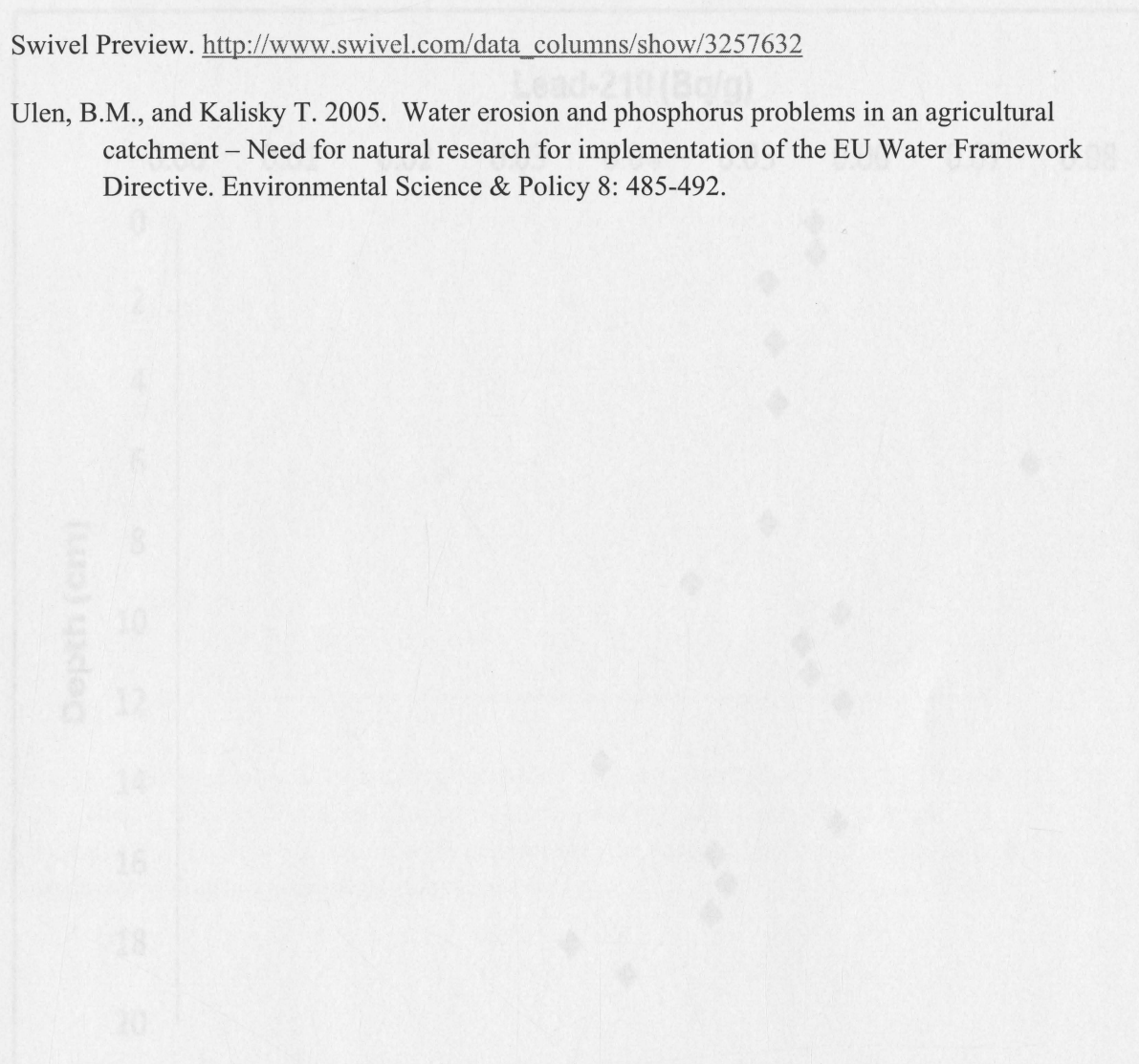
Figure 1. Lead-210 levels versus depth show that the top 10-12 cm has been homogenized. However, below this depth there is a suggestion of a decreasing trend in the Lead-210, which is typical of most samples.
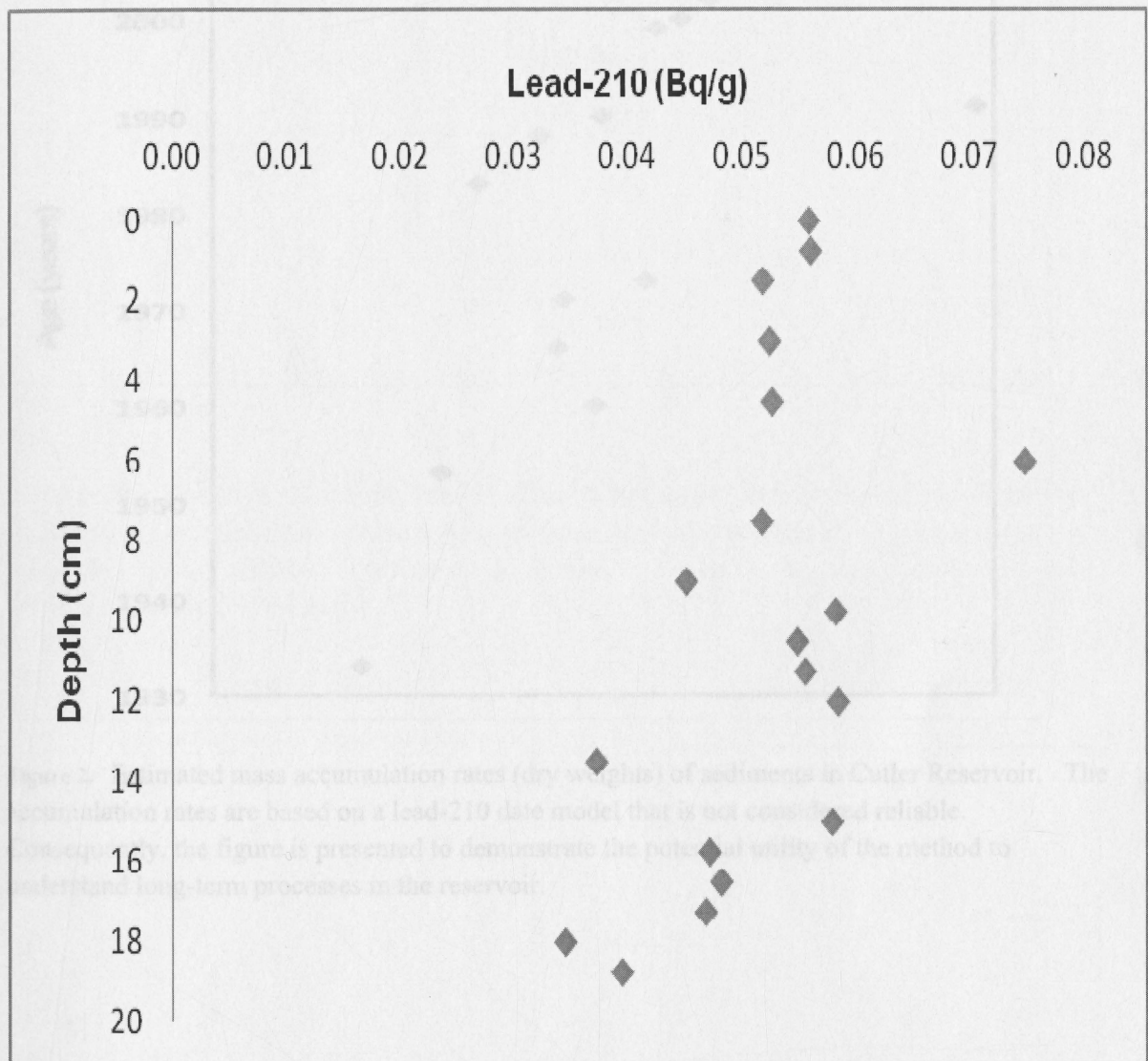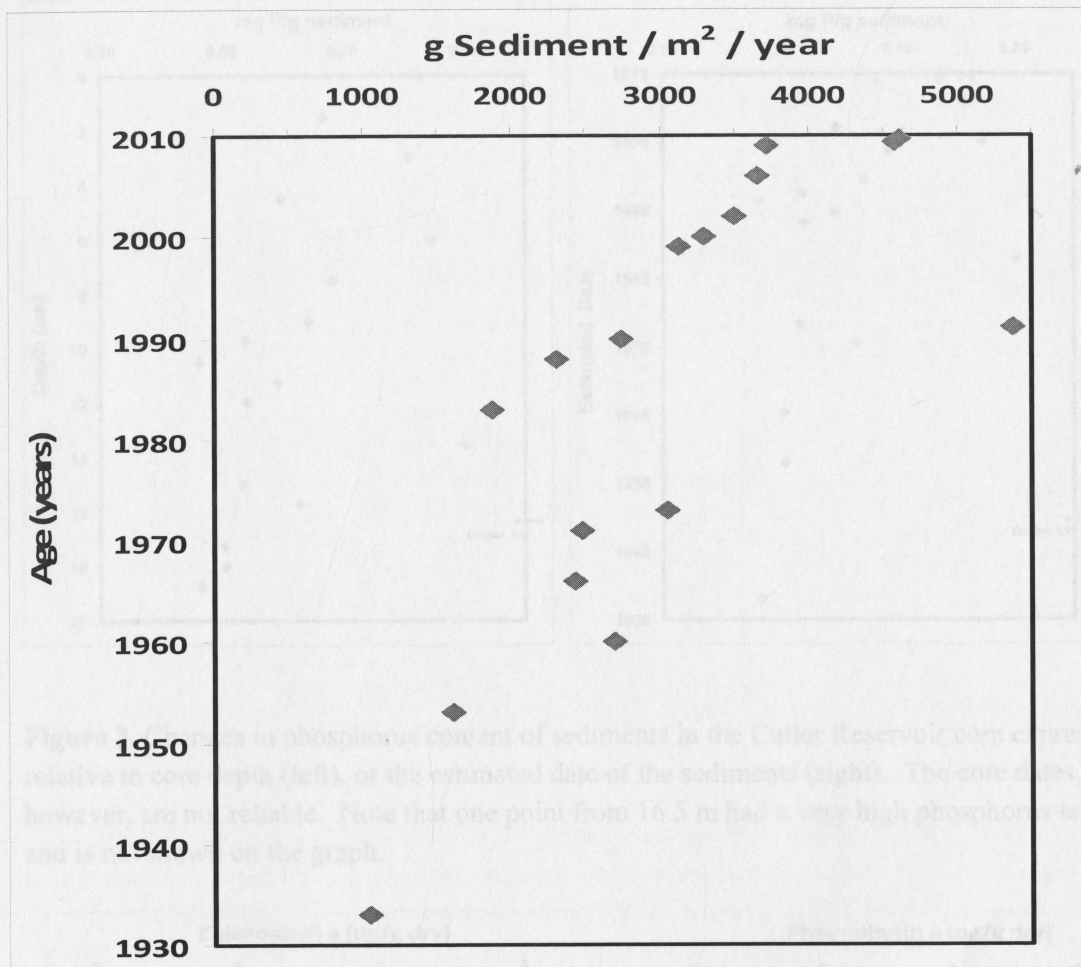
Figure 2. Estimated mass accumulation rates (dry weights) of sediments in Cutler Reservoir. The accumulation rates are based on a lead-210 date model that is not considered reliable. Consequently, the figure is presented to demonstrate the potential utility of the method to understand long-term processes in the reservoir.
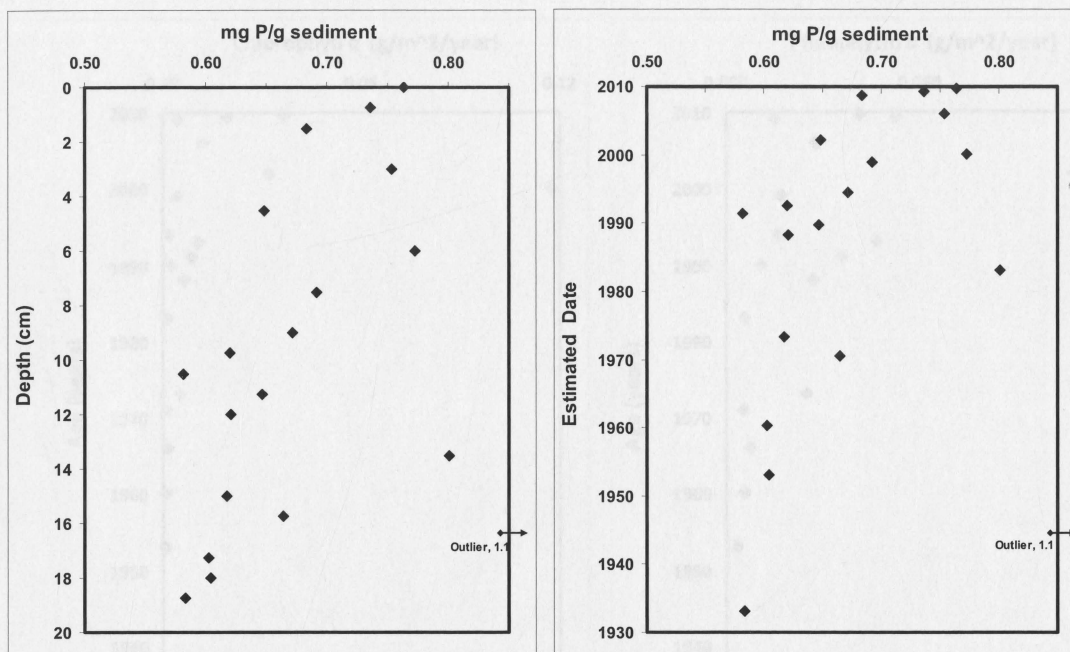
Figure 3. Changes in phosphorus content of sediments in the Cutler Reservoir core expressed relative to core depth (left), or the estimated date of the sediments (right). The core dates, however, are not reliable. Note that one point from 16.5 m had a very high phosphorus level and is not shown on the graph.
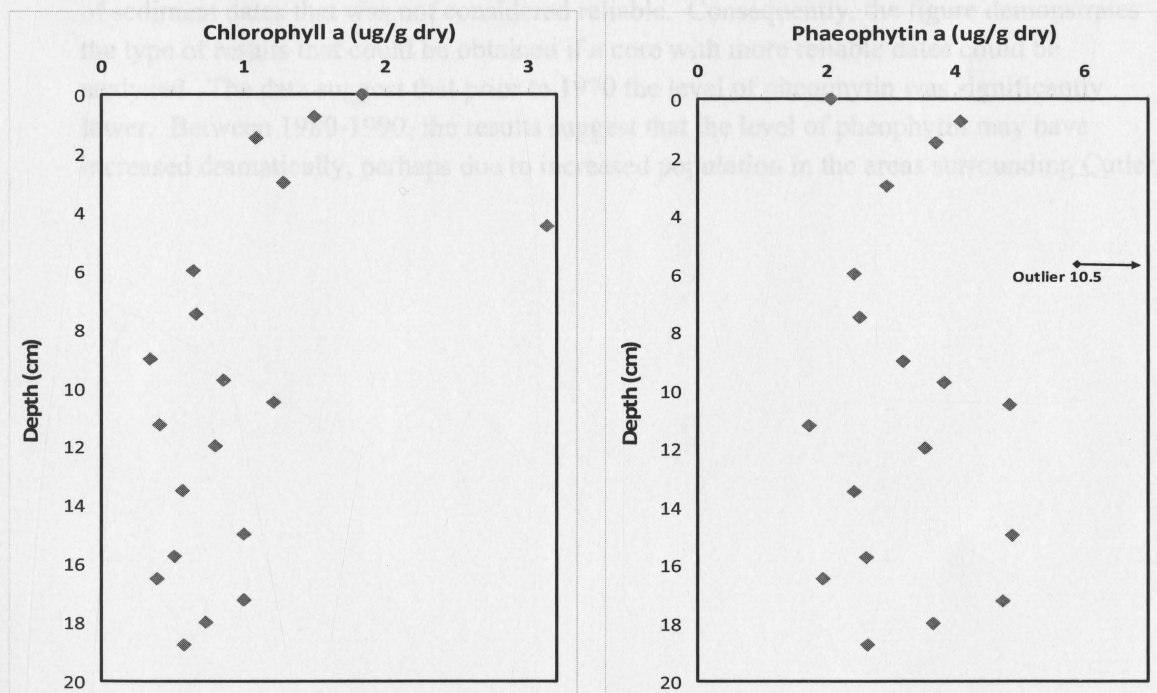

Figure 4. Changes in chlorophyll (left) and phaeophytin (right) with depth in a sediment core taken from Cutler Reservoir.
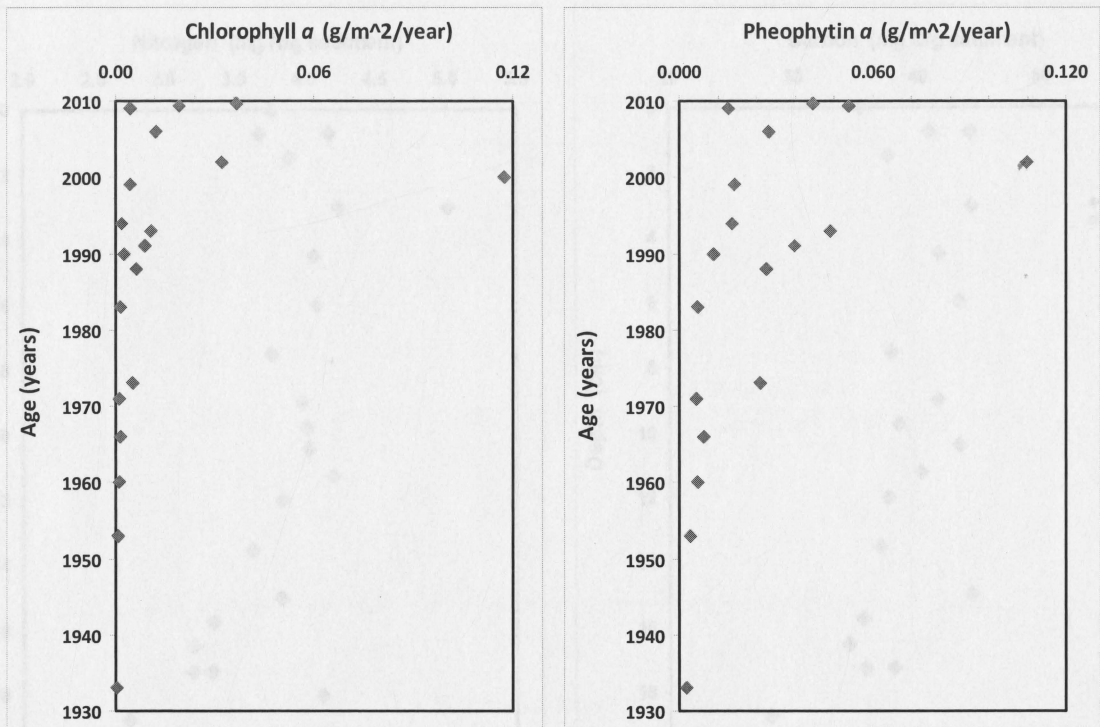
Figure 5. Changes in pigment deposition rates estimated from the Cutler Reservoir core. The rates were based on sediment accumulation rates, which in turn, were based on a model of sediment dates that was not considered reliable. Consequently, the figure demonstrates the type of results that could be obtained if a core with more reliable dates could be analyzed. The data suggest that prior to 1970 the level of pheophytin was significantly lower. Between 1980-1990, the results suggest that the level of pheophytin may have increased dramatically, perhaps due to increased population in the areas surrounding Cutler.

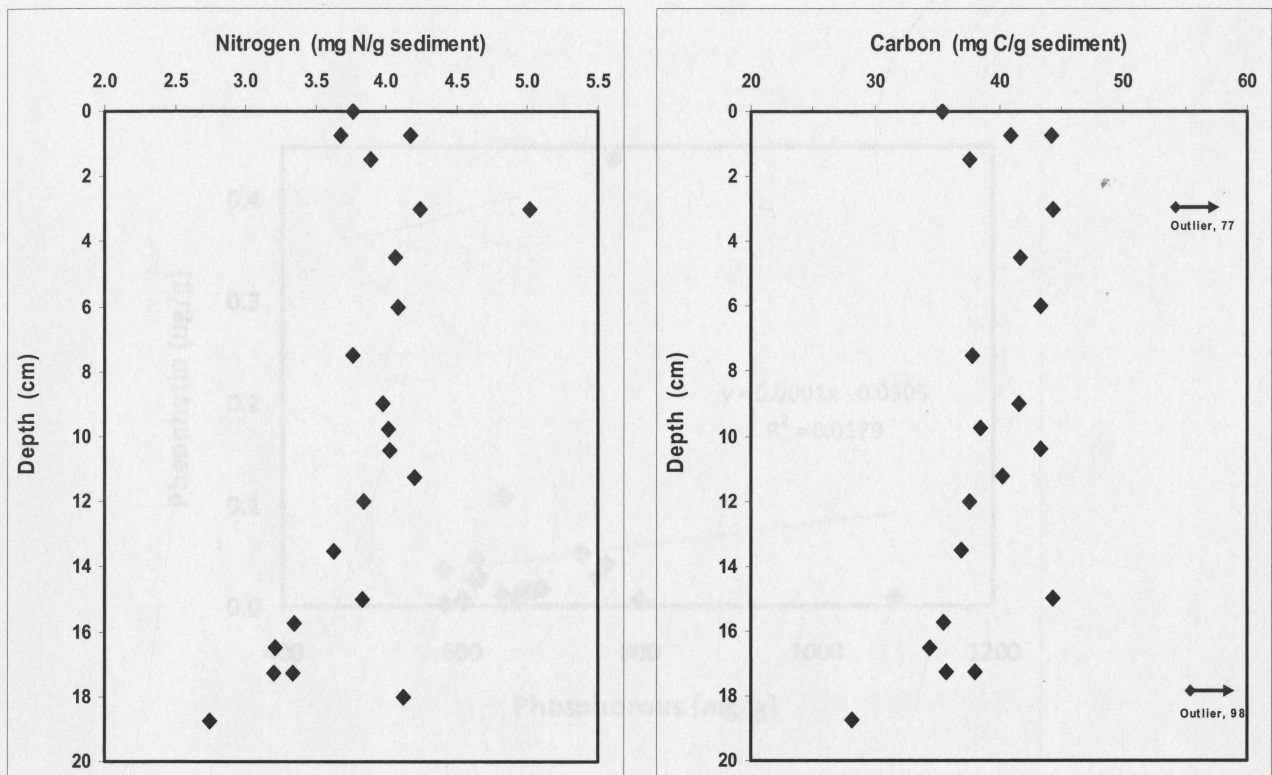**Figure 6.** Concentrations of N (left) and carbon (right) at different depths in the sediment core taken from Cutler Reservoir. Note how changes parallel those of lead-210 isotopic activity (Figure 1). Data adapted from Dan Lamarra.
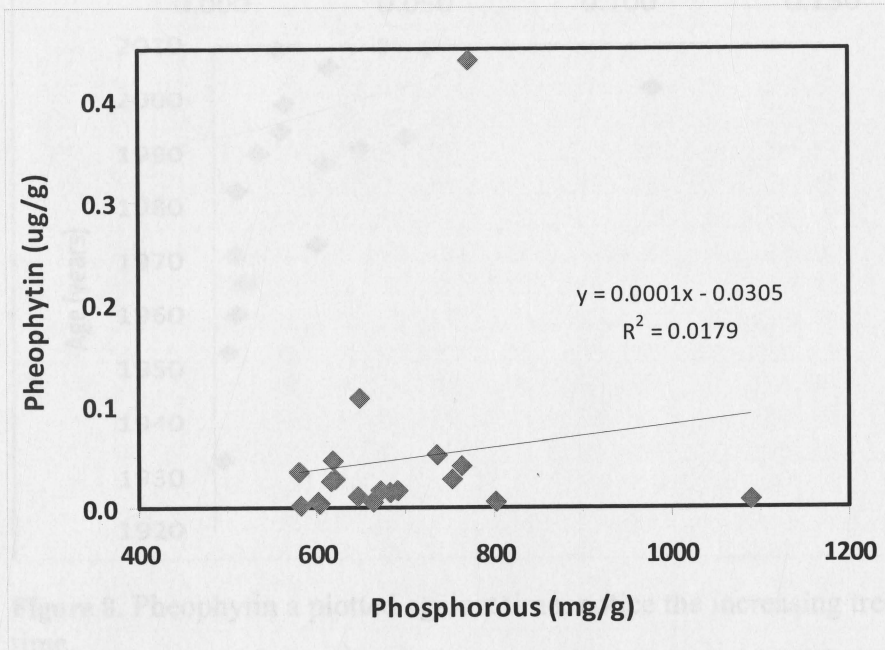
Figure 7. Relationship between phosphorus content and phaeophytin content in different Cutler Reservoir core slices. There was no relationship between phosphorus and pigment concentrations.
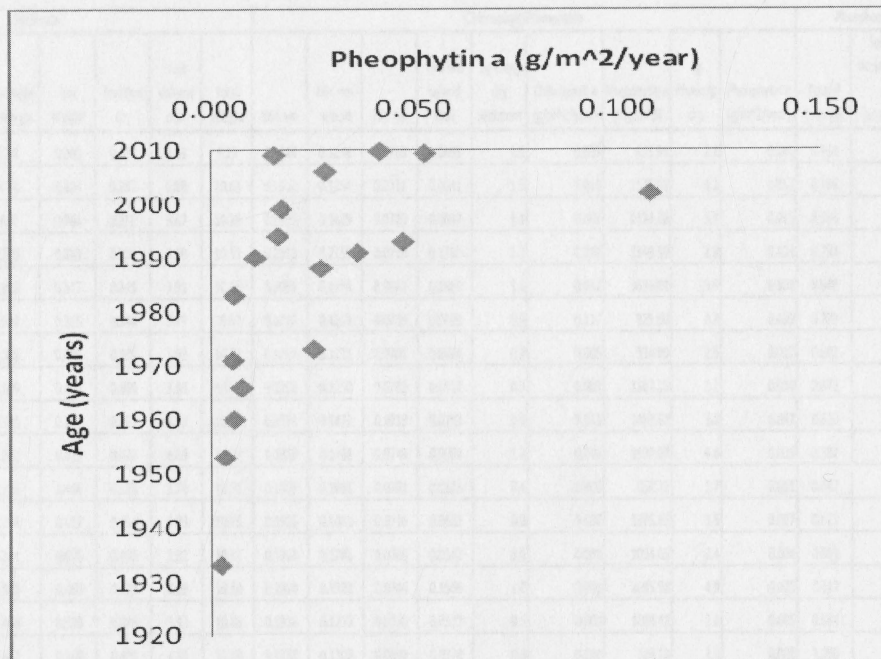
Figure 8. Pheophytin a plotted against time, notice the increasing trend with time.

Core Strata								Chlorophyll/Pheophytin									Phosphorus		Isotopic Analysis			
Top (cm)	Bottom (cm)	Age at top of section (year)	Sub-sample Wet Weight	Dry Weight	Fraction dry	Wet Volume equ.	Total Weight	664 nm	664 nm w/acid	750 nm	664 nm w/acid final	ug Chl/g dry sediment	Chlorophyll a (g/m^2/year)	Pheophytin a (mg/m^3)	ug Phaeo/g dry	Pheophytin a (g/m^2/year)	Total P (mgP/g)	Sediment accumulation rate (g/m2/year)	d15N	d13C	mg N/g	mg C/g
0.00	0.75	2009.7	0.692	0.205	0.297	2.49	9.22	0.1444	0.1248	0.0788	0.0465	1.8	0.037	615.94	2.1	0.041	0.764	4906	5.82	-27.58	3.76	35.46
0.75	1.50	2009.3	1.068	0.307	0.287	3.88	10.63	0.1512	0.1394	0.0711	0.0642	1.5	0.019	1172.61	4.1	0.053	0.736	4585	5.57/7.1	27.7/-27.3	4.2/3.7	44.3/41.0
1.50	2.25	2009	2.687	0.964	0.359	3.67	10.39	0.1555	0.1439	0.0720	0.0687	1.1	0.005	1324.36	3.7	0.015	0.684	3719	5.6	-27.6	3.9	37.7
3.00	3.75	2006	1.103	0.883	0.801	3.60	10.32	0.2393	0.2159	0.0718	0.1350	1.3	0.012	2349.50	2.9	0.028	0.754	3505	6.6/6.1	27.7/-27.1	4.2/5.0	44.4/76.0
4.50	5.25	2002	0.918	0.317	0.345	3.91	10.65	0.3093	0.2755	0.0862	0.1889	3.1	0.032	3636.01	10.5	0.108	0.648	3133	6.3	-27.7	4.1	41.7
6.00	6.75	2000	0.924	0.315	0.340	3.79	10.52	0.1267	0.1204	0.0748	0.0419	0.6	0.117	825.90	2.4	0.439	0.773	55784	6.3	-27.5	4.1	43.3
7.50	8.25	1999	1.203	0.446	0.371	3.92	10.67	0.1283	0.1211	0.0701	0.0474	0.7	0.005	934.80	2.5	0.017	0.692	2735	6.3	-27.5	3.8	37.8
9.00	9.75	1994	1.105	0.404	0.366	3.84	10.56	0.1292	0.1250	0.0702	0.0517	0.3	0.002	1167.25	3.2	0.016	0.672	1873	7.5	-27.5	4.0	41.6
9.75	10.50	1993	0.746	0.283	0.379	3.59	10.32	0.1561	0.1472	0.0718	0.0713	0.9	0.011	1453.62	3.8	0.047	0.620	3051	7.7	-27.6	4.0	38.5
10.50	11.25	1991	1.002	0.413	0.412	4.03	10.77	0.1920	0.1788	0.0743	0.0999	1.2	0.009	1997.07	4.8	0.036	0.582	2481	5.2	-27.7	4.0	43.3
11.25	12.00	1990	1.176	0.444	0.378	3.78	10.51	0.1107	0.1061	0.0701	0.0324	0.4	0.003	652.72	1.7	0.011	0.647	2432	6.3	-27.5	4.2	40.2
12.00	12.75	1988	1.058	0.417	0.394	4.03	10.75	0.1525	0.1442	0.0716	0.0682	0.8	0.006	1386.85	3.5	0.027	0.621	2697	5.9	-27.2	3.8	37.7
13.50	14.25	1983	1.344	0.605	0.450	3.82	10.53	0.1361	0.1281	0.0708	0.0542	0.6	0.001	1094.05	2.4	0.006	0.801	1045	6.4	-27.5	3.6	37.0
15.00	15.75	1973	0.958	0.409	0.427	4.09	10.80	0.1910	0.1783	0.0746	0.1006	1.0	0.005	2085.98	4.9	0.025	0.617	1647	6.4	-27.3	3.8	44.3
15.75	16.50	1971	1.408	0.589	0.419	3.92	10.66	0.1336	0.1273	0.0710	0.0527	0.5	0.001	1099.41	2.6	0.005	0.664	952	8.8	-27.3	3.3	35.6
16.50	17.25	1966	0.662	0.268	0.405	4.35	11.09	0.1153	0.1102	0.0692	0.0378	0.4	0.002	784.12	1.9	0.008	1.090	856	6.3	-27.3	3.2	34.4
17.25	18.00	1960	1.722	0.755	0.438	4.06	10.78	0.1901	0.1775	0.0733	0.1007	1.0	0.001	2077.77	4.7	0.006	0.602	686	8.1/5.2	27.2/-27.2	3.3/3.2	38.1/35.7
18.00	18.75	1953	1.279	0.680	0.531	4.33	11.05	0.1836	0.1725	0.0752	0.0936	0.7	0.001	1946.73	3.7	0.003	0.604	385	5.6	-26.2	4.1	98.4
18.75	19.50	1933	0.847	0.395	0.467	6.51	13.25	0.1432	0.1359	0.0714	0.0604	0.6	0.0005	1238.31	2.7	0.002	0.583	235	6.8	-27.0	2.7	28.2

Appendix 1. Characteristics of a sediment core taken from Cutler Reservoir. For the isotopic analyses, two replicate samples were processed at three depths in the core.