UNIVERSITY OF CALIFORNIA
RIVERSIDE

A Characterization of the Phytoplankton, Zooplankton, and Benthic Invertebrate Communities of Lake Elsinore

A Thesis submitted in partial satisfaction
of the requirements for the degree of

Master of Science

in

Environmental Sciences

by

Michelle Elaine Tobin

December 2011

Thesis Committee:
Dr. Michael Anderson, Chairperson
Dr. William Walton
Dr. James Sickman

The Thesis of Michelle Elaine Tobin is approved:

Committee Chairperson

University of California, Riverside

Acknowledgements

Thank you very much to Dr. Michael Anderson who took me on as a master's student and allowed me to work with and learn about critters that I otherwise would remain sadly ignorant of. I enjoyed the experience immensely and appreciated all of the advice and knowledge that you gave me. I am also grateful for the advice, help, and support of my other committee members, Dr. James Sickman and Dr. William Walton, and of everyone in the Environmental Sciences Department, especially Paul Koster and my lab-mates Joy McCullough, Nicole Reid, and my sister Jennifer Tobin. Thanks as well to Dee Lucero and Woody Smith who did the more technical parts of my sample processing and Jeff Janik who did a very helpful, thorough analysis of the phytoplankton community. I could not have done all of this without you guys.

ABSTRACT OF THE THESIS

A Characterization of the Phytoplankton, Zooplankton, and Benthic Invertebrate Communities of Lake Elsinore

by

Michelle Elaine Tobin

Master of Science, Graduate Program in Environmental Sciences
University of California, Riverside, December, 2011
Dr. Michael A. Anderson, Chairperson

Lake Elsinore is a shallow, polymictic lake in the southwest corner of Riverside County in Southern California. It has a history of poor water quality (algal blooms, low dissolved oxygen, and periodic fish kills) and was listed on the State of California's 303d list for impairments due to nutrients and other factors. Water column measurements and nutrient concentrations have been closely monitored since 2000. Less is known about the biological condition of the lake. A 1-yr biological monitoring study was conducted in 2010 to quantify the abundance, diversity, and richness of phytoplankton, zooplankton, and benthic invertebrates in Lake Elsinore. This report summarizes the results of that study and also includes findings from an experiment to determine the response of the phytoplankton community to nutrient inputs.

The phytoplankton community exhibited very strong variation in Simpson's diversity (~0.2 -0.8) and biomass (~2,000 -30,000 mg m^{-3}) during the study period. The lowest diversities and highest biomasses occurred during the 2010 summer bloom of *Pseudanabaena limnetica*. The highest diversities and lower overall phytoplankton biomasses occurred during winter and spring of 2010 when green algae and/or diatoms were a prominent component of the biomass.

The zooplankton community was comprised primarily of smaller zooplankters with rotifers dominating summer through fall and cyclopoid copepods being more prominent during cooler seasons. Large cladocerans were scarce and only found early in 2010 after heavy rainfall caused Canyon Lake to spill over into Lake Elsinore.

The benthic community was the most impoverished, consisting predominantly of organisms (Chironomids and *Ilyocryptus*) with red hemoglobin-like pigments that aid in the binding and transfer of oxygen. This reflects the low DO concentrations in the bottom waters and sediments of Lake Elsinore during much of the year. Invertebrate densities varied strongly between sites and sampling dates and were consistently higher at the shallower sites with higher DO and lower organic carbon concentrations in the sediments.

Results from the nutrient addition experiment suggest that phytoplankton were mainly N-limited, but that the diatoms were limited mostly by P. They also suggest that an excess of N relative to P contributes to the formation of *Pseudanabaena limnetica* blooms.

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

CHAPTER ONE

INTRODUCTION

The nutrient inputs, residence time, and depth of a lake can have significant influences on its general water quality and trophic status. A lake's trophic state reflects its overall level of productivity, and, thus, is a reflection of the nutrient levels, aquatic communities, and conditions in the lake. There are three main trophic states used to classify aquatic systems (Wetzel, 2001). At the lowest trophic status, oligotrophic lakes typically have lower nutrient inputs and low nutrient levels (TP concentrations averaging 8.0 mg m^{-3}, TN concentrations averaging 661 mg m^{-3}, Wetzel, 2001) in the lake, which support only limited productivity (mean maximum chlorophyll a production of 4.2 µg L^{-1}, Wetzel, 2001) and the lowest abundances of organisms. High elevation lakes in regions dominated by primary rocks are often oligotrophic (Wetzel, 2001) because there is little input of weathering products from runoff in the watershed that could bring in nutrients from land. On the other end of the trophic scale, eutrophic lakes are highly productive (mean maximum chlorophyll a production of at least 42.6 µg L^{-1}, Wetzel, 2001) and nutrient rich (TP concentrations averaging 84.4 mg m^{-3}, TN concentrations averaging 1875 mg m^{-3}, Wetzel, 2001), with higher nutrient inputs and/or nutrient retention in the lake. They typically contain higher abundances of fast-growing organisms that are tolerant of poor water quality. Shallow, lowland lakes in agricultural and/or highly populated areas are more likely to have high nutrient levels than deep, high elevation lakes for several reasons. They have a much larger watershed area draining into them than mountain lakes at the top of a watershed, so there is a greater potential for nutrient input from land and from primary production upstream. Agricultural and urban areas

also have relatively greater amounts of fertilizers, wastes, and anthropogenic chemicals that can enter the watershed from drains and add to the nutrient load of waters downstream. Shallow lakes also have greater surface water and sediment areas relative to the lake volume. This means that the nutrients that come into the lake from the surface (via aerosols, acidic deposition, or nitrogen fixation by cyanobacteria) or are recycled from stores in the sediments (internal nutrient cycling) have a relatively smaller volume of water to be mixed into, leading to higher concentrations in the water column.

The length of time that water typically spends in a lake or reservoir can also influence its productivity and trophic status. In lakes with short (days to a few weeks) residence times (meaning that the average water molecule does not spend much time in the lake since inflow and outflow quickly cycle water through), nutrients move relatively quickly through the system and are flushed out of the lake before they can accumulate in the food web or the sediments. In lakes with long residence times (months to years), however, the nutrients that are washed in with inflow remain in the system for a long time before being flushed out and, thus, have ample time to accumulate in the system through uptake by organisms and deposition and recycling in the sediments. Evapoconcentration over longer time scales may also significantly increase the concentration of the nutrients in the system. In these ways, long residence times often promote high nutrient, eutrophic systems.

Aside from the chemical and physical measurements that are often taken to assess water quality (nutrient concentrations, chlorophyll fluorescence, dissolved oxygen concentration, etc.), another way to look at the quality of water in a particular area is to look at the organisms present. Different organisms have different tolerances for conditions in a lake including nutrient concentrations (or trophic status), dissolved

oxygen, salinity, and temperature. Aquatic insects have been widely used for water quality assessments since the early twentieth century due to their ubiquity, generally high abundances and easy collection, and the vast range of conditions they can survive in (Merritt et al., 2008). The EPT (Ephemeroptera, Plecoptera, Trichoptera or, more commonly, mayfly, stonefly, caddisfly) richness and rapid bioassessment protocols are two methods that are commonly used to assess water quality through the aquatic insect community (Merritt et al., 2008).

Bioassessments can be extended beyond insects. Phytoplankton (microscopic, photosynthetic organisms) and zooplankton (larger, heterotrophic planktonic animals) are also potentially abundant in lakes and, as such prominent members of the aquatic community, may be useful indicators of the conditions in the lake (Della Bella & Mancini, 2009; Gannon and Stemberger, 1978; McCormick & Cairns, 1994; Stoermer, 1978) as well as influence conditions in the water column and, thus, water quality (Wetzel, 2001). Diatoms, green algae, and cyanobacteria are three common phytoplankton groups that are often observed to peak sequentially. Diatoms tend to prefer cool conditions (5-20^{o}C, Stevenson et al., 1996 and references therein) and require silica for their intricate outer covering (frustule). They may be found in lakes in late winter or early spring after rains wash nutrients and Si-rich sediments in from the watershed (Wetzel, 2001). Once Si is depleted and the water warms somewhat to15-30^{o}C (Stevenson et al., 1996 and references therein), green algae (which generally prefer somewhat higher temperatures than diatoms and do not require Si) can out compete diatoms for the nutrients and light they need and begin to peak. A greater temperature increase in the lake water later in the year may exceed the upper limit of thermal tolerance for the green algae and N may start to become depleted by the phytoplankton. At this time, cyanobacteria, which

generally have the highest thermal preferences (temperatures >30°C, Stevenson et al., 1996 and references therein) and contain some species that can fix atmospheric N_2, often become the best competitors (Wetzel, 2001).

Cyanobacteria are notorious for taking advantage of high nutrient levels, such as those in eutrophic systems (when P is not limiting), to form algal blooms that are unpleasant and potentially toxic. Furthermore, N-fixing cyanobacteria may add N to an aquatic system (Cook et al., 2010), increasing the already high nutrient levels. All phytoplankton (including bloom- forming cyanobacteria) fix carbon (photosynthesize) using light-absorbing photosynthetic pigments, require light for photosynthesis, and are limited to the photic zone in the water column (the region in the lake with ample light penetration for photosynthesis). Therefore, phytoplankton biomass in the surface waters of the lake can influence the depth to which light and its energy/heat can penetrate the water column as well as dissolved oxygen levels in the lake (through photosynthesis, which generates O_2, and respiration, which consumes O_2). The productivity of phytoplankton is positively correlated with oxygen deficits in the hypolimnion (Wetzel, 2001). When the organisms from blooms die and sink to the sediments, aerobic decomposition of their bodies by bacteria consumes dissolved oxygen and can result in anoxia in the isolated bottom waters of a stratified lake (hypolimnion). An anoxic hypolimnion is unsuitable for benthic insects that are important food sources for fish, birds, and other wildlife, and potentially rotten-smelling reduced gases can build up at the bottom of the lake. Their unpleasant odor can be noticeable for miles downwind when these gases are released to the atmosphere, an unpleasant experience for residents of the area. Oxygen depletion resulting from processes such as those

described above is also responsible for the massive fish kills that have occurred in eutrophic lakes with algal blooms.

Phytoplankton are also important for the quality of a lake because they are food for other organisms, including zooplankton. Many types of zooplankton graze upon phytoplankton and, consequently, require phytoplankton to be present at or above their threshold food concentrations. However, certain types of phytoplankton provide better food for zooplankton than others. Cyanobacteria, for example, may be toxic to zooplankton or too long and cumbersome to be taken into the food groove, thus inhibiting filtering and decreasing survivorship of zooplankters, especially large cladocerans such as *Daphnia* (Demott, 1999; Gliwicz, 1990 and references therein; Sahuquillo et al., 2007). Therefore, the nutrient and other conditions in the lake that influence phytoplankton populations also influence zooplankton populations and, by feeding on the phytoplankton, zooplankton can influence their concentrations. Large *Daphnia* species tend to be particularly efficient at filtering phytoplankton out of the water column relative to other zooplankton genera and are desirable for their ability to keep the phytoplankton populations in check and maintain water clarity (Gliwicz, 1990 and references therein; Vakkilainen et al., 2004). Therefore, a large, healthy *Daphnia* population in a lake is indicative of a healthy lake that is more likely to have clearer water. Smaller zooplankters, on the other hand, tend to be more tolerant of cyanobacterial filaments and toxins and other polluting contaminants (Chang et al., 2005; Hanazato, 1991; Havens & Hanazato, 1993; Kirk & Gilbert, 1992; Walz, 1995 & references therein) and may be less efficient at controlling algal biomass (Gliwicz, 1990; Vakkilainen et al., 2004). Therefore, a lake dominated by small zooplankton, such as

rotifers and small cladocerans, could indicate a less ecologically healthy lake that may be plagued by eutrophy.

Considering the above, a lake's health is clearly a function of its ecology as well as nutrient levels. The importance of ecological monitoring for maintaining water resources is increasingly being recognized. One of the most notable examples of this is the European Union's Water Framework Directive. Two main goals of this directive are to protect and improve the aquatic environment and contribute to sustainable, balanced, and equitable water use through regulating, monitoring, and managing both chemical and ecological aspects of entire, interconnected water systems rather than single water bodies or streams (Lanz and Scheuer, 2001). Under the Water Framework Directive, the status of waters is assessed primarily based on the aquatic biota present (Hering et al., 2010; Lanz and Scheuer, 2001). Since its adoption in 2000, benefits have arisen from the implementation of the Water Framework directive. Much knowledge has been gained on the ecology of surface waters in Europe, new methods for sampling and investigating aquatic ecosystems have been developed, and many procedures for sampling and analyzing these ecosystems have been standardized across Europe (Hering et al., 2010). However, since it generally takes at least a decade for water bodies to fully recover from prolonged degradation (Hering et al., 2010), it is still too soon to know whether or not the directive is achieving success towards restoring ecological and water quality. Nonetheless, valuable insight toward preserving aquatic ecosystems and water resources could be gained in other regions of the world by adopting a similar approach to assessing water quality based on biota rather than simply relying on chemical analysis.

Lakes with histories of water quality issues likely experience such problems due, at least in part, to the functioning of their aquatic communities. Lake Elsinore, a naturally formed, shallow, eutrophic lake in Southern California, is one such lake. Among other ailments, this lake has been plagued by cyanobacterial blooms and fish kills. Previous and ongoing monitoring at the lake has substantially improved understanding of the water quality in Lake Elsinore as based upon traditional types of analysis of chemical and physical parameters. However, no previous study has simultaneously analyzed the phytoplankton, zooplankton, and benthic invertebrate communities in the lake to see what they may indicate about conditions at the time of sampling or the application of methods to improve water and ecological quality.

A biological monitoring program was thus developed to assess the biological health and ecological functioning of Lake Elsinore and the effects of nutrient additions on the phytoplankton community. A series of different measurements were conducted that evaluate the zooplankton, benthic invertebrate, and phytoplankton communities in the lake, and consider them in light of the physical, chemical, and water quality conditions. This thesis covers the types and amounts of organisms present, how these organisms tended to covary with each other, and how these observations compared with those from previous studies. This knowledge can be used to gain new insight about what to expect under certain conditions (such as periods with relatively high nutrient concentrations or low water transparency) and, thus, what a relatively quick assessment of one or more of the three communities being sampled (phytoplankton, zooplankton, benthic invertebrates) indicates about conditions in the lake. With this work, we provide an ecological benchmark so that future aquatic communities in Lake Elsinore can be

compared to those in this study along with any other chemical or physical measurements made during the relevant time periods and so that water quality problems and potential solutions can be linked to the lake's ecology as well as chemistry.

REFERENCES

Chang, K.H., M. Sakamoto, and T. Hanazato. 2005. Impact of pesticide application on zooplankton communities with different densities of invertebrate predators: An experimental analysis using small-scale mesocosms, Aquatic Toxicology 72:373-382.

Cook, P.L.M., K.T. Aldridge, S. Lamontagne, and J.D. Brooks. 2010. Retention of nitrogen, phosphorus and silicon in a large semi-arid riverine lake system, Biogeochemistry 99:49-63.

Della Bella, V. and L. Mancini. 2009. Freshwater diatom and macroinvertebrate diversity of coastal permanent ponds along a gradient of human impact in a Mediterranean eco-region, Hydrobiologia 634:25-41.

DeMott, W.R. 1999. Foraging strategies and growth inhibition in five daphnids feeding on mixtures of a toxic cyanobacterium and a green algae, Freshwater Biology 42:263-274.

Gannon, J.E., and R.S. Stemberger. 1978. Zooplankton (especially crustaceans and rotifers) as indicators of water quality, Transactions of the American Microscopical Society 97:16-35.

Gliwicz, Z.M. 1990. Why do cladocerans fail to control algal blooms, Hydrobiologia 200:83-97.

Hanazato, T. 1991. Effects of repeated application of carbaryl on zooplankton communities in experimental ponds with or without the predator *Chaoborus*, Environmental Pollution 74:309-324.

Havens, K.E. and T. Hanazato. 1993. Zooplankton community responses to chemical stressors – a comparison of results from acidification and pesticide contamination research, Environmental Pollution 82:277-288.

Hering, D. et al. 2010. The European Water Framework Directive at the age of 10: A critical review of the achievements with recommendations for the future, Science of the Total Environment 408:4007-4019.

Kirk, K.L., and J.J. Gilbert. 1992. Variation in herbivore response to chemical defenses – zooplankton foraging on toxic cyanobacteria, Ecology 73:2208-2217.

Lanz, K., and S. Scheuer. 2001. EEB Handbook on EU Water Policy under the Water Framework Directive, The European Environmental Bureau, Hontelez, J. editor.

McCormick, P.V. and J. Cairns Jr. 1994. Algae as indicators of environmental change, Journal of Applied Phycology 6:509-526.

Merritt, R.W., K.W. Cummins, and M.B. Berg (Eds.). 2008. An Introduction to the Aquatic Insects of North America, Fourth Edition, Kendall Hunt, pg. 63.

Sahuquillo, M., M.G.G. Melao, and M.R. Miracle. 2007. Low filtering rates of *Daphnia magna* in a hypereutrophic lake: Laboratory and in situ experiments using synthetic microspheres, Hydrobiologia 594:141-152.

Stevenson, R.J., M.L. Bothwell, and R.L. Lowe (eds.). 1996. Algal Ecology: Freshwater Benthic Ecosystems, Elsevier Academic Press, pg. 163.

Stoermer, E.F. 1978. Phytoplankton assemblages as indicators of water quality in the Laurentian Great Lakes, Transactions of the American Microscopical Society 97:2-16.

Vakkilainen et al. 2004. Response of zooplankton to nutrient enrichment and fish in shallow lakes: a pan-European mesocosm experiment, Freshwater Biology 49:1619-1632.

Walz, N. 1995. Rotifer populations in plankton communities – energetics and life-history strategies, Experientia 51:437-453.

Wetzel, R.G. 2001. Limnology: Lake and River Ecosystems: Third Edition, Elsevier Academic Press.

CHAPTER TWO

PHYTOPLANKTON, ZOOPLANKTON, & BENTHIC INVERTEBRATES IN LAKE ELSINORE

INTRODUCTION

Lake Elsinore is a shallow, tectonically-formed, polymictic lake in southwestern Riverside County in Southern California (33° 39.5' latitude and 117° 21.0' longitude) that serves as a wildlife habitat and recreation site. The lake is situated in a down-faulted trough at the base of the Santa Ana Mountains, which form the northernmost range of the Peninsular Ranges Province (FERC, 2006). As with other lakes in the region, Lake Elsinore is subject to strong annual variations in lake level as a result of the high evaporation rate for the region (1.4 m/yr) coupled with frequent droughts and occasional El Nino events (Lawson and Anderson, 2007; Kirby et al., 2007).

Lake Elsinore has experienced poor water quality (including algal blooms, low dissolved oxygen (DO) concentrations, and periodic fish kills) that is coupled in a complex way to the hydrologic cycle of the region, and results from external loading of nutrients from the watershed and nutrient recycling from the sediments. As the terminus for the approximately 2000 km^2 San Jacinto watershed (much of which is agricultural and urban in the lower reaches) and an essentially closed basin, Lake Elsinore periodically receives relatively high nutrient inputs that tend to remain in the lake and concentrate, particularly in the sediments. During periods of unusually high rainfall, the lake spills over and some nutrients get flushed out with the water into the Santa Ana River (San Jacinto River Watershed Council, 2005).

The Regional Board first placed Lake Elsinore on the 303(d) list of impaired waters in 1994. A Nutrient Total Maximum Daily Load (TMDL) for the lake was developed and incorporated into Amendments to the Basin Plan in 2005. Water quality measurements have been regularly made at the lake for the past decade, quantifying basic characteristics such as temperature, dissolved oxygen (DO), electrical conductance, and pH, as well as total and dissolved nutrients and chlorophyll a concentrations. These measurements are now part of the In-Lake Nutrient Monitoring Program required by the TMDL.

Since phytoplankton populations can influence dissolved oxygen levels in the water column (through photosynthesis, respiration, and the aerobic decomposition of their dead bodies) and organisms can influence each other and nutrient levels through trophic interactions and the cycling of elements through the food web, a lake's condition is largely a function of its ecology as well as the nutrient levels entering and cycling in the lake. Previous and ongoing monitoring at the lake has substantially improved understanding of the water quality in Lake Elsinore. Additional measurements are needed, however, to fully evaluate attainment of the Warm Freshwater Habitat (WARM) beneficial use that includes preservation and enhancement of aquatic habitats, vegetation, and fish and wildlife, including invertebrates. Improved understanding of the food web in the lake will also provide important new information about biological (e.g., top-down) effects on water quality and identify further opportunities for and effects of biomanipulation at the lake. No previous study has simultaneously analyzed the phytoplankton, zooplankton, and benthic invertebrate communities in the lake to see what they may indicate about conditions at the time of sampling or the application of

methods (such as nutrient manipulation and control or biomanipulation) to improve water and ecological quality.

A biological monitoring program was developed to assess the biological health and ecological functioning of Lake Elsinore. A series of measurements were conducted that evaluate the zooplankton, benthic invertebrate, and phytoplankton communities in the lake, and consider them in light of physical, chemical, and water quality conditions. Results from the monitoring program cover the types and amounts of organisms present, how these organisms tended to covary with each other, and how these observations compared with those from previous studies.. This knowledge can then be used to gain valuable new insight about what to expect under certain conditions (such as relatively high nutrient concentrations and low water transparency) and, thus, what a quick assessment of one or more of the three communities being sampled (phytoplankton, zooplankton, benthic invertebrates) indicates about conditions in the lake. We would also like to provide an ecological benchmark so that future aquatic communities and chemical or physical measurements in Lake Elsinore can be compared to those in this study.

APPROACH

Study Sites and Sampling

Biological sampling for this project was initiated on November 17, 2009, and was conducted bimonthly through the end of 2010. In 2010, samples were collected on February 17, April 15, June 22, August 19, October 21, and December 16 from the 3

TMDL sampling stations (Fig. 2.1, yellow squares). A 4th site was added in April 2010 and sampled on all subsequent dates for benthic invertebrates and zooplankton (Fig. 2.1). Locations and average depths for the sampling sites are shown in Table 2.1.

Table 2.1. Locations and average depths of sampling sites

	Latitude	Longitude	Depth (m)
Site 1	33°40.278' N	117°21.931' W	6.2
Site 2	33°39.844' N	117°21.439' W	7.4
Site 3	33°39.392' N	117°20.793' W	6.2
Site 4	33°39.749' N	117°20.030' W	2.7

Fig. 2.1. Study site, showing sampling locations for phytoplankton, zooplankton and benthic invertebrates (yellow squares)

Triplicate samples were collected and composited from each of the 3 sites (4 for zooplankton and benthic invertebrates beginning in April 2010) (Fig. 2.1). The northernmost site (site 1) served as the QA site for our sampling; as a result, 2 additional replicate samples for phytoplankton, zooplankton, and benthic invertebrates were collected at the site for statistical analyses.

Phytoplankton

Integrated 0-2 m surface water samples for phytoplankton analyses were collected from sites 1-3 using a tube-sampler from November 17, 2009 to December 16, 2010. On all dates, Secchi depth (Wetzel & Likens, 1991), chlorophyll, and light/dark bottle measurements (Sala et al., 2000) were also taken. Chlorophyll measurements were taken in the field using an *Aqua*Fluor™ Handheld Fluorometer. Phytoplankton samples returned to the lab were promptly preserved using 1.5-3.0 mL Lugol's solution (Clesceri et al., 1998) per 100 mL of sample and stored in the refrigerator until analysis. Phytoplankton identification and enumeration was conducted by Dr. Jeff Janik. Species richness and diversity were calculated for each site. Simpson's 1-D index of diversity (Washington, 1984) was calculated using the formula

$$\text{Simpson's 1-D} = 1 - \Sigma(n/N)^2$$

where n is the abundance (cells/mL) of each species (or genus if no more specific identification was made) and N is the total phytoplankton abundance (cells/mL) for the portion of sample counted. Values range from zero to one with higher values indicating higher diversity. Adsorption of iodine from the Lugol's solution in the November 2009 samples onto the Nalgene storage bottles reduced its effectiveness for preservation, so phytoplankton speciation was not conducted on the preliminary samples. The

concentration of Lugol's solution was approximately tripled to 1.5-3 mL for all subsequent samples, preserving all phytoplankton in good condition and suitable for identification and enumeration. Therefore, results are shown for the 6 sampling dates from February to December 2010.

The productivity of the phytoplankton community was assessed through measurements of oxygen evolution and consumption using light-dark bottles (Wetzel & Likens, 1991). Oxygen is produced by all green plants through the process of photosynthesis; as a result, the rate of O_2 evolution in sealed transparent BOD bottles provides a direct measure of the net rate of primary productivity (i.e., rate of photosynthesis minus the rate of O_2 consumed through respiration). Changes in DO concentrations in sealed bottles in which no light is allowed in (i.e., dark bottles) reflects the rate of respiration. Light/dark bottle measurements of oxygen consumption and evolution in the water column were taken in the field at a separate location near site 3. Six glass 300 mL BOD bottles (3 translucent, 3 covered by wrapping black tape around them to block out light) were filled with water from the top 2m of the water column and capped. The initial DO concentration and temperature of the water used to fill the bottles was determined using an Oakton DO 6 Acorn Series or YSI ProOBOD DO/temperature probe; the bottles were then capped, immersed to a depth equivalent to the measured Secchi depth by suspending over the buoy cable surrounding docking station 4, and left to incubate in the lake for a known period of time (generally 2-3 h). The BOD bottles were then retrieved from the lake and the DO concentration and temperature remeasured. The data were then used to calculate rates of net oxygen production or consumption in the light and dark bottles, respectfully.

Zooplankton

Triplicate vertical tows for macrozooplankton identification and enumeration were collected from the entire water column (except for the lowest 0.5 m directly above the sediments) and composited at each site using a 12 cm diameter 63 μm Wisconsin plankton net (Clesceri et al., 1998). Zooplankton were quantitatively transferred to labeled wide-mouth, 125 mL polypropylene bottles with 70% ethanol and stored on ice for return to the lab where they were kept in a 4°C refrigerator until analysis. Zooplankton samples were inspected under a Nikon E600 compound microscope at 40-100 x total magnification and individuals were counted and identified from 1 ml subsamples on a 1mm gridded Sedgewick-Rotifer cell (Clesceri et al., 1998). Results from these counts were used to estimate species richness and diversity (Simpson's 1-D). Approximately 150 L of water was filtered through the 63 μm Wisconsin net at each site in the November 2009 sample and samples later in the summer, while greater volumes (about 230 L) were filtered through the net during the triplicate tows at each site in February 2010 owing to the greater depth in the lake that resulted from the rain and runoff in January 2010.

Benthic Invertebrates

Benthic invertebrate samples were collected using a 15 cm x 15 cm Ekman dredge, with sediment passed through a 500 μm stainless steel sieve to collect the organisms (Clesceri et al., 1998). The contents remaining on the top of the sieve were rinsed into a labeled jar using lake water, put on ice in the field, refrigerated in the lab, and were sorted in the lab as soon as possible with the aid of a Nikon SMZ800 dissecting scope when needed. The numbers of chironomid larvae, phantom midge

larvae, and benthic cladocerans of the genus *Ilyocryptus* were recorded and specimens were subsequently preserved in labeled polypropylene bottles with 70% ethanol. Unless there were obvious differences in structure or color, no attempt was made to identify and separate the organisms beyond the above categories. As with phytoplankton and zooplankton, count results were used to estimate benthic invertebrate richness and diversity (Simpson's 1-D), although these estimates provide only very coarse-scale information given the lack of species-specific identification.

Stable Isotopes

Samples of chironomids, phantom midges, copepods, threadfin shad, and carp collected from the lake on December 16, 2010 were sorted in lab, then rinsed and lyophilized before analysis. Approximately 0.5-1.0 mg of each freeze-dried sample was weighed into a Sn tin using small pieces of multiple organisms were used where possible in order to achieve a more isotopically homogenous sample. A Costech ECS 4010 elemental analyzer (EA) coupled to a Thermo Delta-V Advantage isotope ratio mass spectrometer (IRMS) to analyze carbon and nitrogen isotope ratios ($\delta^{13}C$ and $\delta^{15}N$). Simultaneous measurements of both C and N isotopes were possible due to the use of a continuous flow IRMS. Delta notation (δ) is used to report stable isotope ratios, which are expressed in permil (‰).

RESULTS

Phytoplankton

Abundance of phytoplankton in the lake can be somewhat inferred from Secchi depth measurements. Transparency in the lake increased from about 0.4 m in the initial measurements made in November 2009 to approximately 0.5 m in February 2010, and then subsequently decreased through the spring and summer, reaching a minimum Secchi depth of 0.3 m in August (Fig. 2.2). Secchi depth then increased in October, exceeding 0.7 m for all three sites, before decreasing again in December to values similar to those observed at the start of the study in November 2009 (Fig. 2.2). The measured Secchi depths during the study period tended to be higher than those measured from June 2002 to June 2004, but lower than the Secchi depths measured in 2000 and 2001 (Veiga-Nascimento, 2004).

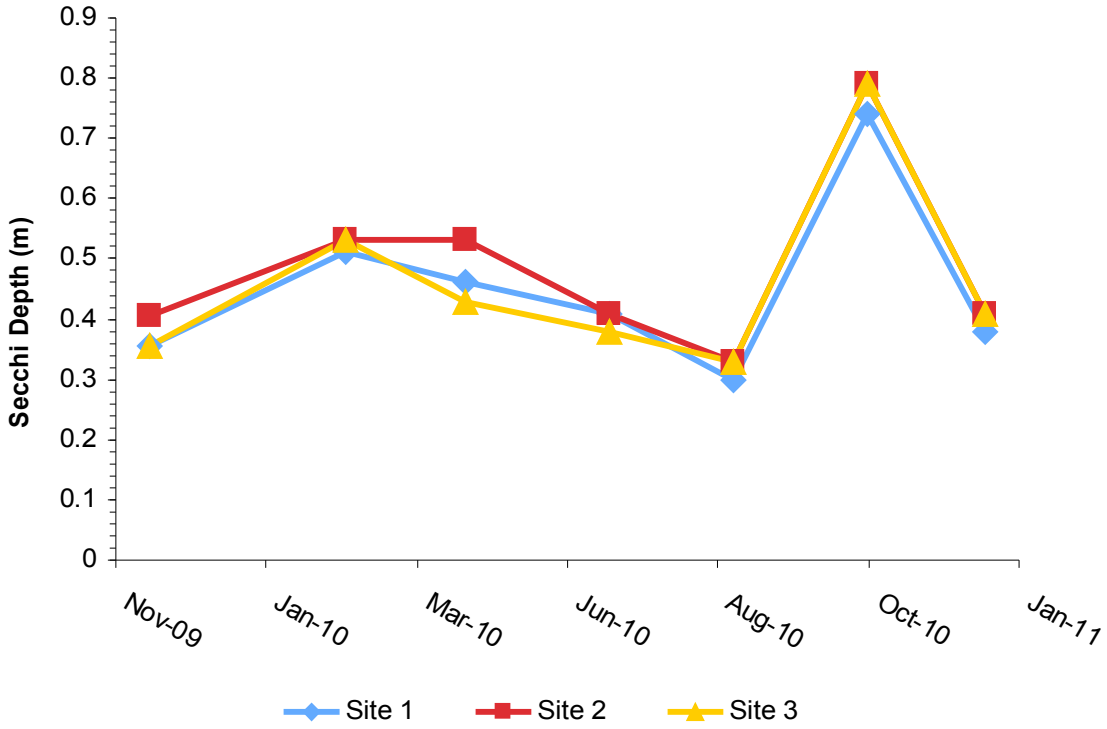

Fig. 2.2. Secchi depth at the 3 sampling sites during 2009 and 2010.

Chlorophyll concentrations measured using in situ fluorescence increased from February to April at all three sites, with the deepest site, site 2, having the lowest increase. Field measurements for chlorophyll ranged from 113.7-137.0 μg L^{-1} in February and from 122.9-157.9 μg L^{-1} in April. The lowest measurements were at site 2 for both sampling dates (Fig. 2.3). October chlorophyll concentrations ranged from 37.3 μg L^{-1} at site 3 to 73.6 μg L^{-1} at site 4. December chlorophyll values were considerably higher, ranging from 284.5 μg L^{-1} at site 4 to 376.8 μg L^{-1} at site 1 (Fig. 2.3). These values are comparable to those noted by Nascimento (2004), but the values observed in December are higher than any peak observed since 2004 (Anderson, 2010). The June and August chlorophyll measurements may have been complicated by relatively high turbidity and/or readings higher than the linear range for the secondary standard used,

resulting in “sample quenching” and readings unexpectedly low for the high summer phytoplankton biomass (Turner Designs, 2004) (Fig. 2.3). A July 2011 draft for the Lake Elsinore and Canyon Lake Nutrient TMDL Annual Water Quality Monitoring Report shows considerably higher chlorophyll a concentrations of about 100-275 µg L^{-1} for the summer period (Santa Ana Regional Water Quality Control Board, 2011).

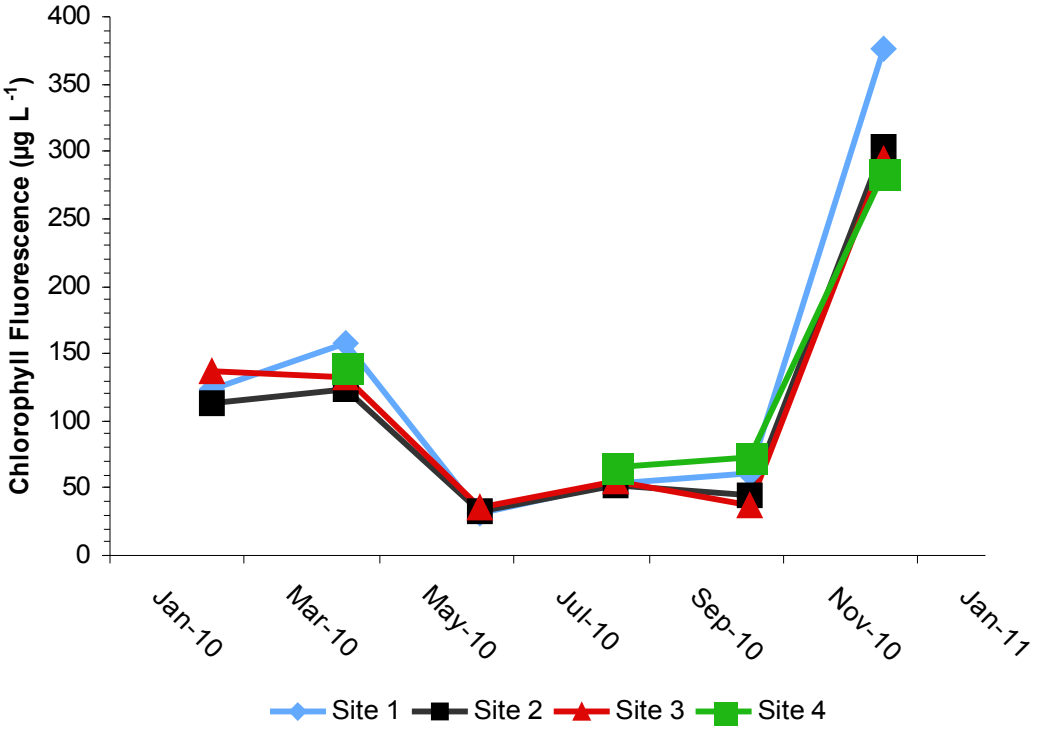

Fig. 2.3. Chlorophyll fluorescence at the 3 sampling sites during 2009 and 2010.

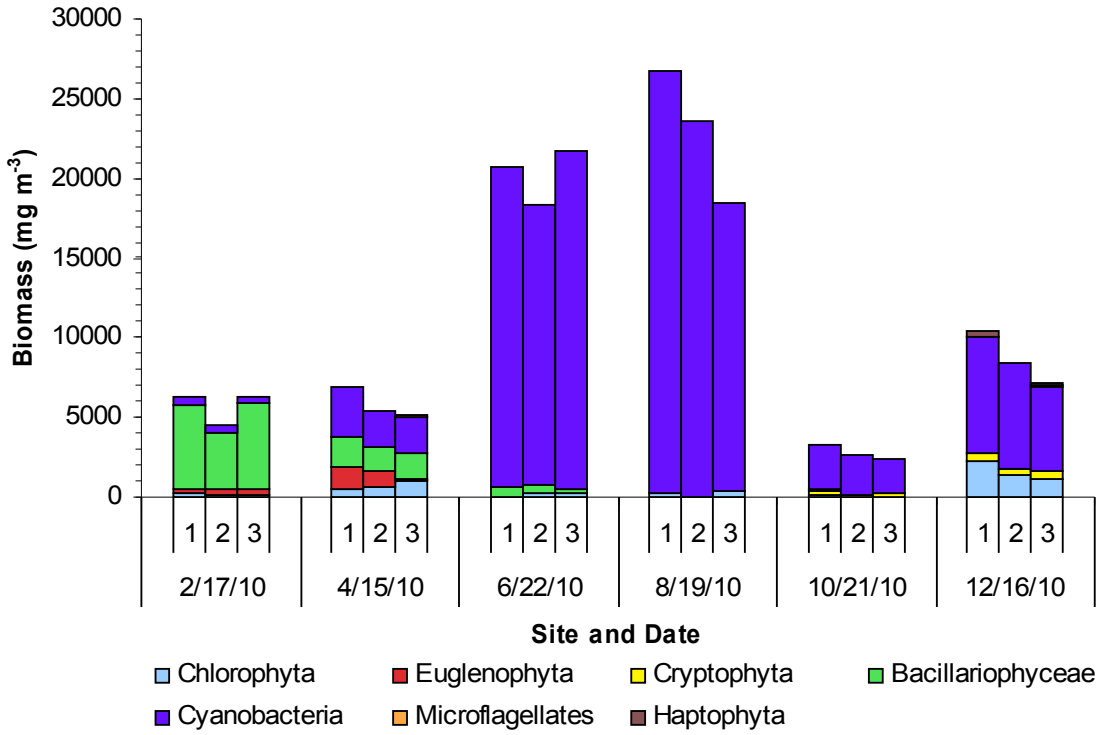

Fig. 2.4. Phytoplankton biomass by major algal groups at the 3 sampling sites during 2010.

Total phytoplankton biomass varied greatly between sampling dates, but was similar at the three TMDL sites for a single date (Fig. 2.4). Total biomass for the phytoplankton community was approximately 5,000 mg m^{-3} in February, with diatoms (Bacillariophyceae) dominating the phytoplankton community (75-86% of the total biomass) (Fig. 2.4). Smaller amounts of blue-green algae (Cyanobacteria), euglenoids (Euglenophyta) and green algae (Chlorophyta) were also present (Fig. 2.4). As a group, diatoms are adapted to cooler water than other phytoplankton and are thus often the dominant type during cool winter months and in cold, high elevation lakes. Phytoplankton biomass remained similar in April, although the community was already shifting toward larger amounts of blue-green algae, comprising 41-46% of the total

biomass, with green algae and euglenoids also slightly more abundant (Fig. 2.4). Phytoplankton biomass increased markedly in June, to about 20,000 mg m^{-3}, and was comprised predominantly of blue-green algae. The sampling in August yielded similar total biomass, although somewhat greater differences were found across the 3 sites. Total biomass declined dramatically in October, however, to only 2,000-3,000 mg m^{-3} (Fig. 2.4). Biomass subsequently increased in December due to return of green algae to the algal community, occurrence of cryptophytes and smaller numbers of haptophytes, and increased blue-green algal biomass (Fig. 2.4).

Species composition also changed notably between sampling dates. In February and April, only 4 genera of diatoms were found, but these genera changed from *Nitzschia*, *Skeletonema*, *Cyclotella* and *Stephanocyclus* (one species each) in February to *Chaetoceros* (1-2 species), *Cyclotella* (at least 1 species), *Nitzschia* (1 species), and *Synedra* (1 species) in April. *Nitzschia* and *Cyclotella* were found in June samples, *Chaetoceros* in October samples, and *Nitzschia* in December samples. No diatoms were found in August samples. The most numerically abundant cyanobacteria in February were *Pseudoanabaena limnetica* (formerly *Oscillatoria limnetica*) and *Anabaenopsis circinalis*, while, in April, *Aphanocapsa holsatica* and *Chroococcus disperses* were clearly numerically dominant at site 1, *Aphanocapsa incerta* at site 2, and *Aphanocapsa delicatissima* and *Chroococcus disperses* at site 3. The diatoms *Skeletonema subsalsum* and *Stephanocyclus meneghiniana* each comprised more than 10% of the biomass at sites 1 and 2 in February with *S. subsalsum* comprising more than 50% of the total biomass at all three sites at this time. In April, *Phacus sp.*, *Cyclotella sp.* 6-10 µm, and *Pseudanabaena catenata* comprised more than 15% of the total biomass at sites 1 and 2, *Cyclotella sp.* 6-10 µm and *P. catenata* comprised more

than 20% of the total biomass at site 3, and *Pseudanabaena limnetica* comprised 10% of the biomass at sites 2 and 3. The marine diatom, *Chaetoceros*, which may indicate brackish conditions, comprised 0.4% of the biomass at site 2 in February and 1-4% of the biomass at all 3 sites in April (with a minimum of 750 cells mL^{-1} at site 3). This is higher than in April 2003 when there was less than 400 cells mL^{-1} of phytoplankton in this genus (Oza, 2003).

In contrast to the earlier phytoplankton communities, the biomass during the summer phytoplankton bloom in 2010 was largely made up of a single species and total abundance was almost entirely comprised of just two species of cyanobacteria. June abundance was dominated by the cyanobacteria *Cylindrospermopsis c.f. catemaco* and *P. limnetica*. *P. limnetica* alone accounted for about 90% of the biomass at all three sites at this time. The August phytoplankton community was similar with *C. catemaco* and *P. limnetica* remaining the two most abundant species at all three sites and *P. limnetica* comprising approximately 75% to 83% of the biomass.

The October community changed modestly compared to June and August in that *Pseudanabaena c.f. acicularis*, rather than *P. limnetica*, was the most abundant species (cells mL^{-1}) at all three sites. The next two most abundant species were *C. catemaco* and *P. limnetica*. At this time, *C. catemaco* was most abundant at site 1 and *P. limnetica* was more abundant at the shallower sites, 1 and 3, than the deepest site, site 2. However, biomass in October was still dominated more by *P. limnetica* (33-42% of total biomass) than by any other species and the species with the second highest proportion of the biomass was *P. acicularis* (22%-23% of total biomass). Notably, Cryptophyta biomass was the highest yet in October, particularly at sites 1 and 3, with values ranging from 194-292 mg m^{-3} for samples from these 2 sites, corresponding to 8-

9% of the total biomass. Also, the marine diatom *Chaetoceros* was found at sites 1 and 3 in October with an abundance of 250 cells mL^{-1} at site 3 and 2550 cells mL^{-1} at site 1. This would be consistent with slightly higher salinities in October than in April, although other factors would also influence the abundance of this species. In December, the chlorophyte *Chlamydomonas*, and the cyanobacteria *Cylindrospermopsis raciborskii*, *P. limnetica*, and *P. c.f. acicularis* each comprised more than 10% of the phytoplankton biomass at all three sampling sites. *P. limnetica* was still the most dominant species by biomass, comprising 41% (site 1) to 48% (site 3) of the total phytoplankton biomass. During this month, both cryptophytes and haptophytes reached their highest biomass during the study with Cryptophyta biomass ranging from 286 mg m^{-3} at site 2 to 572 mg m^{-3} at site 3 (3-8% of total biomass) and Haptophyta biomass ranging from 191 mg m^{-3} at site 3 to 536 mg m^{-3} at site 2 (3-6% of total biomass).

Some of the phytoplankton species and genera present are potentially harmful to humans and/or other organisms. The cyanobacterium *Cylindrospermopsis raciborskii* was found in February, June, August, October, and December samples. *C. raciborskii* has the toxin cylindrospermopsin and saxitoxins which make it hepatotoxic, neurotoxic, and carcinogenic. It has been reported to kill cows (Landsberg, 2002 and references therein). In February, this species was present in very small quantities with an average biomass of just 4 mg m^{-3}, about 0.05% of the total phytoplankton biomass. However, total biomass for this species was several hundred to 1,000 mg m^{-3} in June and October (more than 15% of the biomass in October) and August biomass values were above 3,000 mg m^{-3} for all but one site 1 sample in which this species was still found at a high biomass of about 2,670 mg m^{-3}. Even during December when the water was cooler, *C.*

raciborskii had an average biomass of 1172 mg m^{-3} at site 1, 1302 mg m^{-3} at site 2, and 997 mg m^{-3} at site 3.

Pseudanabaena catenata, another cyanobacterium, is reported to be neurotoxic (Gorham et al., 1982), but no deaths resulting from it have been documented (Landsberg, 2002). It comprised about a fourth of the phytoplankton community biomass at all three sites in April, with biomass ranging from 1162 mg m^{-3} at site 3 to 1753 mg m^{-3} at site 1. A third potentially toxic cyanobacterium was also present in smaller quantities. *Planktothrix agardhii* was present at site 3 in February (biomass of 14.3 mg m^{-3}) and at sites 1 and 3 in April and June (biomass 81 mg m^{-3} and 40 mg m^{-3} respectively in April and 356 mg m^{-3} and 214 mg m^{-3} respectfully in June). This species only comprised a minimum of 0.2% of the phytoplankton biomass at site 3 in February to a maximum of 1.8% of the biomass at site 1 in June. It is hepatotoxic and produces microcystins that are toxic to zooplankton. *P. agardhii* has been reported to kill cows, roach and other fish, waterfowl, and muskrats (Landsberg, 2002 and references therein).

Another genus present that is potentially harmful to zooplankton is a diatom of the genus *Synedra*, which Ban et al. (1997) report to be harmful to copepods. It was found at sites 1 and 2 in April with its greatest biomass of 131 mg m^{-3} at site 1 (1.9% of the total phytoplankton biomass) and a biomass of 44 mg m^{-3} (0.8% of the total phytoplankton biomass) at site 2.

These above mentioned taxa may pose a threat to human or animal health or life if the strains present are toxic and produce large enough quantities of toxins. Furthermore, some strains of the cyanobacterium *P. limnetica* have been reported to cause unpleasant tastes and odors in drinking water (Britton et al., 2005). This species has comprised a greater portion of the biomass than any other species from June and

August (≥ 75% of the biomass at all three sites) through October and December (about 30-40% and 40-50% of the overall biomass at all three sites respectfully). While all of these species may be undesirable and/or potentially hazardous, it should be noted that the production of certain types of toxins and the quantity of toxins produced is quite variable between different strains of a single species or group of similar species and there is not enough information from this study to determine whether or not any of the phytoplankton present in Lake Elsinore actually pose a significant threat to people or wildlife using or visiting the lake.

While fewer species of phytoplankton were specified in the 2003 study of phytoplankton in Lake Elsinore, comparisons of genera (and a couple of species) present and of the dynamics of the major phytoplankton groups can be made. Aside from the previously mentioned presence of the diatom *Chaetoceros* in both 2003 and 2010 studies, other species or genera that were found in both the past and present study include the cyanobacteria *P. aghardii* and *P. limnetica* (both formerly in the genus *Oscillatoria*), the diatoms *Nitzschia* and *Cyclotella*, and green algae of the genus *Scenedesmus* (Oza, 2003). Genera that were noted in the 2003 study, but not the current study include *Aphanizomenon* (cyanobacteria), *Stephanodiscus* (diatom) and *Ankistrodesmus* (green algae) (Oza, 2003). As in 2003, this study found diatoms to be significant components of the phytoplankton community (if not the major components) during February through mid spring, then there was a shift in the phytoplankton assemblage, where cyanobacteria became far more abundant than diatoms, that was evident by June (Oza, 2003). Also similar to the previous study, green algae were never very abundant, but they peaked to noticeable numbers in spring (Oza, 2003). Green

algae had their greatest biomass in December 2010, but no samples were previously taken during this month for comparison.

There were some notable differences between the phytoplankton communities in the 2010 and 2003 studies. *Chaetoceros* was the dominant diatom during most of the study period in 2003, reflecting the high salinities present at that time (Oza, 2003), while in 2010, *Nitzschia*, *Cyclotella*, and *Skeletonema* were at least as important in the diatom community biomass at one time or another and *Synedra* had comparable (though lower) biomass in April. Furthermore, many more genera were noted during this study than the previous study and more major groups were noted. The previous study focused on cyanobacteria/blue-green algae (2 genera), diatoms (4 genera), and green algae (2 genera) for a total of three major groups and 8 genera (Oza, 2003). Throughout 2010, a total of 7 major phytoplankton groups and 38 different genera (18 green algae, 1 Euglenophyta, 2 Cryptophyta, 6 diatoms, 9 cyanobacteria, 1 Microflagellates, 1 Haptophyta) were collected. This is likely due in part to the higher salinity and lower diversity in the lake during the 2003 study, although no doubt also reflects the differences in experience and expertise between the professional hired for this study and the graduate student who examined the 2003 samples.

The detailed algal speciation described above can be used to develop simple metrics for the phytoplankton community and its evolution over time. Species richness, simply the number of species identified in any given sample, exceeded 35 at sites 1 and 3 in February 2010, but declined to 30 species in April, and then decreased more dramatically by June (Fig. 2.5). Species richness remained low (10-12 species) in August and October samples, before increasing slightly in December (Fig. 2.5). These results may all be quite a bit lower than what may be expected for a productive lake of

the size of Lake Elsinore (roughly 10^3 ha) based on a study by Dodson et al. (2000) of 33 well-studied lakes and the relationships between primary productivity, lake size, and species richness.

Fig. 2.5. Phytoplankton species richness at the 3 sampling sites during 2010.

Similar to species richness, diversity of the phytoplankton community also varied markedly over time. Values for the Simpson's 1-D diversity index ranged from 0.69-0.86 in February (Fig. 2.6), reflecting both the high species richness and abundances well-distributed across the different species. Simpson's 1-D values edged down slightly in April (to 0.69-0.75), but decreased much more dramatically in June (Fig. 2.6), to values of about 0.2, reflecting the low overall diversity of a phytoplankton community dominated by a relatively small number of blue-green algal species. Values increased somewhat,

but remained low in August, before increasing to about 0.6 in October and returning to about 0.8 by December (Fig. 2.6). Although December biomass was still comprised mostly of cyanobacteria (Fig. 2.4), higher amounts of green algae, as well as Cryptophytes and Haptophytes and a few diatoms and microflagellates in the lake contributed to the increased diversity.

Overall, green algae (which contained the greatest number of species found) may be the most important group for a high diversity phytoplankton community, followed by the diatoms. It was only when green algae made up at least 2% of the total biomass that species diversity got much above 0.6 and diversity only reached 0.8 when green algae comprised at least 15% of the biomass, or diatoms at least 75% of the biomass. Therefore, the relative biomass of the major phytoplankton groups is important for diversity in the phytoplankton community of Lake Elsinore. The decrease in species diversity and richness in summer was due largely to a bloom of the cyanobacterium *Psuedanabaena limnetica* (which comprised almost all of the June and August biomass). A similar decrease in Simpson's diversity (from 0.75 in February to <0.2 in mid-June) was noted in 2003 as the phytoplankton community shifted from containing a large proportion of diatoms to containing primarily cyanobacteria (Oza, 2003).

Phytoplankton diversity is important because it indicates the variety of food from which algivorous species in the lake (such as rotifers and cladocerans) can select. High species diversity indicates that organisms that feed on algae have a wide selection of food on which to feed and are, therefore, more likely to find a type of algae that is favorable for their growth. Low phytoplankton species diversity, on the other hand, suggests that the organisms that feed on algae either have to survive with what is present to eat or perish. Thus, different types of zooplankton may be expected to exist

with a low diversity phytoplankton population than one with high phytoplankton diversity. This may, in turn, influence the zooplanktivores, including threadfin shad. Thus, changes in the diversity of the phytoplankton community are likely to have noticeable impacts on other communities in the lake and the functioning of the ecosystem as a whole.

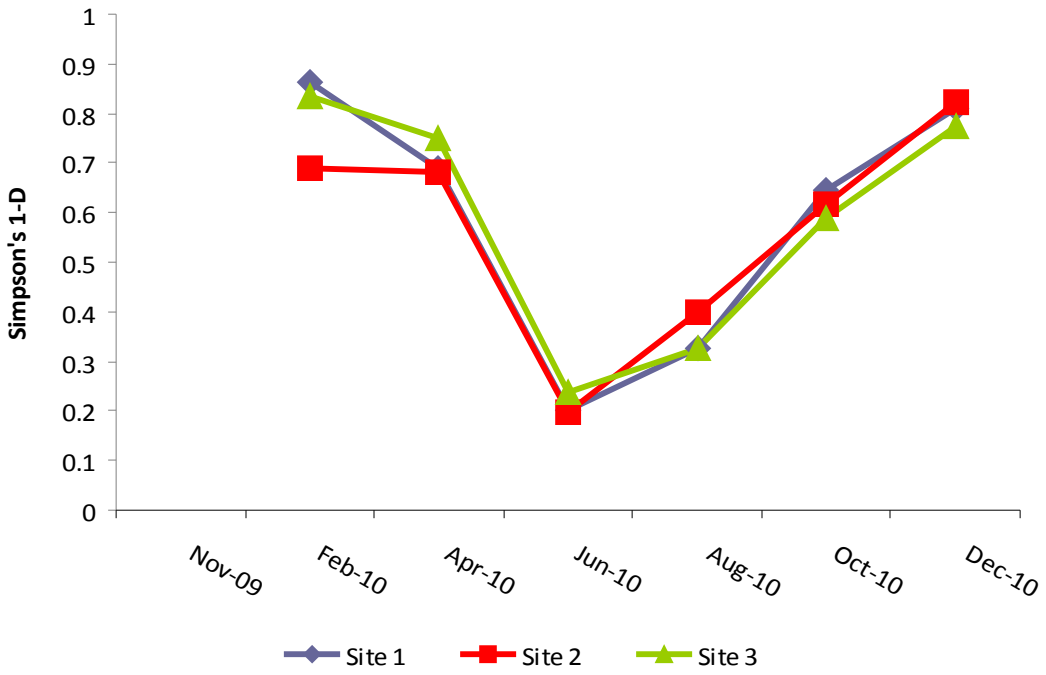

Fig. 2.6. Phytoplankton diversity at the 3 sampling sites during 2010.

Oxygen was produced at a rate of 0 to 2 mg L^{-1} h^{-1} during mid-day measurements at the lake (Fig. 2.7, light bottles). With the exception of June, the mean rates of oxygen evolution from the light/dark bottle experiments tended to follow the same trends as phytoplankton biomass during 2010, with a peak in August 2010 at 2.1 mg O_2 L^{-1} hr^{-1}. August also had the greatest phytoplankton biomass (Fig. 2.4) and lowest Secchi depth (Fig. 2.2). There were considerably lower rates of oxygen evolution (around 0.5-1 mg O_2

L^{-1} hr^{-1}) at other times during this study (Fig. 2.7). An additional date of field sampling (4/5/11) revealed that April 2011 oxygen evolution rates were slightly above 0.5 mg O_2 L^{-1} hr^{-1} (not shown in figure). This value is lower than those for December 2010, but still within the same range mentioned above.

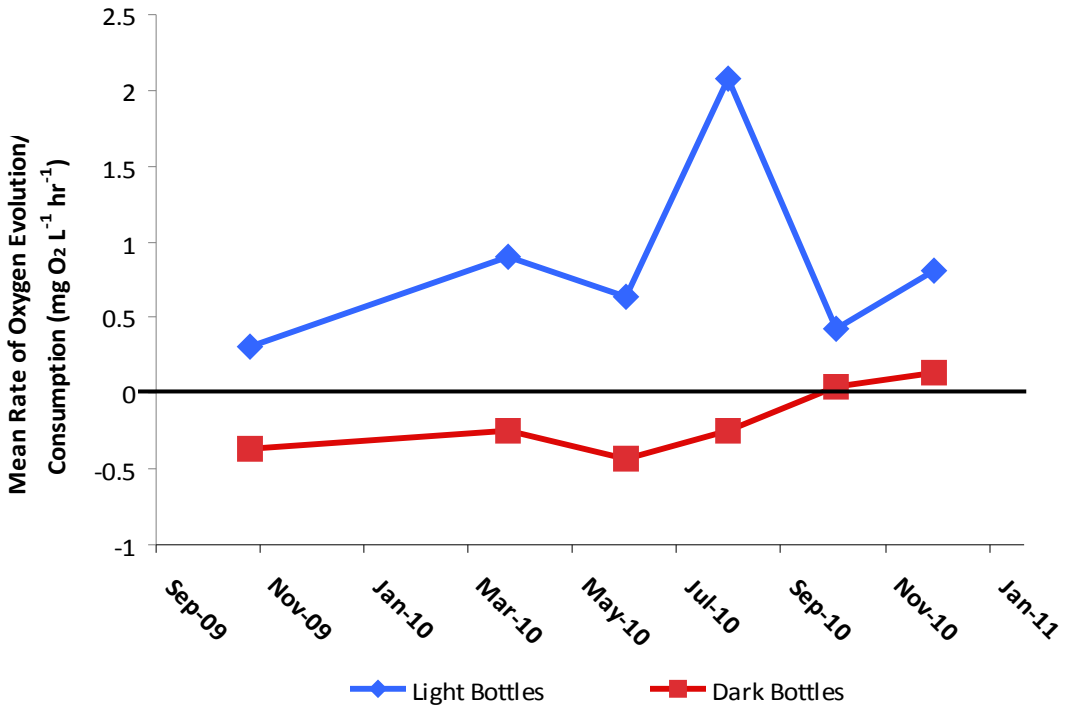

Fig. 2.7. Light/Dark bottle results during 2009 and 2010.

A plot of total phytoplankton biomass at site 3 (the site nearest the spot where the light/dark measurements were taken) versus oxygen evolution rates is best fitted to a polynomial function with R^2=0.58 (not shown). The reason for the non-linearity of this plot is likely because other factors such as water temperature, irradiance, and the composition of the phytoplankton community also influenced the net rate of oxygen evolution in the lake. Higher temperatures may allow faster rates of photosynthesis and

oxygen evolution, but rates of respiration also vary with temperature. Greater irradiances mean a greater supply of photons for the splitting of water to generate oxygen in photosynthesis. The highest biomasses corresponded to phytoplankton communities dominated by filamentous cyanobacteria (mainly just a single species), and this majority of the peak biomass may not have been as efficient as primary producers as other species present in lower biomass or at different times of year.

Mean oxygen consumption rates were generally of a lower magnitude than rates of oxygen evolution for all the dates and were usually between 0 to –0.5 mg O_2 L^{-1} hr^{-1} (Fig. 2.7). In October and December, however, the oxygen consumption rates were slightly positive (oxygen concentration increased) (Fig. 2.7). The reasons for this are not known for sure, though it is probably due to error resulting from the method itself and/or the instrument used for the measurements. For example, in addition to being influenced by the phytoplankton community composition and biomass, rates of oxygen production and loss in the bottles may also be influenced by incubation time. Oxygen production rates for this study tended to decrease (become less positive or more negative) for both the light and dark bottles as incubation time increased. However, at the longest incubation times, the oxygen production rates suddenly switched from negative to positive implying that oxygen concentrations increased in the dark bottles only during the longest incubation times (graphs not shown). There are many other biotic and abiotic factors that could influence dissolved oxygen levels in bottles and likely contribute to some of the results shown for this study (Patten et al., 1964 and references therein; Pratt & Berkson, 1959). Bacterial respiration (in addition to the respiration by the primary producers that the method principally measures) is, perhaps, among the most important of these factors.

Zooplankton

The zooplankton community in Lake Elsinore varied seasonally, and to a lesser extent, by site, both in terms of abundance and composition. The arithmetic mean of the abundances measured at each site was low in the fall of 2009, averaging 71.6±38.2 organisms L^{-1} (Fig. 2.8). Average lake-wide abundance then reached a peak in April 2010 at 142.7±44.4 organisms L^{-1} and decreased in June when it was at its lowest value (56.7±11.4 organisms L^{-1}) (Fig. 2.8). As summer progressed, the zooplankton community abundance increased somewhat by August, and more dramatically in October, when zooplankton abundance was at its highest during the sampling period at sites 1 through 3 and was also relatively high at site 4 (though lower than the April value for this site) (Fig. 2.8). The average abundance for all four sites in October was 376.2±133.8 organisms L^{-1} before a marked drop in December to abundances similar to those for November 2009 and June 2010 (Fig. 2.8). In general, zooplankton abundance tended to be lower at the deepest site (site 2) and higher at sites 1 and 3, while the shallowest site (site 4) tended to have either the highest zooplankton abundance (spring) or the lowest (fall) (Fig. 2.8). However, as the result of many different influencing variables, there did not appear to be any consistent differences between sites that held for all sampling dates.

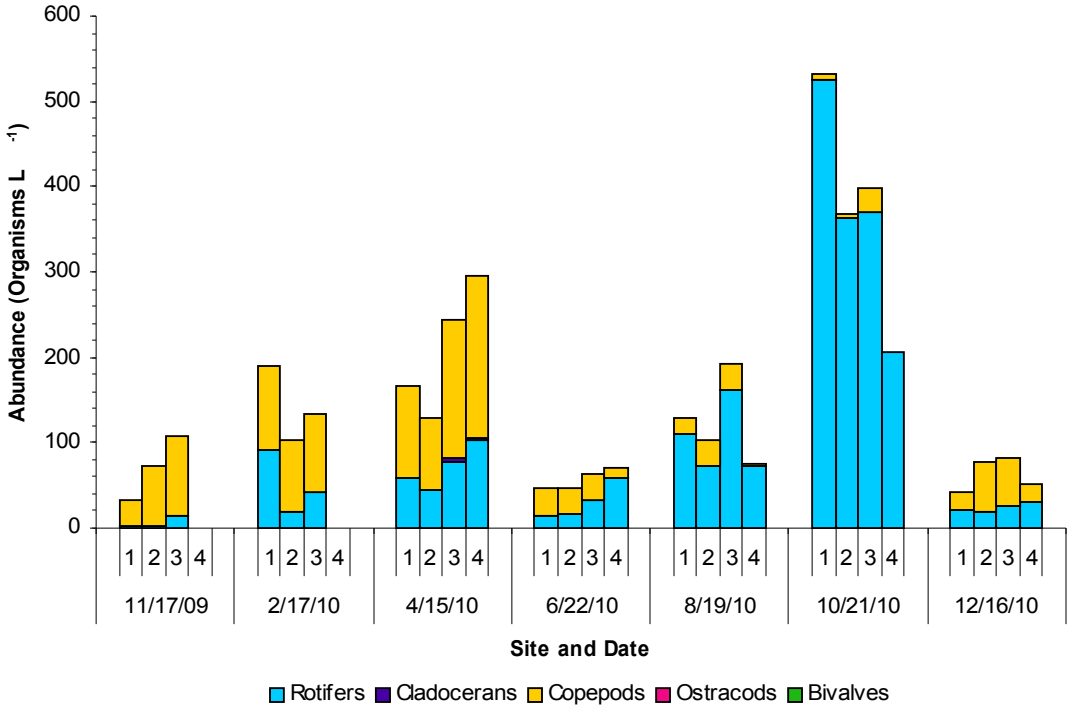

Fig. 2.8. Zooplankton abundance by major groups at the 4 sampling sites from November 2009 until December 2010.

Estimated zooplankton species richness was greatest in February 2010 (8-9) with a second, slightly lower peak in October 2010 (6.3-8) and the lowest values in June 2010 (3-4) (Fig. 2.9). Zooplankton diversity increased from very low numbers in November 2009 and peaked in February-April 2010 for the shallower sites (1 and 4) and April-June for the deeper sites (2 and 3), but then decreased at most sites in August and October (Fig. 2.10). Diversity values in December 2010 were very close to their October values at sites 2, 3, and 4, but increased considerably from an average of 0.24 to 0.60 at site 1 (Fig. 2.10).

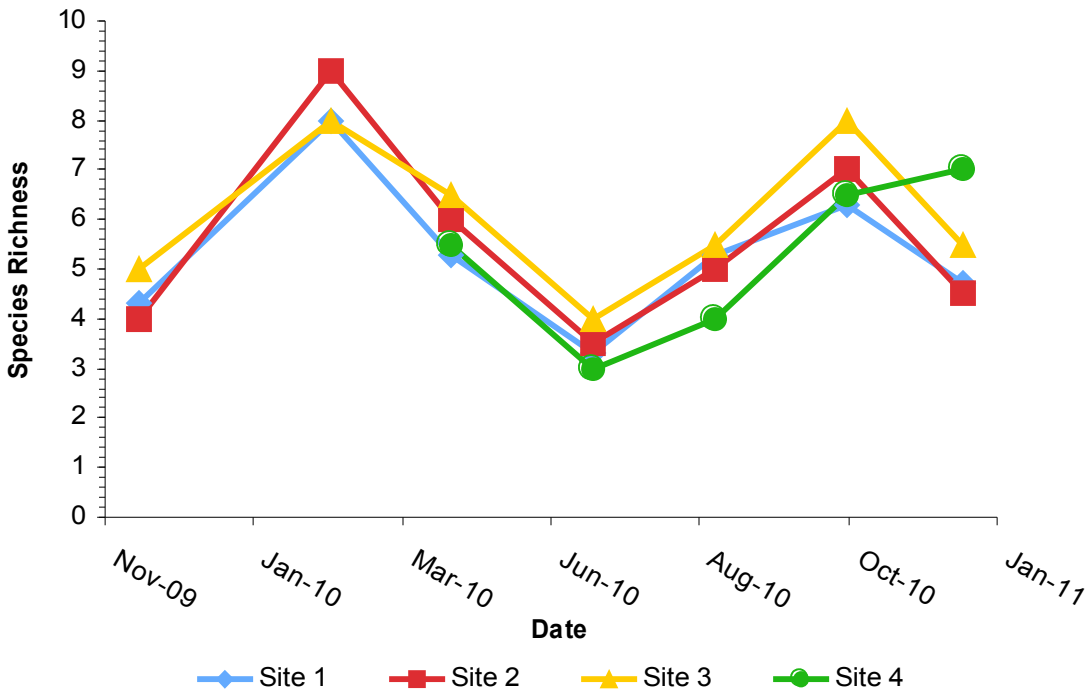

Fig. 2.9. Zooplankton species richness at the 4 sampling sites from November 2009 until December 2010.

Diversity values reflect the compositions of the zooplankton community, which shifted throughout the seasons and between sites. Generally, sampling dates with a more even distribution of the major zooplankton groups had higher zooplankton diversity (Figs. 2.8 & 2.10). In November, the zooplankton community abundance in the lake was comprised almost entirely of copepods (92% on average) with only 8% rotifers. By February, rotifers had become more prominent, numerically comprising 44% of the zooplankton community abundance on average with copepods at 56% and cladocerans at 0.14%. Copepod relative abundance increased again in April, as did that of cladocerans, so that the spring zooplankton community was, on average, 65% copepods, 34% rotifers, and 0.6% cladocerans. By summer, rotifers began to dominate

the zooplankton community, which contained an average of 52% rotifers and 48% copepods by abundance in June and mostly rotifers in August and October (83% rotifers, 17% copepods in August and 98% rotifers, 2% copepods in October, on average). The December 2010 zooplankton community was, on average, 62% copepods and 38% rotifers (Fig. 2.8). These trends may be somewhat different than those for 2003 and 2004 in Lake Elsinore when all types or zooplankton (rotifers, cladocerans, and copepods) apparently had peak populations during summer and around the beginning of 2004 (Veiga-Nascimento, 2004) rather than peaking at different times of the year. Bivalves and ostracods were always a very minor component of the zooplankton community, never comprising more than 0.1% of the total abundance. However, quagga veligers were found in all three samples for site 1 (and only site 1) on August 19, 2010 with an average abundance of 0.0976 organisms/L (or approximately one veliger for every 10.25 L). No veligers were found before or since this sampling date. Ostracods, when present, tended to be found at the two shallower sites (1 and 4) during cooler times of the year.

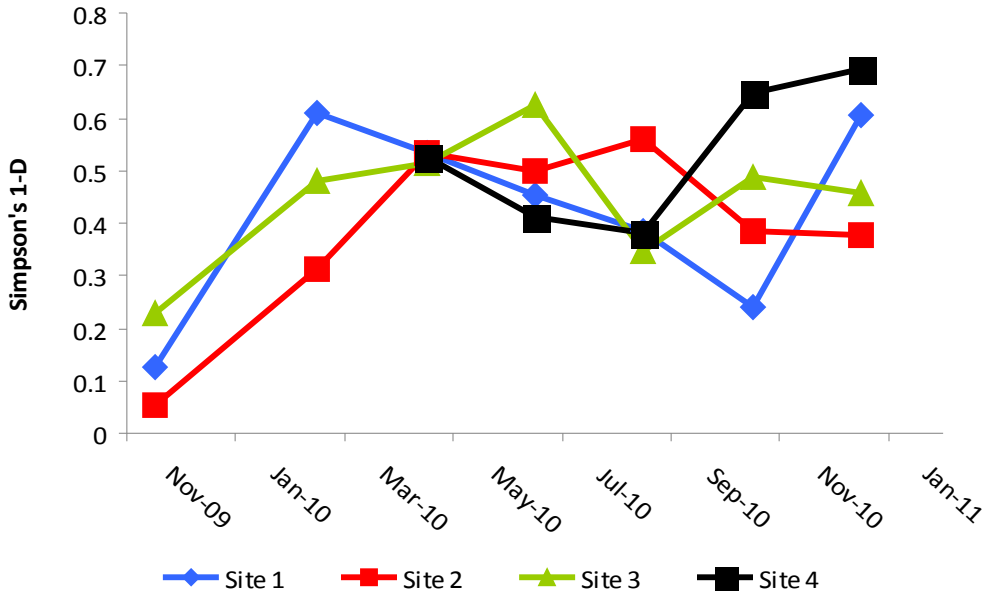

Fig. 2.10. Zooplankton diversity at the 4 sampling sites from November 2009 until December 2010.

Rotifers

Average rotifer abundance in the lake was at its minimum of only 6.1±6.9 organisms L^{-1} in November 2009, but increased for February and April 2010 (Figs. 2.8 & 2.11). Rotifer abundance then decreased until June and, over the summer, exploded to a lake-wide average of 104.0±42.9 organisms L^{-1} in August, then the maximum of 366.2±130.9 organisms/L in October. Rotifer abundances dropped considerably in December 2010 (Figs. 2.8 & 2.11).

Along with changes in total rotifer abundance, there were also changes in the genera of rotifers present and their relative abundances from one sampling date to the next. *Polyarthra* dominated the rotifer community from November through June, with *Synchaeta* being the second most abundant rotifer in February, *Keratella* in April, and

Brachionus in June (Fig. 2.11). There was a striking shift in the rotifer community from June to August. In August, *Polyarthra* was no longer the dominant rotifer. The genus *Hexarthra* took its place and became by far the dominant zooplankter found in August samples (Fig. 2.11). *Polyarthra* and *Brachionus* were found in low numbers at this time and very few *Keratella* were counted. Similarly, *Polyarthra* was not the dominant rotifer in October, at which time *Brachionus* comprised most of the large rotifer community and *Asplanchna* appeared in numbers more comparable to those of *Polyarthra* (Fig. 2.11). Also found in October samples were a few specimens of *Hexarthra* at sites 2 and 3, 2 specimens of *Keratella* at site 3, 2 specimens of a new genus, *Filinia*, at site 3, and a few individual rotifers of genera that have not been identified. The December rotifer community was dominated by *Filinia* and *Synchaeta* at all four sites, with *Filinia* generally being the more abundant genus. Aside from its highest October peak, the large, predatory rotifer *Asplanchna* first peaked with small numbers in February when the soft-bodied, easily edible *Synchaeta* had its first peak (Fig. 2.11).

Differences were also present between sampling sites. In general, *Keratella* was most prominent at site 3, less at site 2, and least at site 1, while *Brachionus* followed the opposite trend, being most prominent at site 1. Also, rotifer abundances at the two shallowest sites (1 and 4) tended to peak earlier than at the two deepest sites (2 and 3). However, despite some differences, general seasonal trends in species composition and abundance tended to occur across all sites on the lake (Fig. 2.11).

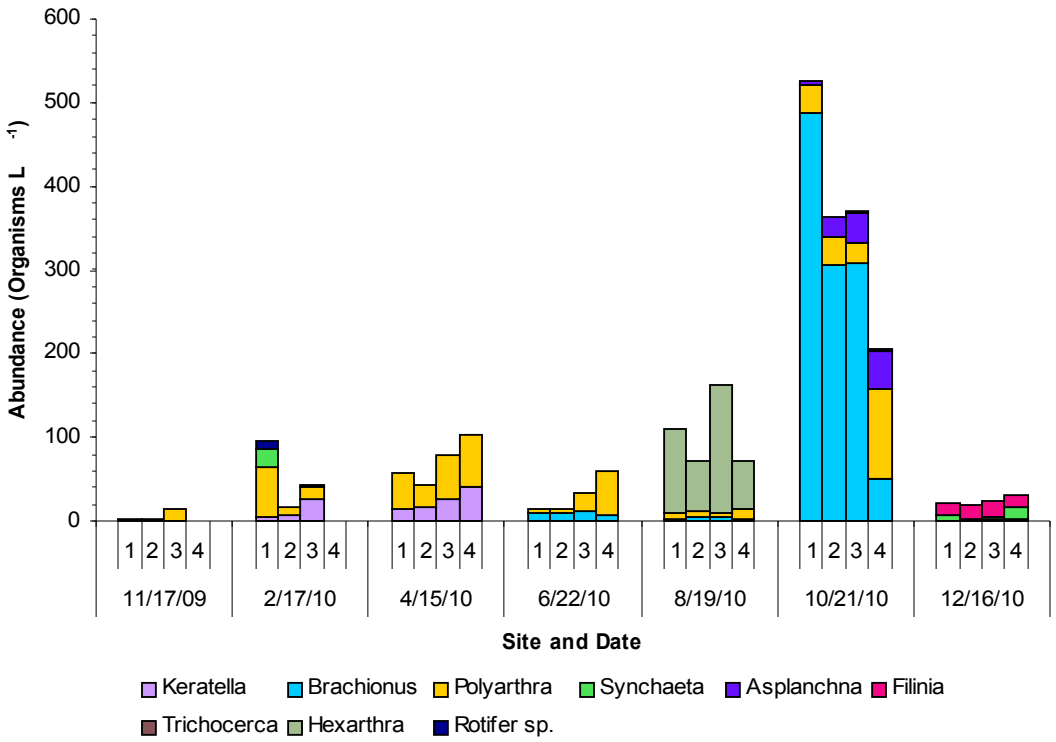

Fig. 2.11. Rotifer Abundances by genus at the 4 sampling sites from November 2009 until December 2010.

Aside from trends in rotifer genera, there were also noticeable seasonal differences in the species of the genus *Brachionus* that were found in the lake. Of the species that were identified (at least tentatively), *B. variabilis* was found only in February and April 2010 and then seemed to be replaced by a species (perhaps *B. plicatilis* or *B. rotundiformis* which inhabit marine or brackish waters) that was apparently present in good numbers in all seasons though its abundances were much lower in winter. The relatively huge abundance of rotifers in the lake in October was due primarily to a very large population of *B. calyciflorus*. *B. urceolaris* (or perhaps a form of *B. quadridentatus* depending on the reference used for identification) was found in fairly high abundances in October of 2010 with a few in December as well. In October, *B. havanaensis* was

also found, though with much lower abundances than the other two species. A different species, *B. angularis*, was found in December samples. As with the other two *Brachionus* species identified in December (*B. urceolaris* and *B. plicatilis/rotundiformis*), its abundance was quite low at this time. Other genera besides *Brachionus* may have also exhibited seasonal changes in species composition, but individual species were either not identified or not present during enough seasons for comparison. The only non-*Brachionus* species identified was *Asplanchna brightwellii*, which was the only species of this genus found in the lake.

In comparison with a previous 2003-2004 study (Veiga-Nascimento, 2004), some similarities and differences are apparent. The June peak in *Brachionus* numbers was also observed in 2003 and 2004 to various extents and the October peak was observed in 2003 (Veiga-Nascimento, 2004). (Upon reexamination of archived samples, it was found that *Brachionus* was mistakenly identified as *Asplanchna* in this earlier study.) *Polyarthra* and *Hexarthra* were not found in the past and *Filinia* was previously found in higher numbers (Veiga-Nascimento, 2004). Also, previous rotifer abundances were frequently reported in thousands of organisms per liter in the lake (Veiga-Nascimento, 2004), while, during this study, rotifers were only found in hundreds of organisms per liter at the most. More frequent sampling in the past may account for some of these differences as peaks and genera could have been missed in between present sampling dates.

Copepods

Copepod concentrations varied over time, increasing from an average across the sampling sites of 66±33 organisms L^{-1} in November 2009 to 136±49 organisms L^{-1} in late

April (Fig. 2.12). Their abundances dropped sharply by June 2010 and remained comparatively low through the summer and fall (Fig. 2.12). Nauplii comprised a relatively large fraction of the copepod community through April 2010, but their abundances declined by more than 95% at sites 1-3 and by 88% at site 4 in June 2010 and adult cyclopoids made up almost the entire copepod community. Nauplii once again became dominant from August to October, but adults were more abundant at sites 1, 2, and 3 in December 2010 (Fig. 2.12). Calanoid copepods appeared with small concentrations in April (average 1.23 ± 0.46 organisms L^{-1}) (Fig. 2.12). Calanoid copepods seem to have decreased since 2003 and 2004 when their concentrations were 44 organisms L^{-1} and 216 organisms L^{-1} respectively, at moderate depth sites in May (Veiga-Nascimento, 2004). Cyclopoid copepods were by far the most dominant type present in the lake. Due to the difficulty in identifying copepods to the genus and species level, it is not known which, or how many, copepod species are present in Lake Elsinore, but *Acanthocyclops robustus* may be among the cyclopoids and the calanoids resemble diaptomids.

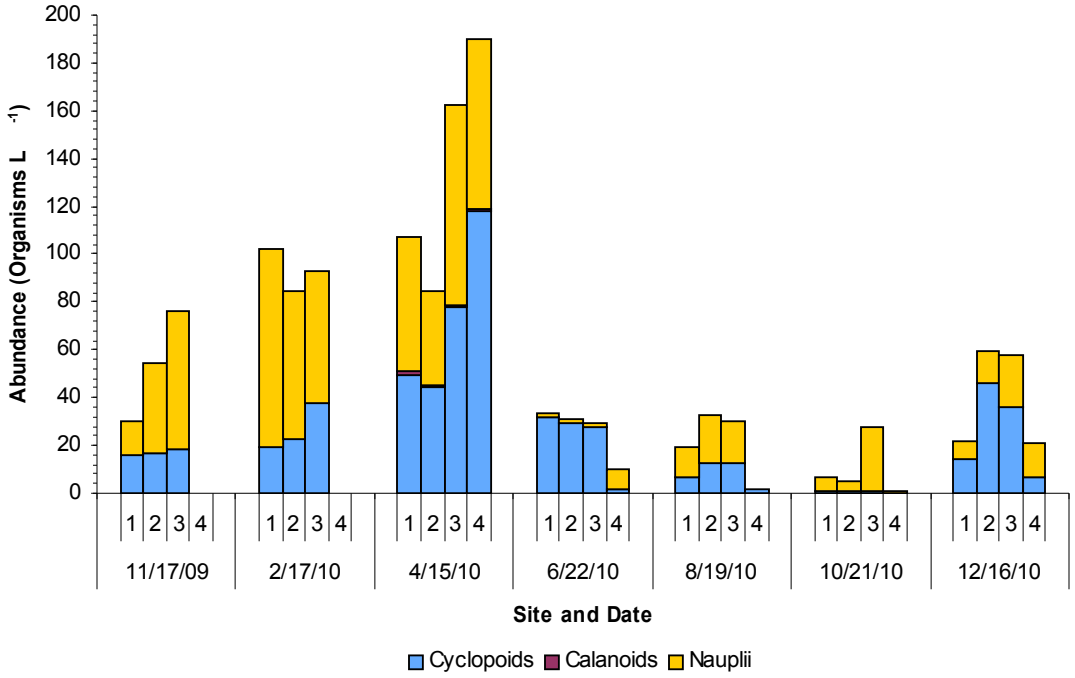

Fig. 2.12. Copepod Abundances by type at the 4 sampling sites: November 17, 2009 through December 16, 2010.

Cladocerans

The cladoceran community in the lake was very small to nonexistent (Fig. 2.13). During 2009-2010, cladocerans were only found in cooler months of the year. Cladocerans were found in February (sites 1 and 2), April (all four sites), and December (site 4) with total numbers in the lake peaking at an average of 1.3±0.9 organisms L^{-1} in April. The maximum concentration of cladocerans found was 2.58 organisms L^{-1} at site 3 in April (Fig. 2.13). *Daphnia* species identified include *D. lumholtzi* (which may have come in with runoff from Canyon Lake following recent rainfall) and *D. ambigua*. In contrast, *Daphnia exilis* was reported as the only *Daphnia* species present in the 2003-2004 samples. During the peak in cladoceran abundance in April 2010, *Bosmina* was

also found, as was a specimen of the genus *Leydigia*. The cladoceran found at site 4 in December 2010 was a chydorid of a different genus than *Leydigia*, probably *Alona* (Fig. 2.13). Cladoceran concentrations appear to have decreased considerably since 2003-2004 (Veiga-Nascimento, 2004).

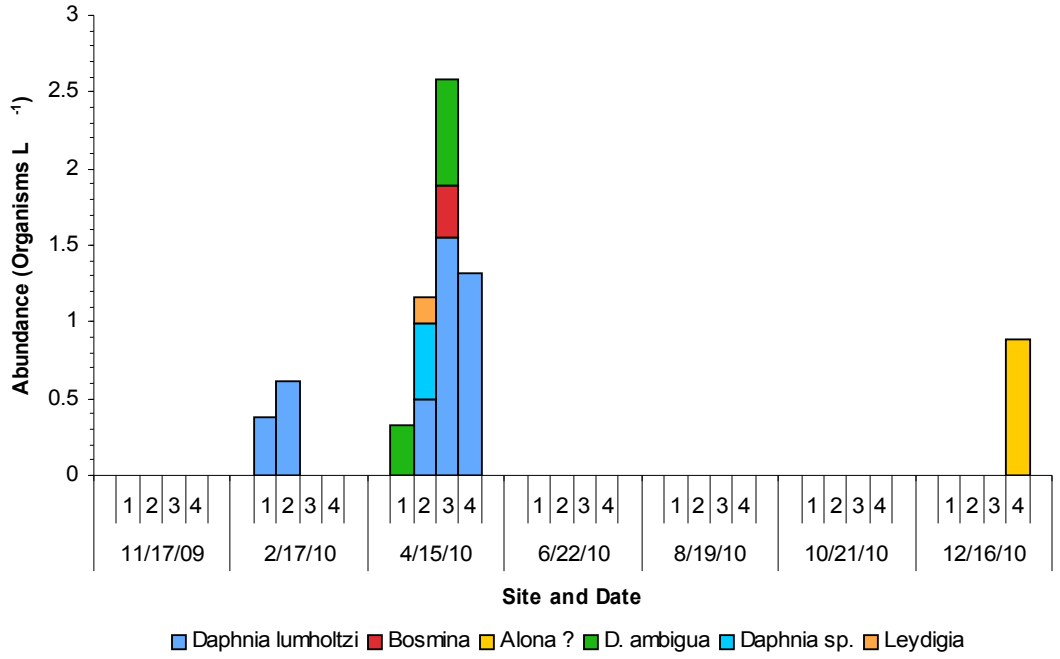

Fig. 2.13. Cladoceran abundances by species at the 4 sampling sites: November 17, 2009 through December 16, 2010.

Benthic Invertebrates

The density of benthic invertebrates was highly variable among the sites, sampling dates, and replicates for site 1. Density was always much lower at the two deeper sites (2 and 3) than at the two shallower sites (1 and 4) (Fig. 2.14). This may be due to lower oxygen levels at deeper sites, or simply because the deeper sites tend to be further from shore and are, therefore, less easily accessible to the terrestrial adult

females when they lay their eggs, but is likely a combination of these and other factors such as sediment composition. Benthic densities tended to be highest at site 4, the shallowest, site closest to shore (Fig. 2.14).

Total benthic densities, including chironomid midges (believed to be mostly, if not entirely, red members of the genus *Chironomus*), phantom midges of the genus *Chaoborus* (believed to be the species *C. punctipennis*), and the red benthic cladoceran *Ilyocryptus* (resembling *I. sordidus*), were highest in November 2009 at site 1 (2473 organisms m^{-2}). There were smaller peaks in benthic densities in April 2010 at sites 1 and 4 (1383 organisms m^{-2} at site 1 and 2066 organisms m^{-2} at site 4) and in December at site 1 (1133 organisms m^{-2}) (Fig. 2.14). Since sampling was started at site 4 in April, total benthic densities were never lower than their August value of 430 organisms m^{-2} until October when no benthic invertebrates were found at this site. Site 4 abundances increased again in December to 445 organisms m^{-2}. At sites 1, 2, and 3, other than the peaks mentioned, benthic densities did not exceed 500 organisms m^{-2} at site 1 and were always less than 200 organisms m^{-2} at sites 2 and 3 until December when site 3 had a total benthic density of 229 organisms m^{-2} (Fig. 2.14).

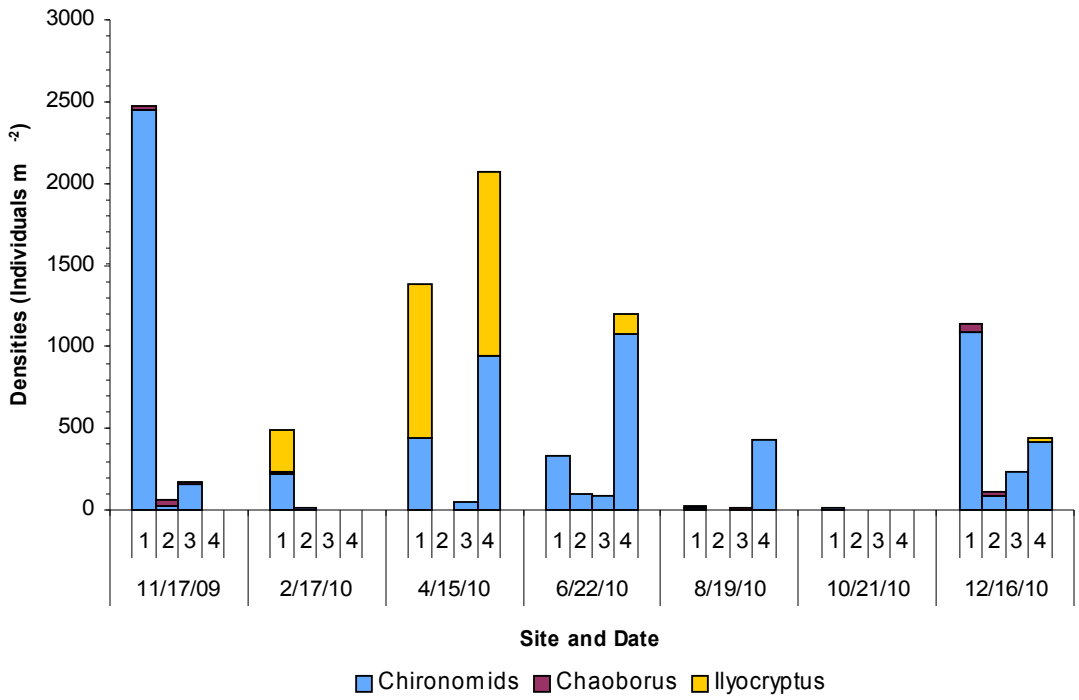

Fig. 2.14. Benthic densities by taxon (genus or family) at the 4 sites: November 17, 2009 to December 16, 2010.

Densities of fly larvae followed trends similar to those for total benthic densities, as fly larvae typically comprised the majority of the benthic assemblage during the sampling period. Chironomid densities were by far the highest at site 1 in November (2445 organisms m^{-2}), with much lower abundances at the other, deeper sites on this 2009 date (prior to addition of the shallowest site 4 to the sampling scheme). Chironomid densities varied strongly over time and across the different sampling sites through the year, although abundances were consistently higher in the shallower sites (sites 1 and 4) (Fig. 2.14). The semi-planktonic predatory midge larvae of the genus *Chaoborus* were present in November 2009, February, August, and December 2010 samples, but were not found in spring, early summer, or October 2010 (Fig. 2.14). When present, its

densities were always low with the highest density being 48 organisms m^{-2} at site 1 in December (however, one of the triplicate samples at site 1 had a phantom midge density of 43 organisms m^{-2} in November 2009).

Ilyocryptus, a benthic cladoceran, was present at site 1 from February to April (peaking at 947 organisms m^{-2} in April) and persisted through June at site 4 after an April peak of 1119 organisms m^{-2} (Fig. 2.14). This benthic cladoceran reappeared at site 4 in December with low densities of 29 organisms m^{-2}. It was not found at sites 2 and 3 (Fig. 2.14). Cole (1955) found a similar result in a study of two lakes where *I. sordidus* was much more abundant in the littoral region than the sublittoral region and was absent from deeper waters that were depleted of oxygen in summer.

Benthic diversity was always quite low. The Simpson's 1-D never exceeded 0.5 (it was frequently below 0.2, Fig. 2.15) and there was never more than 3 families/tribes found at any one site on any date. Diversity may have been found to be higher if specimens had been identified to species.

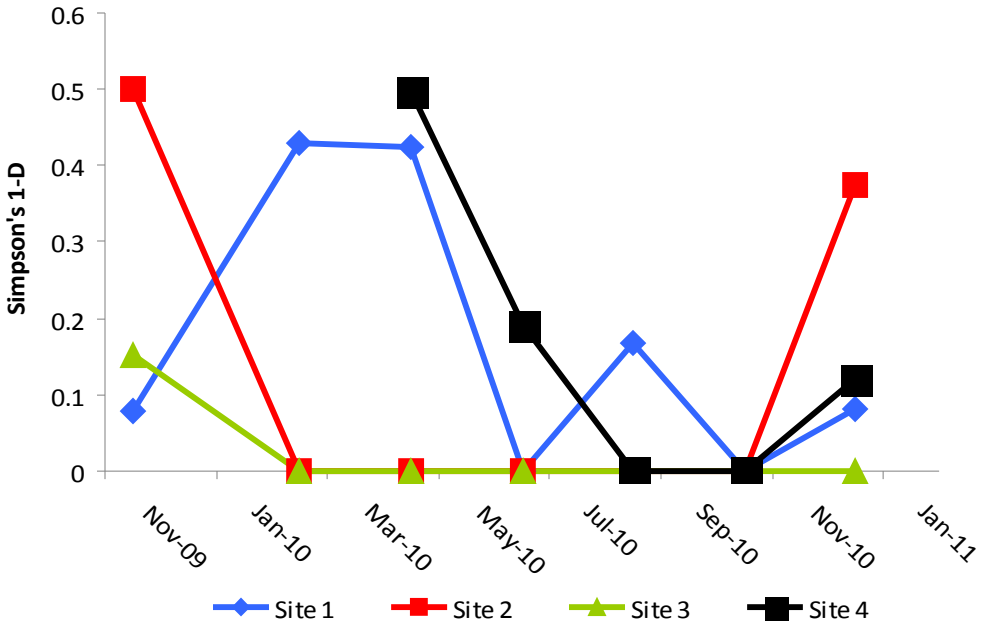

Fig. 2.15. Benthic diversities (down to family or tribe level) at the 4 sites: November 17, 2009 to December 16, 2010.

Stable Isotopes

Stable isotope measurements made on samples taken on December 16, 2010 offer some insight into the communities of Lake Elsinore. Assuming an increase of one trophic level for every increase of about 3 permil for $\delta^{15}N$ (‰), chironomids occupy the lowest trophic level (most likely as detritivores), phantom midges, threadfin shad, and copepods occupy the next highest trophic level (indicating that they are predaceous on organisms at lower trophic levels such as chironomids and herbivorous zooplankton), and carp (as generalist bottom feeders) are at the highest trophic level of these selected species (Fig. 2.16).

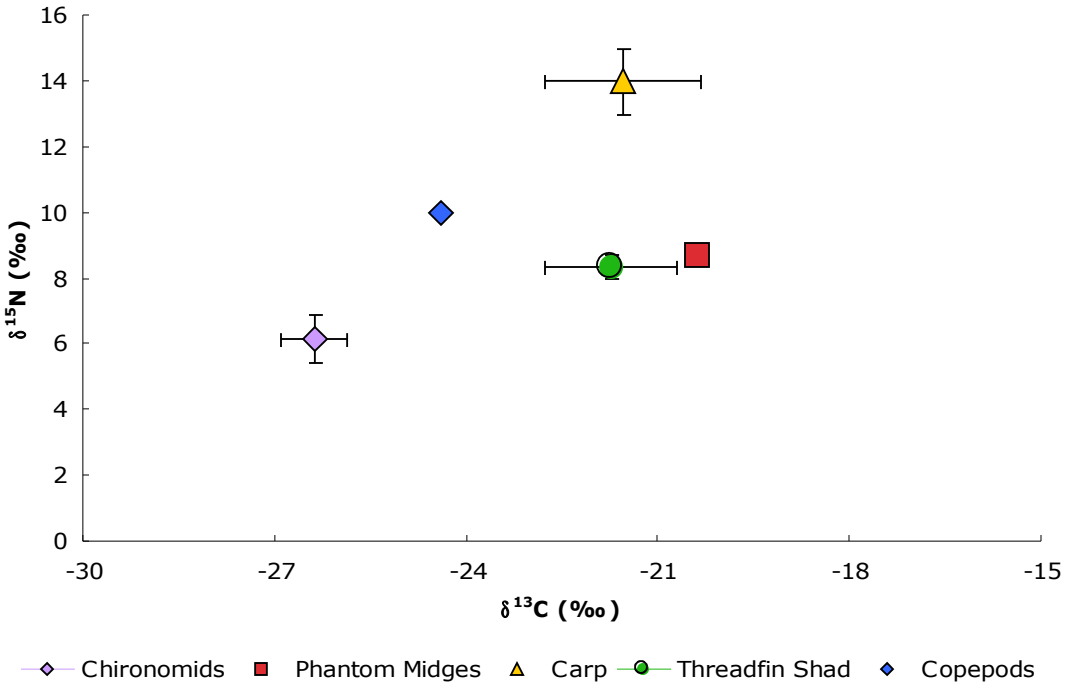

Fig. 2.16. Stable isotope results from December 16, 2010 sampling.

DISCUSSION

Phytoplankton

Overall, the phytoplankton community of Lake Elsinore contained a complex assemblage of genera and species, and followed a seasonal succession that may be expected for a shallow eutrophic lake. The February diatom peak (Fig. 2.4) is consistent with the cool winter temperatures and inputs of runoff containing high concentrations of dissolved Si. Silicon is an important structural component of diatoms' intricate outer shells that can limit future diatom growth as it is depleted from the photic zone and transported to the sediments with the sinking bodies of dead diatoms during an early diatom peak. As is often the case, this diatom peak was followed in April by a small

increase in abundance of green algae (which prefer somewhat higher temperatures than diatoms and do not require Si), and then a bloom of cyanobacteria (which prefer the highest water temperatures and also do not require Si) in June that persisted through the summer. Furthermore, the permanent dominance of Oscillatoriales (such as *P. limnetica*) during summer and autumn has been noted in eutrophic lakes in Central Europe (Berger and Sweers, 1988) as well as Lake Elsinore. Large blooms of the cyanobacterium *P. limnetica*, accompanied by a sharp increase in the phytoplankton biomass around the beginning of summer (and potentially including a high winter population) have been observed in shallow, eutrophic waters near the Black Sea (Soylu and Gonulol, 2009) and in Donghu Lake of China (Lei et al., 2005). *P. limnetica* was also found to be one of the dominant cyanobacteria species in shallow eutrophic lakes (the Loosdrecht Lakes) in the Netherlands which, like Lake Elsinore, lacked submerged aquatic macrophytes due to limitations in light energy (Hofstra & Liere, 1992). Such large populations of *P. limnetica* have resulted in low species diversity in other lakes as well (Soylu and Gonulol, 2009).

The dominance of cyanobacteria that lasted from June through December of 2010 (Fig. 2.4) may be a symptom of accelerated eutrophication of lakes and reservoirs (Moss et al., 1997). Similar phytoplankton assemblages (*P. agardhii*, *P. limnetica*, *C. raciborskii*, and *Aphanizomenon* species) and successions (cyanobacteria dominant in summer through fall) to those observed in Lake Elsinore were found in three eutrophic lakes (shallow and deep) in Eastern Germany where the dominance of filamentous cyanobacteria in the S-1 functional group (temperate-lake, photoadapting solitary filamentous cyanobacteria, Reynolds et al., 2002) seemed to be favored mainly by the characteristics of the underwater light regime (Nixdorf et al., 2003). A shallow,

hypereutrophic lake, Albufera of Valencia, Spain, also showed a similar composition of genera to Lake Elsinore and some similar seasonal trends (Romo and Miracle, 1994). Also, cyanobacteria tend to develop more strongly in summer when water residence times are longer while diatoms and green algae are often dominant in summer during periods when water residence times are short (Wetzel, 2001). Thus, the phytoplankton communities in Lake Elsinore reflect the high nutrient levels and conditions that are characteristic of a terminal basin with long residence times and increasing eutrophication.

In order to use the phytoplankton community to assess the conditions in the lake, the easiest, simplest measurement, Secchi depth, may be the best. Measures of diversity depend on many variables besides trophic state and are, therefore, inadequate for assessment of a lake's trophic level (Danilov & Ekelund, 1999; Karydis & Tsirtsis, 1996; Rakocevic-Nedovic & Hollert, 2005; Spatharis et al., 2011). Similarly, estimates of species richness, while potentially useful in some circumstances (Karydis & Tsirtsis, 1996), may not always be reliable for indicating trophic state (Danilov & Ekelund, 1999). Similar arguments could also be made for using phytoplankton diversity and richness as indicators of water quality, except in regions with severe water problems that only a few particularly hardy species could tolerate. In addition, Secchi depth is a much easier, faster, and less costly measurement that correlates well with certain important aspects of the phytoplankton community. Although Secchi depth measurements may be affected by non-phytoplankton substances in the water column (such as suspended sediment) (Carlson, 1977), they showed a good, negative correlation with total phytoplankton biomass (R^2=0.82) as a power function (Fig. 2.17a). Species richness and diversity, on the other hand, were less strongly negatively correlated with total phytoplankton biomass

(R^2= 0.30 and 0.66 for an exponential and polynomial function respectfully, Fig. 2.17b). Furthermore, relatively deep Secchi depths (>0.5m) were only found with low cyanobacteria biomass (<5000 mg m^{-3}), and cyanobacteria biomass was negatively correlated with Secchi depths (R^2=0.53 for an exponential function, not shown) so deeper Secchi depths indicate fewer cyanobacteria and, therefore, better water quality. Species richness and diversity correlate better with cyanobacteria biomass (R^2= 0.67 and 0.74 for a logarithmic and polynomial function respectfully, not shown), but these are much harder and more expensive measurements to reach the same conclusion as with a Secchi depth measurement. Of course, these correlations are only verified for the year 2010, so they should be used with some caution and may need to be verified further over longer time periods to be certain of their reliability.

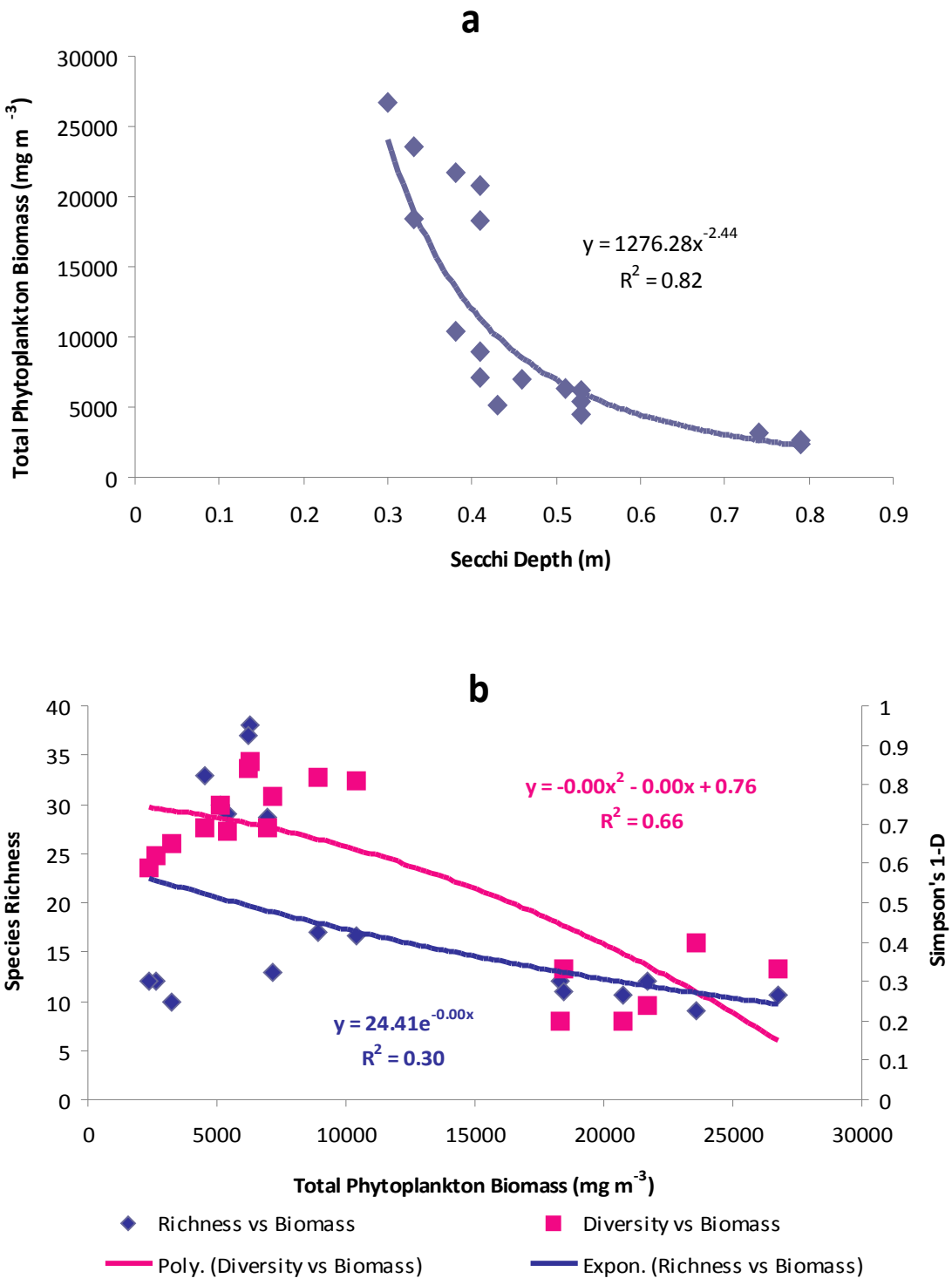

Fig. 2.17. Total phytoplankton biomass versus a) Secchi depth, b) species richness and Simpson's 1-D at the 3 sampling sites.

Zooplankton

Rotifers

The rotifer genera present in the lake tended to be dominated by those more indicative of mesotrophic to eutrophic conditions with a somewhat more mesotrophic assemblage (more *Synchaeta*, *Keratella*, and *Polyarthra*) in cooler months of the year with more rain (Mäemets, 1983). The rotifer genus *Brachionus* and species of *Hexarthra* may indicate eutrophic conditions (Gannon & Stemberger, 1978; Mäemets, 1983). Furthermore, low species diversity, with only a few numerically abundant species, is typical for nutrient-enriched environments (Lennon et al., 2003 and references therein), though the opposite may also be true (Ejsmont-Karabin & Kuczynska-Kippen, 2001; Radwan & Popiolek, 1976; Santangelo et al., 2007). The *Brachionus calyciflorus* peak in October (Fig. 2.11) may be due to a combination of factors including the peak in *Asplanchna brightwelli*. This predatory rotifer may have reduced competition with *B. calyciflorus* by other non-predatory rotifers through its preference of consuming other rotifer species that do not have such long posterior spines as *B. calyciflorus* (Urabe, 1992). The possibility that *Brachionus* (including *B. calyciflorus*) may be uninhibited by cyanobacteria and can eat them (Starkweather, 1981; Starkweather & Kellar, 1983; Walz, 1995 and references therein), and the observation that *B. calyciflorus* could also feed on bacteria (Starkweather et al., 1979) thus providing it with an alternative food source during the cyanobacterial bloom that not all rotifers utilize, are further potential reasons for the dominance of *B. calyciflorus* in October. *Hexarthra* may have been favored so strongly relative to other genera in August (Fig. 2.11) during the cyanobacterial bloom (Fig. 2.4) because it feeds on detritus

and bacteria (Ooms-Wilms, 1997; Pereira et al., 2002) and favors warm waters (Herzig & Koste, 1989). *Hexarthra mira* was found to be associated with the highest total phosphorus level related to low water transparency with more detritus and a higher bacteria content in Funil Reservoir in Brazil (Branco et al., 2002).

Copepods

In general, cyclopoids and their nauplii were more abundant during rainy seasons in Lake Elsinore in 2009-2010 (Fig. 2.12). This trend was also noted by Matsumura-Tundisi and Tundisi (1976) in a shallow, turbulent tropical reservoir (Broa Reservoir) in Brazil. The dominance of copepods and some cladoceran species (such as *Daphnia*) may be typical of tropical eutrophic lakes while, on the other hand, copepod dominance may also be typical of oligotrophic lakes at higher latitudes with rotifers or small cladocerans dominating the zooplankton of eutrophic lakes in these regions (Matsumura-Tundisi & Tundisi, 1976 and references therein). The latter trend seems to fit the results from this study for which copepods were found to be more dominant during cooler periods with fresher water after high rainfall and rotifers were present in relatively high numbers at other times of year.

Cladocerans

The warm temperatures, high levels of cyanobacteria (poor food quality), possible salinity impairments (Veiga-Nascimento, 2004), and predation by the large population of the zooplanktivorous threadfin shad may be hindering the development and maintenance of a large *Daphnia* population in Lake Elsinore.

Zooplankton Summary

As with the phytoplankton community, the zooplankton community indicates that Lake Elsinore is eutrophic. Lake Elsinore resembles the highly eutrophic Loosdrecht Lakes in the Netherlands in its zooplankton community as well as its phytoplankton community. Both showed a rotifer peak following a spring crustacean peak, though the rotifer peak occurred later in the year at Lake Elsinore than in Lake Loosdrecht (Gulati et al., 1992).

Although low zooplankton species richness may be more typical in nutrient rich than nutrient poor waters, the absolute and relative abundances of certain species in the zooplankton community may be better indicators of water quality in lakes than measures of community composition (such as diversity indices) that do not take into account the identity of the organisms present (Gannon & Stemberger, 1978 and references therein). It is not always true that urban pollution or eutrophication lead to reductions in zooplankton diversity and richness (Ejsmont-Karabin & Kuczynska-Kippen, 2001; Santangelo et al., 2007). However, eutrophic waters do tend to have higher total zooplankton abundances than oligotrophic waters (Blancher, 1984; Gannon & Stemberger, 1978 and references therein; Siegfried et al., 1989).

Periods when the zooplankton community of Lake Elsinore is dominated by the rotifer genera *Brachionus* or *Hexarthra* (Fig. 2.11) may indicate the poorest water quality (high cyanobacterial biomass (Fig. 2.4), brackish conditions, low water transparency). In contrast, periods with the persistence of larger cladocerans (*Daphnia* species) and copepods and lower concentrations of rotifers with the rotifer assemblage containing relatively high concentrations of genera such as *Keratella* or *Synchaeta* (which were found in good numbers only at low to moderate phytoplankton biomasses and

proportions of cyanobacteria, Figs. 2.11 and 2.4) may indicate some improvement in water quality. Blancher (1984) surveyed 8 lakes of different trophic states in Florida and found that rotifers dominated the eutrophic lakes while the more oligotrophic systems were dominated by copepods. Also, because calanoid copepods may be best adapted to oligotrophic conditions while cyclopoid copepods and cladocerans are relatively more abundant in eutrophic waters (at least in the Great Lakes) (Gannon & Stemberger, 1978), the small relative abundance of calanoid copepods (Fig. 2.12), appearing only after rains increased water levels significantly (freshening the lake waters somewhat), is a further indication of eutrophic conditions. Their presence may be a sign that conditions in the lake have improved. Veiga-Nascimento (2004) found the highest concentrations of calanoid copepods around October 2003 when nutrient levels were relatively low in the lake. Similarly, during the present study, total N was relatively low and total P was decreasing in the lake in April when calanoids were found (Santa Ana Regional Water Quality Control Board, 2011). However, longer periods of study, more specific species identification of the zooplankton, and more comparisons with phytoplankton and nutrient quantities would be needed to confirm and elucidate these relationships between zooplankton and water quality.

Differences in the zooplankton communities from 2003-2004 to the present study may be due to differences in lake level, salinity, and nutrient concentrations. In the previous study, maximum lake levels were considerably lower than for 2009 through 2010 (~4.5-6 m previously versus ~6.6-8.2 m at site 2) (Veiga-Nascimento, 2004). Based upon an overview of nutrient data from 2010 (not shown, Santa Ana Regional Water Quality Control Board, 2011), it also appears that nutrient and salt concentrations may be lower than in 2003-2004 (Veiga-Nascimento, 2004).

Benthic Invertebrates

The dominance of invertebrates containing red pigments (including *Ilyocryptus* and all but one of the chironomids found) is not surprising considering that Lake Elsinore is eutrophic and, therefore, has a high sediment oxygen demand from the decomposition of organic C in the sediments. Like the hemoglobin in our blood, these red pigments bind and retain oxygen so they help organisms to survive in aquatic environments with low dissolved oxygen (Armitage et al., 1995; Fryer, 1973). In contrast to the permanently benthic Chironomid larvae and *Ilyocryptus*, *Chaoborus* larvae do not stay in the sediments all the time. Phantom midge larvae have air bladders that allow them to migrate up into the more oxygenated water column to prey upon zooplankton and they may be the only insects that are significant members of the plankton (Merritt et al., 2008). In addition to these adaptations, Chironomids and phantom midges also have metabolic adaptations that help them to survive anoxic conditions for a little while (Hoback & Stanley, 2001 and references therein).

Zbikowski and Kobak (2007) examined the macrozoobenthos in shallow, eutrophic, polymictic lakes in Northern Poland and were able to distinguish three different lake types. One of these lake types included deeper, phytoplankton-dominated lakes that had a shaded bottom, high sediment oxygen demand, and a sparse zoobenthos community that was dominated by *Chironomus* and *Chaoborus* larvae. This is similar to what was observed in Lake Elsinore (Fig. 2.14). The genus *Chironomus* is tolerant of pollution and high organic matter concentrations (Morais et al., 2010 and references therein) and has been found to be an indicator of eutrophic or urban conditions in other studies (Morais et al., 2010; Oliveira et al., 2010). Consequently, the benthic assemblage is indicative of low oxygen levels in sediments and of eutrophic

conditions that are likely to result in these low oxygen levels. It may be anoxia in the sediments that was at least partially responsible for the crash in the benthic invertebrate community during late summer and fall of 2010.

Community Summary

A rotifer-dominated zooplankton community can be expected for warm, shallow, eutrophic waters with planktivorous fish and large abundances of cyanobacteria (Beaver and Havens, 2003; Bouvy et al., 2001; He et al., 2006; O'Brien, 1979 and references therein; Rublee, 1992; Shao et al, 2001; Siegfried et al., 1989; Walz, 1995 and references therein; Wright and Shapiro, 1990). Many *Daphnia* species (especially native *Daphnia*) do not fare well in such conditions (DeMott, 1983; Ghadouani et al., 2003; Gulati et al., 2008; Havel and Graham, 2006; Lennon et al., 2001; Moore et al., 1996; Pattinson et al., 2003; Brooks, 1969). Rotifers and other small zooplankters tend to be more tolerant of warm eutrophic conditions and less negatively affected by cyanobacterial filaments and toxins (Bays & Crisman, 1983; Branco et al., 2002; Gannon & Stemberger, 1978 and references therein; Gilbert, 1990; Gliwicz, 1990; Kirk & Gilbert, 1992; Lampert & Sommer, 1997; Moore et al., 1996; Walz, 1995 & references therein). They are also less subject to predation by visually hunting fish species than *Daphnia* species and other large zooplankton (Brooks & Dodson, 1965; Gannon & Stemberger, 1978 and references therein; O'Brien, 1979 & references therein; Shao et al., 2001; Vakkilainen et al., 2004). Often, it is the combination of food limitation (dominance by filamentous cyanobacteria with little to no edibility) and predation by young fish around the beginning of summer that result in the large *Daphnia* species being wiped out to a greater extent than smaller zooplankters (Gulati et al., 2008). Other mechanisms such

as competition between rotifers and cladocerans may also be responsible for the frequent observation in natural zooplankton communities that planktonic rotifers tend to only be common when small cladocerans, but not large ones, are present (MacIsaac & Gilbert, 1989). Small cladocerans tend to feed on small edible algae without feeding on cyanobacteria, which may be how smaller cladocerans can coexist with cyanobacteria better than large ones (Hanazato, 1996). Therefore, the fact that the planktonic cladoceran community is dominated by smaller species (*D. lumholtzi*, *D. ambigua*, *Bosmina*, and the chydorids *Alona* and *Leydigia*) is a further indication, not just of the presence of fish, but of the eutrophic conditions in the lake and relatively abundant cyanobacteria.

The zooplankton community is influenced, not just by physical and chemical conditions in the lake, but also by the phytoplankton community that grows under these conditions and by the presence of other zooplankton species and of zooplanktivorous fish. There tended to be a fairly strong significant (generally positive) correlation (R^2=0.74 for a polynomial function, graph not shown, Table 2.2) between the proportion of cyanobacterial biomass relative to total phytoplankton biomass and the proportion of *Brachionus* + *Hexarthra* abundance relative to total rotifer abundance. The decreasing portion of the polynomial function at low proportions of cyanobacteria may be indicative of one or more different species of *Brachionus* in the lake that are less tolerant of cyanobacteria and, therefore, may be indicators of better water quality. *Brachionus variabilis* was found during the period with high relative diatom biomass and the lowest relative cyanobacteria biomass. Similarly, there was also a significant correlation between total phytoplankton biomass and total zooplankton abundance (R^2=0.59 for a polynomial function, graph not shown, Table 2.2) with the lowest zooplankton

abundances at intermediate phytoplankton biomasses. This suggests at least two different types of zooplankton assemblages, those that are unable to compete and survive during cyanobacterial blooms (during which the highest total phytoplankton biomasses were reached) and those that have a competitive advantage during these algal blooms because they can tolerate high amounts of cyanobacteria.

Table 2.2. Summary of regression results used in the Community Summary (n = 18). All correlations are significant at α=0.05

Dependent Variable	Independent Variable	Regression Line	R^2
Proportion of cyanobacterial biomass relative to total phytoplankton biomass	Proportion of *Brachionus* plus *Hexarthra* abundance relative to total rotifer abundance	polynomial	0.74
Total phytoplankton biomass	Total zooplankton abundance	polynomial	0.59
Proportion of diatom biomass relative to total phytoplankton biomass	Copepod abundance	polynomial	0.77
Proportion of diatom biomass relative to total phytoplankton biomass	Cladoceran abundance	polynomial	0.67
Biomass of green algae plus diatoms	Copepod abundance	polynomial	0.64
Proportion of green algae plus diatom biomass relative to total phytoplankton biomass	Copepod abundance	polynomial	0.75
Proportion of green algae plus diatom biomass relative to total phytoplankton biomass	Cladoceran abundance	polynomial	0.47

As with zooplankton in general, copepod and cladoceran abundances also appear to be influenced by the phytoplankton community. Both copepod and cladoceran abundance showed fairly strong significant polynomial relationships with diatom biomass as a proportion of the total phytoplankton biomass (R^2=0.77 and 0.67 respectfully, graph not shown, Table 2.2). Their abundances tended to be highest at intermediate relative

proportions of diatoms (and total diatom biomass as well). Furthermore, copepod abundance was significantly related to the total biomass of green algae plus diatoms (R^2=0.64 for a polynomial function, graph not shown, Table 2.2) and both copepod and cladoceran abundance were significantly related to the proportion of green algae plus diatom biomass relative to total phytoplankton biomass (R^2=0.75 and 0.47 respectfully for polynomial functions, graph not shown, Table 2.2). This suggests that diatoms are an important food source for the less predatory young cyclopoid copepods (nauplii) and cladocerans and/or that both these types of zooplankton are best adapted to conditions that also favor diatoms and green algae. Cladoceran abundance tends to decrease at the higher end of diatom and green algae biomass and proportions. This is consistent with the lower food thresholds of the larger *Daphnia* species (Gliwicz, 1990 and references therein) that comprised the majority of the peak cladoceran abundance. Copepod abundance, on the other hand, generally increased with diatom and green algae biomass and proportions. A slight decrease in copepod abundance at the highest diatom biomasses and proportions represents the data from February when the phytoplankton community was comprised mainly of diatoms, most of which were of the relatively large and long genus *Skeletonema*, which may not have been a favorable food source. Also, nauplii and adults of the genus *Acanthocyclops* may have greater survival (and reproduction for adults) at lower algal densities (Garcia et al., 2011). Both copepod and cladoceran abundances peaked in April when the biomasses and relative proportions of the green algae genera *Scenedesmus* and *Cosmarium* and of the diatom genus *Cyclotella* also peaked. These may or may not be important food sources for these zooplankters. Since abundances of both copepods and cladocerans decrease at high total phytoplankton and cyanobacteria biomass healthy populations of copepods

and cladocerans may indicate better water quality with more diatoms and green algae in the lake than cyanobacteria and relatively low phytoplankton biomasses.

There is also a potential interaction between copepods and diatoms of the genus *Synedra*, which Ban et al. (1997) report to be harmful to copepods. *Synedra* was found in significant quantities (40-131 mg m^{-3}) only at sites 1 and 2 in April. It is notable that, in April, total copepod concentrations (nauplii, copepodids, and adults) at sites 1 and 2 (107.2 individuals L^{-1} and 84.8 individuals L^{-1} respectfully) were considerably lower than at site 3 (162.4 individuals L^{-1}) where this diatom was not found and that total copepod concentrations at sites 1 and 2 showed only a slight increase from February to April (102.0 individuals L^{-1} to 107.2 individuals L^{-1} & 84.4 individuals L^{-1} to 84.8 individuals L^{-1} respectively), while, at site 3, copepod concentrations nearly doubled (92.5 individuals L^{-1} to 162.4 individuals L^{-1}) in this same time frame.

Grazing by the zooplanktivorous threadfin shad does not appear to have had much of an impact on the zooplankton community in general as the zooplankton community composition in December 2010 (after a roughly 10-fold increase in the threadfin shad population since March 2010; Anderson et al., 2011) was similar to that of November 2009 (Fig. 2.9) shortly after a crash in the shad population. The threadfin shad may have, however, contributed to the decimation of the population of larger cladocerans (*Daphnia* species not including the tiny *D. ambigua*), which were only found in early 2010 before the shad population boom (Fig. 2.14).

The rotifer and copepod communities may be an example of how one zooplankton group can influence another. Overall, there is a general trend toward decreasing rotifer abundance with increasing copepod abundance. This relationship is somewhat evident in Figure 2.9, which shows that rotifers are usually uncommon when

copepods are at relatively high abundances and vice versa. This may indicate predatory pressure of cyclopoid copepods on rotifers (Miracle et al., 2007). However, this trend may also be due, at least in part, to each group being favored by different conditions in the lake that are generally more unfavorable to the other group.

The low dissolved oxygen levels in the lake that followed a phytoplankton bloom and resulted in a fish kill in the summer of 2009 may have been detrimental to other aerobic organisms besides the fish in the lake. Total zooplankton abundances, as well as abundances for individual groups of zooplankton (rotifers, copepods, cladocerans), are all relatively low in November 2009, but then gradually increase (Fig. 2.9). Yet, other factors are certainly also responsible for these trends. The copepod community appears to have also been adversely affected by the 2010 cyanobacteria bloom as copepod abundance plummeted at its beginning in June 2010 and remained low throughout the rest of the year as cyanobacteria continued to dominate the phytoplankton biomass (Figs. 2.4 & 2.13). This is perhaps another indication that the young copepods in the lake feed on non-cyanobacterial phytoplankton.

The benthic community appeared to be influenced more by conditions in the sediments than by other communities in the lake and are, therefore, the best indicators of variability at the bottom of the lake throughout time and space. High abundances and relative abundances of red invertebrates indicate low oxygen while a complete lack of benthic invertebrates (as in deeper parts of the lake in summer) may suggest strong anoxia in the sediments.

Stable Isotopes

The stable isotope results (Fig. 2.16) are comparable to those found in a study of the foodweb in Copper Basin Reservoir (McCullough et al., 2010). In both studies, the carp plot at about -21 ‰ on the carbon axis and 14 ‰ on the nitrogen axis, the chironomids are around -25 ‰ on the carbon axis, and the copepods are at about 11 ‰ on the nitrogen axis. However, at Lake Elsinore, the chironomids have a lower nitrogen value (indicating a lower trophic level for the different species of midges at Lake Elsinore) and the copepods have a less negative carbon value (suggesting different carbon sources between the two lakes) than in Copper Basin Reservoir (McCullough et al., 2010).

Unfortunately, due to difficulties in collecting enough pure biomass for analysis, not all components of the food web were analyzed for this study. Small, herbivorous zooplankton (such as rotifers) are among these missing components. However, because they are primary consumers that graze upon the primary producers (phytoplankton), they could be expected to have $\delta^{15}N$ values in the range of one trophic level (3 permil (‰)) lower than the secondary consumers (copepods, threadfin shad, and phantom midges) that feed at least partially on them. Thus, rotifers and the small cladocerans that were present in the lake in December would probably plot on a level similar to the chironomids. However, since the chironomids are part of a benthic food chain and receive C from a difference source than members of the pelagic food web (phytoplankton, zooplankton, planktonic phantom midges, and fish), the herbivorous zooplankton would have less negative $\delta^{13}C$ values than the chironomids, plotting more in the range of –21 to –24 permil (‰) with the other pelagic organisms.

The information provided by studies of food webs allows inferences to be made about what may happen to organisms on one trophic level if those on another are manipulated. For example, adult cyclopoid copepods eat zooplankton. If they feed preferentially on one type of zooplankton over another, they may have a significant influence over the composition of the zooplankton community (Brandl, 2005; Nagata and Hanazato, 2006). Different rotifers and other zooplankters tend to feed on different types of phytoplankton or other food particles (Walz, 1995 and references therein), so, in a similar fashion, they may be expected to influence the composition of the phytoplankton community. Thus, any variable that affects the population of cyclopoid copepods in the lake may, in turn, affect the zooplankton community and the phytoplankton community. Through these trophic interactions, the composition and diversity of each community in the lake may be reliant on one or more other communities, and the lake ecosystem functions as a unit with all communities potentially responding to changes in the environment or in the populations of other organisms in the lake. A more complete analysis of the food webs through stable isotopes would help to understand how these lake-wide shifts could occur as conditions in the lake change.

SUMMARY AND CONCLUSIONS

Of the three communities examined in Lake Elsinore, the phytoplankton community exhibited the greatest variation in diversity and abundances. Diversity was very low (0.2 or less) during cyanobacterial blooms (summer 2010) and high (up to ~0.8) when green algae and/or diatoms were a prominent component of the biomass (winter and spring of 2010). Cyanobacteria were the dominant component of the phytoplankton

biomass from June through December 2010 and were responsible for a high peak in biomass during the summer of 2010, with values reaching 20,000-30,000 mg m^{-3}. This bloom was comprised almost entirely of the species *Pseudanabaena limnetica* (formerly known as *Oscillatoria limnetica*). Phytoplankton biomasses were much lower at other times of year, averaging around 5,000 mg m^{-3} in February and April, 2,500 mg m^{-3} in October, and 5,000-10,000 mg m^{-3} in December.

The phytoplankton community was similar to that observed almost a decade ago. Oza (2003) also found diatoms to be a significant component of the phytoplankton community during February through mid spring and observed a shift to cyanobacteria dominance (almost entirely of the genus *Oscillatoria*) that was evident by June. This shift toward cyanobacteria was accompanied by a decrease in Simpson's diversity that is comparable to that observed recently. In both studies, green algae were never very abundant, but they peaked to noticeable numbers in spring (Oza, 2003). Based on these past and present findings, *P. limnetica* appears to be a problem species in Lake Elsinore and it seems desirable that the green algae and diatom populations in the lake be maintained in order to improve diversity, reduce phytoplankton biomass, and improve the food supply available to the algivorous zooplankton.

While somewhat less drastically variable than the other two communities, the zooplankton community did show definite seasonal variation in composition and abundance. Zooplankton abundances increased through the winter and spring of 2010, but there was a strong decline in summer with the increasing temperatures and the *P. limnetica* bloom. This decline was followed by a sharp, but short-lived, peak in rotifer densities that was largely due to the species *B. calyciflorus*. Similarly to the sharp changes in zooplankton abundances, the warm temperatures and high cyanobacteria

biomass of summer 2010 also appear to have contributed to a drastic alteration in the zooplankton community composition. After April 2010, the zooplankton community shifted from being almost entirely copepods in fall 2009, to a mix of copepods and rotifers with a few cladocerans in winter through spring 2010, and, finally, to almost entirely rotifers (mainly the genera *Hexarthra* then *Brachionus*) in summer and early fall 2010. By December 2010, once temperatures had cooled and the cyanobacteria population had decreased, the zooplankton community returned to roughly the same composition as in fall 2009, though the taxa were different (particularly for the rotifers).

The zooplankton community has changed since 2003-2004, with at least three new species, *Daphnia lumholtzi*, *Polyarthra* sp., and *Hexarthra* sp., appearing that were not reported previously (Veiga-Nascimento, 2004). However, there were also some similarities between this and the previous study. Perhaps the most notable is the consistently small cladoceran population. As observed by Veiga-Nascimento (2004) under broadly similar conditions, cladocerans were absent through much of the year. However, small numbers of *Daphnia lumnholtzi* and other *Daphnia* species were found in the spring, perhaps delivered with the winter-spring rains and runoff into the lake (*D. lumholtzi* has previously been found in Canyon Lake in moderate abundances). Warm temperatures, poor food quality, possible salinity impairments, and predation by threadfin shad are thought to have constrained development of a large *Daphnia* population in Lake Elsinore.

The high abundances of rotifers combined with a cladoceran community comprised mainly of smaller species and genera can be expected for warm, shallow, eutrophic waters with planktivorous fish and large abundances of cyanobacteria. Ostracods and bivalves were a negligible proportion of the zooplankton abundance, but

it is important that quagga veligers were found in August 19, 2010 site 1 samples with an average abundance of 0.0976 organisms/L (or approximately one veliger for every 10.25 L). Though no other veligers have been found before or since then or at any other site in the lake, it is still unclear whether or not they may establish themselves in the lake at a later date.

The benthic community was the most impoverished of the three studied. Benthic diversity was very low as it consisted almost entirely of organisms (Chironomids and a benthic cladoceran, *Ilyocryptus*) with red hemoglobin-like pigments that aid in the binding and transfer of oxygen. This reflects the low DO concentrations in the bottom waters and sediments of the lake. Densities of benthic invertebrates varied strongly between sites and sampling dates. However, densities were consistently higher at the shallower sites with higher dissolved oxygen and lower organic carbon concentrations in the sediments. The benthic community density peaked in fall 2009, spring 2010, and winter 2010 during periods with cooler, more oxygenated water, but crashed in summer. The summer paucity of benthic organisms was, perhaps, due to depleted oxygen levels resulting from warmer waters, some thermal stratification, and aerobic decomposition of organic matter (probably mostly from phytoplankton blooms) in the sediments.

Considering all of the above, the ecology and food web in Lake Elsinore is consistent with a eutrophic lake containing a phytoplankton community with low species richness, low diversity, and strong blue-green algae production (especially *P. limnetica*) during much of the year. At the same time, intense grazing pressure by threadfin shad and other effects (e.g., low DO, poor food quality) constrains the development of a beneficial zooplankton community, and thus limits grazing on phytoplankton and natural

control of algal levels in the lake. Low dissolved oxygen levels may also be limiting the capacity for benthic invertebrate production in the lake.

With the knowledge gained from this study, the organisms responsible for the ecological functioning of Lake Elsinore, and for the water quality issues that may arise because of it, are now better known and more fully understood. Furthermore, additional insight has been gained concerning how these organisms interact and affect one another because multiple communities were studied simultaneously. Therefore, the information in this chapter provides a basis for improving future ecological management and water quality.

REFERENCES

Anderson, M.A. 2010. *Bathymetric, Sedimentological and Retrospective Water Quality Analysis to Evaluate Effectiveness of the Lake Elsinore Recycled Water Pipeline Project: Summer, 2010*. Draft Final Report to the Lake Elsinore & San Jacinto Watersheds Authority. pg. 41.

Anderson, M.A., M. Tobin, and J. Tobin. 2011. *Biological Monitoring of Lake Elsinore*: *June, 2011*. Draft Final Report to the Lake Elsinore-San Jacinto Watershed Authority and Santa Ana Watershed Project Authority. pgs. 40-44.

Armitage, P.D., P.S. Cranston, and L.C.V. Pinder (Eds.). 1995. The Chironomidae. The Biology and Ecology of Non-Biting Midges. Chapman and Hall, London.

Azemar, F. et al. 2010. Rotifers in the Schelde estuary (Belgium): a test of taxonomic relevance, Journal of Plankton Research 32:981-997.

Ban, S.H., C. Burns, J. Castel, … 1997. The paradox of diatom-copepod interactions, Marine Ecology-Progress Series 157:287-293.

Bays, J.S., and T.L. Crisman. 1983. Zooplankton and trophic state relationships in Florida lakes, Canadian Journal of Fisheries and Aquatic Sciences 40:1813-1819.

Beaver, J. and K. Havens. 2003. Seasonal and spatial variation in zooplankton community structure and their relation to possible controlling variables in Lake Okeechobee, Freshwater Biology 36:45-56.

Berger, C., and H.E. Sweers. 1988. The Ijsselmeer and its phytoplankton with special attention to the suitability of the lake as a habitat for *Oscillatoria-agardhii* GOM, Journal of Plankton Research 10:579-599.

Blancher, E.C. 1984. Zooplankton-trophic state relationships in some north and central Florida lakes, Hydrobiologia 109:251-263.

Boggero A. et al. 2006. Littoral chironomid communities of Alpine lakes in relation to environmental factors, Hydrobiologia 562:145-165.

Bouvy, M., M. Pagano, and M. Troussellier. 2001. Effects of a cyanobacterial bloom (Cylindrospermopsis raciborskii) on bacteria and zooplankton communities in Ingazeira reservoir (northeast Brazil), Aquatic Microbial Ecology 25:215-227.

Branco, C.W.C. 2002. Limnological features of Funil Reservoir (R.J., Brazil) and indicator properties of rotifers and cladocerans of the zooplankton community, Lakes & Reservoirs: Research & Management 7:87-92.

Brandl, Z. 2005. Freshwater copepods and rotifers: Predators and their prey, Hydrobiologia 546:475-489.

Britton, C.H., R.E. Pruitt, C.A. Lembi. 2005. Report for 2005IN174B: Rapid detection of toxic and taste and odor causing cyanobacteria in Indiana surface water, abstract booklet produced by the Aquatic Plant Management Society, San Antonio, TX, page 35.

Brooks, J.L. 1969. Eutrophication and changes in the composition of the zooplankton, Eutrophication: causes, consequences, correctives, National Academy of Sciences, Washington, D.C., pp. 236-255.

Brooks, J.L., and S.I. Dodson. 1965. Predation, body size, and composition of plankton, Science 150:28-35.

Callisto, M., J.F. Goncalves, and P. Moreno. 2005. Invertebrados aquaticos como bioindicadores, In GOULART, EMA. (Eds.). Navegando o Rio das Velhas das Minas aos Gerais. p. 555-567.

Carlson, R.E. 1977. A trophic state index for lakes, Limnology and Oceanography 22:361-369.

Chang, K.H., M. Sakamoto, and T. Hanazato. 2005. Impact of pesticide application on zooplankton communities with different densities of invertebrate predators: An experimental analysis using small-scale mesocosms, Aquatic Toxicology 72:373-382.

Clesceri, L.S., A.E. Greenberg, and A.D. Eaton (Eds.). 1998. Standard Methods for the Examination of Water and Wastewater, 20th Edition, American Public Health Association, pgs. 10-5, 10-7, 10-14, 10-28, 10-60, 10-64.

Cole, G.A. 1955. An ecological study of the microbenthic fauna of two Minnisota lakes, Am. Midl. Nat. 53:213-230.

Cook, P.L.M., K.T. Aldridge, S. Lamontagne, J.D. Brooks. 2010. Retention of nitrogen, phosphorus and silicon in a large semi-arid riverine lake system, Biogeochemistry 99:49-63.

Danilov, R. and N.G.A. Ekelund. 1999. The efficiency of seven diversity and one similarity indices based on phytoplankton data for assessing the level of eutrophication in lakes in central Sweden, Science of the Total Environment 234:15-23.

Della Bella, V. and L. Mancini. 2009. Freshwater diatom and macroinvertebrate diversity of coastal permanent ponds along a gradient of human impact in a Mediterranean eco-region, Hydrobiologia 634:25-41.

DeMott, W.R. 1983. Seasonal succession in a natural *Daphnia* assemblage, Ecological Monographs 53:321-340.

DeMott, W.R. 1999. Foraging strategies and growth inhibition in five daphnids feeding on mixtures of a toxic cyanobacterium and a green algae, Freshwater Biology 42:263-274.

Dodson, S.I., S.E. Arnott, and K.L. Cottingham. 2000. The relationship in lake communities between primary productivity and species richness, *Ecology* 81:2662-2679.

Ejsmone-Karabin, J. and N. Kuczynska-Kippen. 2001. Urban rotifers: structure and densities of rotifer communities in water bodies of the Poznan agglomeration area (western Poland), Hydrobiologia 446-447:165-171.

Fryer, G. 1973. Evolution and adaptive radiation in the Macrothricidae (Crustacea: Cladocera): A study in comparative functional morphology and ecology, Philosophical Transactions of the Royal Society of London. Series B, Biological Sciences 269:243-261.

Gannon, J.E., and R.S. Stemberger. 1978. Zooplankton (especially crustaceans and rotifers) as indicators of water quality, Transactions of the American Microscopical Society 97:16-35.

Garcia, C.E., S. Nandini, and S.S.S. Sarma. 2011. Demographic characteristics of the copepod *Acanthocyclops americanus* (Sars, 1863) (Copepoda: Cyclopoida) fed mixed algal (*Scenedesmus acutus*)-rotifer (*Brachionus havanaensis*), Hydrobiologia 666:59-69.

Ghadouani, A., B. Pinel-Alloul, and E.E. Prepas. 2003. Effects of experimentally induced cyanobacterial blooms on crustacean zooplankton communities, Freshwater Biology 48:363-381.

Gilbert, J.J. 1990. Differential-effects of *Anabaena-affinis* on cladocerans and rotifers – mechanisms and implications, Ecology 71:1727-1740.

Gliwicz, Z.M. 1990. Why do cladocerans fail to control algal blooms, Hydrobiologia 200:83-97.

Gorham, P.R., S. McNicholas, and E.A.D. Allen. 1982. Problems encountered in searching for new strains of toxic planktonic cyanobacteria, South African Journal of Science 78:357-362.

Gulati, R.D. et al. 1992. The dynamics and role of limnetic zooplankton in Loosdrecht lakes (the Netherlands), Hydrobiologia 233:69-86.

Gulati, R.D., L.M.D. Pires, and E. Van Donk. 2008. Lake restoration studies: Failures, bottlenecks and prospects of new ecotechnological measures, Limnologica 38:233-247.

Hanazato, T. 1991. Effects of repeated application of carbaryl on zooplankton communities in experimental ponds with or without the predator *Chaoborus*, Environmental Pollution 74:309-324.

Hanazato, T. 1996. Toxic cyanobacteria and the zooplankton community, from Toxic Microcystis, Chapter 5, editors Watanabe, M.F., K. Harada, W.W. Carmichael, and H. Fujiki, pgs. 79-98.

Hanazato, T. 1998. Response of a zooplankton community to insecticide application in experimental ponds: a review and the implications of the effects of chemicals on the structure and functioning of freshwater communities, Environmental Pollution 101:361-373.

Hanazato, T. and F. Kasai. 1994. Effects of the organophosphorus insecticide fenthion on phyto- and zooplankton communities in experimental ponds, Environmental Pollution 88:293-298.

Hanazato, T. and M. Yasuno. 1990. Influence of time of application of an insecticide on recovery patterns of a zooplankton community in experimental ponds, Archives of Environmental Contamination and Toxicology 19:77-83.

Havel, J.E. and J.L. Graham. 2006. Complementary population dynamics of exotic and native *Daphnia* in North American reservoir communities, Archiv Fur Hydrobiologie 167:245-264 DOI 10.1127/0003-9136/2006/0167-0246.

Havens, K.E. and T. Hanazato. 1993. Zooplankton community responses to chemical stressors – a comparison of results from acidification and pesticide contamination research, Environmental Pollution 82:277-288.

He, X., M. Scheurell, P. Soranno, and R. Wright. 2006. Recurrent response patterns of a zooplankton community to whole-lake fish manipulation, Freshwater Biology 32:61-72.

Herzig, A. and W. Koste. 1989. The development of *Hexarthra* spp in a shallow alkaline lake, Hydrobiologia 186:129-136.

Hoback, W.W. and D.W. Stanley. 2001. Insects in hypoxia, Journal of Insect Physiology 47:533-542.

Hofstra, J.J., and L.V. Liere. 1992. The state of the environment of the Loosdrecht lakes, Hydrobiologia 233:11-20.

Karydis, M. and G. Tsirtsis. 1996. Ecological indices: a biometric approach for assessing eutrophication levels in the marine environment, Science of the Total Environment 186:209-219.

Kirby, M.E., S.P. Lund, M.A. Anderson, and B.W. Bird. 2007. Insolation forcing of Holocene climate change in Southern California: a sediment study from Lake Elsinore. *J. Paleolimnol.* 38:395-417.

Kirk, K.L., and J.J. Gilbert. 1992. Variation in herbivore response to chemical defenses – zooplankton foraging on toxic cyanobacteria, Ecology 73:2208-2217.

Lampert, W., and U. Sommer. 1997. Limnoecology: the Ecology of Lakes and Streams. Oxford Univ. Press, Oxford, 382 pp.

Landsberg, J.H. 2002. The effects of harmful algal blooms on aquatic organisms, Reviews in Fisheries Science 10:113-390.

Lawson, R. and M.A. Anderson. 2007. Stratification and mixing in Lake Elsinore, California: An assessment of axial flow pumps for improving water quality in a shallow eutrophic lake, *Water Research* 41:4457-4467.

Lei, A.P., Z.L. Hu, J. Wang, Z.X. Shi, and F.Y.N. Tam. 2005. Structure of the phytoplankton community and its relationship to water quality in Donghu Lake, Wuhan, China, Journal of Integrative Plant Biology 47:27-37.

Lennon, J.T., V.H. Smith, and K. Williams. 2001. Influence of temperature on exotic *Daphnia lumholtzi* and implications for invasion success, Journal of Plankton Research 23:425-434.

Lennon, J.T., V.H. Smith, A.R. Dzialowski. 2003. Invasibility of plankton food webs along a trophic state gradient, OIKOS 103:191-203.

Mäemets, A. 1983. Rotifers as indicators of lake types in Estonia, Hydrobiologia 104:357-361.

Matsumura-Tundisi, T. and J.G. Tundisi. 1976. Plankton studies in a lacustrine environment, *Oecologia* 25:265-270.

McCormick, P.V. and J. Cairns Jr. 1994. Algae as indicators of environmental change, Journal of Applied Phycology 6:509-526.

McCullough, J., J. Sickman, and M. Anderson. 2010. Trophic position and the impact of quagga mussels on the food web in the Colorado River Aqueduct, Review copy submitted to *Freshwater Biology*.

Merritt, R.W., K.W. Cummins, and M.B. Berg (Eds.). 2008. An Introduction to the Aquatic Insects of North America, Fourth Edition, Kendall Hunt, pg. 63.

Miracle, M.R., M.T. Alfonso, and E. Vicente. 2007. Fish and nutrient enrichment effects on rotifers in a Mediterranean shallow lake: a mesocosm experiment, Hydrobiologia 593:77-94.

Moore, M.V., C.L. Folt, and R.S. Stemberger. 1996. Consequences of elevated temperatures for zooplankton assemblages in temperate lakes, Archiv Fur Hydrobiologie 135:289-319.

Morais, S.S. et al. 2010. Diversity of larvae of littoral Chironomidae (Diptera: Insecta) and their role as bioindicators in urban reservoirs of different trophic levels, Brazilian Journal of Biology 70:995-1004.

Moss, B., J. Madgwick, and G. Philips. 1997. A guide to the restoration of nutrient-enriched shallow lakes, W.W. Hawes, UK.

Nagata, T. and T. Hanazato. 2006. Different predation impacts of two cyclopoid species on a small-sized zooplankton community: An experimental analysis with mesocosms, Hydrobiologia 556:233-242.

Nixdorf, B., U. Mischke, and J. Rucker. 2003. Phytoplankton assemblages and steady state in deep and shallow eutrophic lakes – an approach to differentiate the habitat properties of Oscillatoriales, Hydrobiologia 502:111-121.

Nollet, L.M.L. (Ed.). 2000. Handbook of Water Analysis, Marcel Dekker, Inc. pgs. 150-151.

O'Brien, W.J. 1979. Predator-prey interaction of planktivorous fish and zooplankton, American Scientist 67:572-581.

Oliveira, V., R. Martins, and R. Alves. 2010. Evaluation of water quality of an urban stream in Southeastern Brazil using Chironomidae larvae (Insecta: Diptera), Neotropical Entomology 39:873-878.

Ooms-Wilms, A.L. 1997. Are bacteria an important food source for rotifers in eutrophic lakes?, Journal of Plankton Research 19:1124-1141.

Ostroumov, S.A. 2005. Some aspects of water filtering activity of filter-feeders, Hydrobiologia 542:275-286.

Oza, H.I., 2003. *Nutrient levels and phytoplankton abundance in Canyon Lake and Lake Elsinore, CA*. M.S. Thesis. Univ. of California, Riverside, 112-147.

Pattinson, K.R., J.E. Havel, and R.G. Rhodes. 2003. Invasibility of a reservoir to exotic *Daphnia lumholtzi*: experimental assessment of diet selection and life history responses to cyanobacteria, Freshwater Biology 48:233-246.

Patten, B.C., J.J. Norcross, D.K. Young, and C.L. Rutherford. 1964. Some experimental characteristics of dark and light bottles, ICES Journal of Marine Science 28:335-353.

Pereira, R., A.M.V.M. Soares, R. Ribeiro, F. Goncalves. 2002. Assessing the trophic state of Linhos lake: a first step towards ecological rehabilitation, Journal of Environmental Management 64:285-297.

Pratt, D.M. and H. Berkson. 1959. Two sources of error in the oxygen light and dark bottle method, Limnology and Oceanography 4:328-334.

Radwan, S. and B. Popiolek. 1976. Planktonic rotifers as indicators of lake trophy, Ann. UMCS, sec. C, 31, 221-235.

Rakocevic-Nedovic, J. and H. Hollert. 2005. Phytoplankton community and chlorophyll a as trophic state indices of Lake Skadar (Montenegro, Balkan), Environmental Science and Pollution Research 12:146-152.

Reynolds, C.S. et al. 2002. Towards a functional classification of the freshwater phytoplankton, *Journal of Plankton Research* 24:417-428.

Romo, S., and M.R. Miracle. 1994. Population-dynamics and ecology of subdominant phytoplankton species in a shallow hypereutrophic lake (Albufera of Valencia, Spain), Hydrobiologia 273:37-56.

Rublee, P.A. 1992. Community structure and bottom-up regulation of heterotrophic microplankton in arctic LTER lakes, Hydrobiologia 240:133-141.

Sahuquillo, M., M.G.G. Melao, M.R. Miracle. 2007. Low filtering rates of *Daphnia magna* in a hypereutrophic lake: Laboratory and in situ experiments using synthetic microspheres, Hydrobiologia 594:141-152.

Sala, O.E., R.B. Jackson, H.A. Mooney, and R.W. Howarth (Eds.). 2000. Methods in Ecosystem Science, Springer-Verlag New York, Inc. pgs. 74-75.

San Jacinto River Watershed Council. 2005. The San Jacinto Watershed Component of the Santa Ana Integrated Watershed Plan: Prop 50, Chapter 8: Planning Grant Application, pgs. 16, 24, 34.

Santa Ana Regional Water Quality Control Board. 2011. *Lake Elsinore and Canyon Lake Nutrient TMDL Annual Water Quality Report: June 2011*. Draft Report for the Santa Ana Regional Water Quality Board. pgs. 3-6 to 3-7.

Santangelo, J.M. et al. 2007. Zooplankton responses to sandbar opening in a tropical eutrophic coastal lagoon, Estuarine, Coastal and Shelf Science 71:657-668.

Shao, Z.J., P. Xie, and Z.G. Yan. 2001. Long-term changes of planktonic rotifers in a subtropical Chinese lake dominated by filter-feeding fishes, Freshwater Biology 46:973-986.

Siegfried, C.A., J.A. Bloomfield, and J.W. Sutherland. 1989. Planktonic rotifer community structure in Adirondack, New York, U.S.A. lakes in relation to acidity, trophic status and related water quality characteristics, Hydrobiologia 175:33-48.

Sladecek, V. 1983. Rotifers as indicators of water quality, Hydrobiologia 100:169-201.

Soylu, E.N., and A. Gonulol. 2009. Seasonal succession and diversity of phytoplankton in a eutrophic lagoon (Liman lake), Journal of Environmental Biology 31:629-636.

Spatharis, S., D.L. Roelke, P.G. Dimitrakopoulos, and G.D. Kokkoris. 2011. Analyzing the (mis)behavior of Shannon index in eutrophication studies using field and simulated phytoplankton assemblages, Ecological Indicators 11:697-703.

Starkweather, P.L. 1981. Trophic relationships between the rotifer *Brachionus calyciflorus* and the blue-green alga *Anabaena flos-aquae*, Verh. Int. Ver. Limnol. 21:1507-1514.

Starkweather, P.L., J.J. Gilbert, T.M. Frost. 1979. Bacterial feeding by the rotifer *Brachionus calyciflorus*: clearance and ingestion rates, behavior and population dynamics, Oecologia 44:26-30.

Starkweather, P.L., and P.E. Kellar. 1983. Utilization of cyanobacteria by *Brachionus-calyciflorus – Anabaena flos-aquae* (NRC-44-1) as a sole or complementary food source, Hydrobiologia 104:373-377.

Stoermer, E.F. 1978. Phytoplankton assemblages as indicators of water quality in the Laurential Great Lakes, Transactions of the American Microscopical Society 97:2-16.

Turner Designs. 2004. *Aqua*fluor™ Handheld Fluorometer and Turbidimeter User's Manual. 13 May 2011 <http://www.turnerdesigns.com/t2/doc/manuals/aquafluor_manual.pdf>

Urabe, J. 1992. Midsummer succession of rotifer plankton in a shallow eutrophic pond, Journal of Plankton Research 14:851-866.

Vakkilainen et al. 2004. Response of zooplankton to nutrient enrichment and fish in shallow lakes: a pan-European mesocosm experiment, Freshwater Biology 49:1619-1632.

Veiga-Nascimento, R.A. 2004. *Water quality and zooplnakton community in a southern California lake receiving recycled water discharge*. M.S. Thesis. Univ. of California, Riverside, 61-87.

Walz, N. 1995. Rotifer populations in plankton communities – energetics and life-history strategies, Experientia 51:437-453.

Washington, H.G. 1984. Diversity, biotic and similarity indices. A review with special relevance to aquatic ecosystems, Water Research 18:653-694.

Wetzel, R.G. 2001. Limnology: Lake and River Ecosystems: Third Edition, Elsevier Academic Press.

Wetzel, R.G. and G.E. Likens. 1991. Limnological Analyses: Second Edition, Springer-Verlag. pgs. 208-212.

Wright, D. and J. Shapiro. 1990. Refuge availability: A key to understanding the summer disappearance of *Daphnia*, Freshwater Biology, 24, 43.

Zbikowski, J. and J. Kobak. 2007. Factors influencing taxonomic composition and abundance of macrozoobenthos in extralittoral zone of shallow eutrophic lakes, Hydrobiologia 584:145-155.

CHAPTER THREE

ALGAL COMMUNITY RESPONSE TO N, P, AND Si ADDITIONS TO LAKE ELSINORE WATER

INTRODUCTION

Lake Elsinore has had a history of summer cyanobacterial blooms that deteriorate water quality and may contribute to fish kills. A current study (chapter 2) and a previous study (Oza, 2003) have found that a winter to spring assemblage with high amounts of diatoms is replaced by a summer bloom assemblage that is essentially entirely comprised of the filamentous cyanobacterium *Pseudanabaena* (formerly *Oscillatoria*). These cyanobacterial blooms coincide with a drastic reduction in the diversity of the phytoplankton community. This means that there is a very limited choice of food for algivorous zooplankton, particularly since the phytoplankton present are almost entirely comprised of cumbersome filaments that are hard for large *Daphnia* species to handle (Demott, 1999; Gliwicz, 1990 and references therein; Sahuquillo et al., 2007). Consequently, only the few zooplankters (mainly rotifers and small cladocerans) that can survive the algal bloom (by either being able to feed on the filaments or selectively utilize some other food source) remain in the lake (see chapter two and Hanazato, 1996; Kirk & Gilbert, 1992; Ooms-Wilms, 1997; Pereira et al., 2002; Starkweather, 1981; Starkweather et al., 1979; Starkweather & Kellar, 1983; Walz, 1995 & references therein). Zooplankton richness drops considerably, the large cladocerns disappear (see chapter two), and the smaller zooplankters that remain may be less efficient at controlling algal biomass than large cladocerans due to the narrower size range of food particles they graze and higher threshold food levels (Gliwicz, 1990 and

references therein; Vakkilainen et al., 2004). There may be a certain threshold level of cyanobacteria above which large cladocerans are unable to control the phytoplankton population and algal blooms result. Because larger cladocerans are more vulnerable to physical interference or inhibition by filaments, this critical concentration appears to be significantly lower for larger *Daphnia* species than smaller ones (Gliwicz, 1990). Therefore, if management actions can be taken to reduce abundances of filamentous cyanobacteria and increase the relative abundances of other cells, such as diatoms and green algae that may be more edible and less inhibitory to zooplankters, it could theoretically be possible to maintain a zooplankton community capable of keeping the phytoplankton abundance in check. In this way, the zooplankton community could be recruited to help prevent the nuisance summertime algal blooms. This, in turn, could help improve water clarity, ameliorate hypolimnetic anoxia resulting from stratification and eutrophic conditions, and prevent fish kills.

In order for desirable algae to grow, they need the right amounts of certain required nutrients. Diatoms, for example, require Si (Wetzel, 2001) as a structural component in their intricate glassy frustules. When dissolved Si concentrations in the lake are not high enough for diatoms to form their frustules, the diatom community dies out and is replaced by another phytoplankton community (Wetzel, 2001). Often, before this happens, the diatoms will have used up large amounts of other available nutrients as they take advantage of the high nutrient levels brought in with runoff from winter and spring rains and cooler temperatures that are too low for green algae or cyanobacteria to thrive. The nutrients (Si, N, and P) taken up during diatom blooms will be removed from the photic zone (where they are available to other phytoplankton) as dead diatoms sink to the sediments. Therefore, the following algal community may have to be adapted to

low concentrations of one or more nutrients (though, unlike diatoms, they will not require Si). When N is the limiting nutrient, filamentous N-fixing cyanobacteria may be the favored successors of the diatoms (Wetzel, 2001). Similarly, when P is the limiting nutrient, those phytoplankton that are the best competitors for P (those with the lowest P requirement) will follow. It is advantageous to understand how phytoplankton communities shift in response to nutrient concentrations because this knowledge provides insight into why edible algae are replaced by filamentous cyanobacteria as well as the quantities and range of selection of food available to the zooplankton community. Furthermore, such knowledge could also be used to predict whether or not, under certain nutrient conditions and inputs into the lake, the zooplankton community will be able to have a significant grazing impact on the phytoplankton community and prevent algal blooms. A laboratory study was thus undertaken to evaluate the phytoplankton community response to available nutrient concentrations.

MATERIALS AND METHODS

Water was collected from Lake Elsinore on July 12, 2011 in a 20 L Nalgene bottle and returned to lab. The experiment was set up and started on July 18, 2011. The large bottle was mixed and 0.5 L ± 5 mL was added to ten separate plastic 2 L open-top containers to be prepared with 5 different treatments (each with a duplicate). Samples were not filtered to remove zooplankton because zooplankton abundances were low, large cladocerans were essentially absent from the lake, the copepods present were chiefly predatory, and the presence of native zooplankters kept experimental conditions somewhat more similar to those that may be expected in the

lake itself. The 5 treatments used for this experiment were: control (nothing added), Si added, Si + N added, Si + P added, and Si + N + P added. Nutrients were added as sodium salts. The Si was added as $Na_2O_3Si \cdot 9H_2O$, N as $NaNO_3$, and P as $NaH_2PO_4 \cdot H_2O$. Stock solutions of these salts were prepared so that, upon addition of 0.5 mL of the appropriate salts to the appropriate treatments, the concentrations of added Si, N, and P in the treatment containers would be approximately 3 mg Si L^{-1}, 3 mg N L^{-1}, and 0.3 mg P L^{-1}. These concentrations were chosen to mimic those that might be present in regular runoff or recycled water inputs into the lake. After the salt solutions were added, the treatment solutions were mixed and placed into a growth chamber to be incubated approximately 7 days at 25°C, 32 µE m^{-2} s^{-1}, and 95% humidity (to prevent excess evaporation and concentration of the sample water). Samples were taken from the original bulk solution for initial Si and P concentrations, and initial phytoplankton abundances and chlorophyll florescence were measured. The phytoplankton samples were preserved using approximately 1.5 mL Lugol's solution per 60 mL sample and stored in the refrigerator until analysis. The samples for Si, N, and P were preserved with the addition of strong acid and refrigerated until analysis. The samples incubating in the growth chamber were stirred twice daily to prevent the sinking of phytoplankton to the bottom of the container. Chlorophyll fluorescence measurements were taken for each replicate approximately once every 24 hours with an *Aqua*Fluor™ fluorometer. At the end of the incubation (7 days), samples were taken for the final phytoplankton community for each replicate in the same manner as for the initial samples. Final chlorophyll measurements were also taken.

To assess the abundances and composition of the phytoplankton assemblages in the initial lake water and treatments at the end of the incubation, 5 mL of each sample

(two for the initial and two duplicates per treatment) were settled in Utermohl settling chambers (Clesceri et al., 1998; PhycoTech, 2011) for at least five hours (usually overnight). Once fully settled, 0.079 mL of the concentrated sample in the chamber base was pipetted into a *PhycoTech* nanoplankton chamber (PhycoTech, 2011) that was individually calibrated to hold exactly 0.079 mL and overlain with a coverslip. Once the sample had settled in the chamber, random fields were counted at 400 x magnification on a Nikon E600 compound microscope and every cell, filament, or colony tallied in each field. At least 6 fields were counted from every part of the chamber (top, bottom, left, right, and center areas) to ensure that variation in the way the phytoplankton may have settled onto the bottom of the chamber was accounted for. Given the large size and relative scarcity of the diatoms (with the exception of the small *Chaetoceros*), a separate scan of the entire chamber was done at 100 x magnification to ensure that diatoms were not missed in counts and get a better idea of their abundance. *Synedra* and large disk diatoms were the main types tallied in these counts. For *Chaetoceros*, abundances, results for the random field counts were used. This genus was almost entirely present in the initial solution that was used for the treatments. Once the counting was complete, totals were calculated by combining the results for each field from the count of a treatment and abundances were calculated using the formula

$$C=\frac{CAxTCxBV}{FxFAxCVxSV}$$

where C is the total abundance (cells mL^{-1}) CA it the total area of the counting chamber (mm^2), TC is the total count, F is the total number of fields counted, FA is the area of a field of view at the magnification used (mm^2), CV is the volume of the counting chamber (mL), BV is the volume of the base plate of the settling chamber (mL) and SV is the volume of sample that was settled (mL) (Standard Methods, APHA 10200F).

To simplify results for an easier assessment of the main types of phytoplankton of concern for this particular experiment (e.g. filamentous cyanobacteria versus larger diatoms), results were combined into a smaller number of categories than types of phytoplankton found. These categories were: filaments of the cyanobacterium *Pseudanabaena limnetica*, filaments of the cyanobacterium *Cylindrospermopsis raciborskii*, coiled filaments of the cyanobacterium *Planktolyngbya tallingii*, total filamentous cyanobacteria (filament abundance), *Syndera* sp. (total cell abundance), total diatoms (total cell abundance), and other (total abundance of colonial cyanobacteria and green algae and individual green algae and cryptophyte cells). Given the limits to magnification and resolution on the microscope used, only cells, filaments, and colonies that were clearly visible while counting (those larger than about 10 μm) were counted and placed into one of these groups. Simpson's diversity was calculated to assess the diversities of various groups of phytoplankton (cyanobacteria, green algae, and the total phytoplankton community, see chapter two materials and methods for the equation used). For these diversity calculations, filamentous cyanobacteria were identified to species, and other types of algae were identified to genus level. Results are, therefore, not completely comparable with those presented for the phytoplankton communities in chapter 2 for which species level identifications were used for almost all the phytoplankters (morphs for *C. raciborskii*). ANOVAs and Tukey's Method were used to find significant differences between treatments at $\alpha=0.05$.

RESULTS

Initial total dissolved silicon and phosphorus concentrations in the water taken from Lake Elsinore were 1.99 mg L^{-1} and 0.03 mg L^{-1} respectively. These Si concentrations are well above the 0.23 mg L^{-1} below which Wetzel (1983) suggests that low dissolved Si concentrations could adversely affect diatom populations. A July 2011 draft for the Lake Elsinore and Canyon Lake Nutrient TMDL Annual Water Quality Monitoring Report shows TN and TP concentrations in the water column in June 2011 to be approximately 4.7 mg TN L^{-1} and 0.5 mg TP L^{-1} (Santa Ana Regional Water Quality Control Board, 2011). This gives a TN:TP ratio of 9.4 for July 2011, which is just below the range where N and P may be co-limiting. According to Sakamoto (1966), N may be limiting for a TN:TP ratio <10 (by weight) and P may be limiting for a TN:TP ratio >17 (by weight). The initial chlorophyll concentration was 67.4 µg L^{-1} (Fig. 3.1) and the water was noticeably green. Chlorophyll concentrations decreased for all treatments within the first 24 hours after the start of incubation, perhaps due to a programming error that resulted in the growth chamber unexpectedly returning to a much higher (50% versus 8%) irradiance. Once the growth chamber was reprogrammed for 8% irradiance, chlorophyll began increasing for the treatments with N added and then for the Si + P treatment, but remained low for the treatments without N or P (i.e. control and Si addition) (Fig. 3.1). Where chlorophyll increased for the treatments with added nutrients, it began to decrease after reaching a maximum, but increased greatly during the last two days of incubation for the Si + N + P treatment to levels roughly double that of initial concentrations (Fig. 3.1). During the last couple days, the growth chamber irradiance did fluctuate to 28% to 40% for brief periods in the morning (for all samples), but did not

do so for most of the experiment. Aside from this treatment, for which chlorophyll levels were frequently higher than initial levels, chlorophyll concentrations for the remaining treatments were always lower than initial levels (Fig. 3.1). Despite the much higher chlorophyll fluorescence for the Si + N + P treatment, the water was notably clearer, no greener than for the treatments with low final chlorophyll, while the treatment with the second highest final chlorophyll fluorescence (Si + N) was considerably greener than the rest, suggesting differences in community composition.

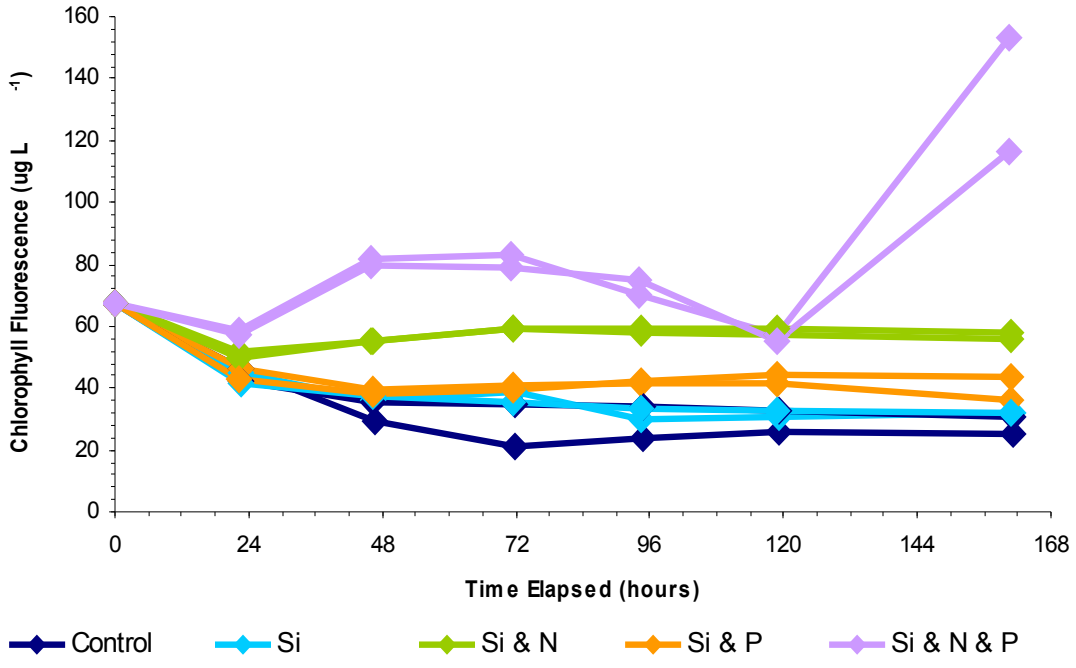

Fig. 3.1. Chlorophyll fluorescence of treatment duplicates during the incubation period

The initial phytoplankton assemblage in the water used for the treatments was dominated by filamentous cyanobacteria, mainly *P. limnetica* with smaller amounts of *P. tallingii* and *C. raciborskii*, with average filament abundance for the two initial samples

approximately 184,000 ± 4,600 filaments mL^{-1} (Figs. 3.2 and 3.4). Diatoms were also present in good numbers (averaging 1430 cells mL^{-1}) (Fig. 3.3), but the vast majority (about 94%) of this total diatom abundance was comprised of the relatively tiny members of the marine genus *Chaetocerus* with only about 6% (89 cells mL^{-1}) of the total diatom abundance consisting of large *Synedra* cells. Centric diatoms were also noted at small abundances (25 cells mL^{-1}) in one of the initial samples.

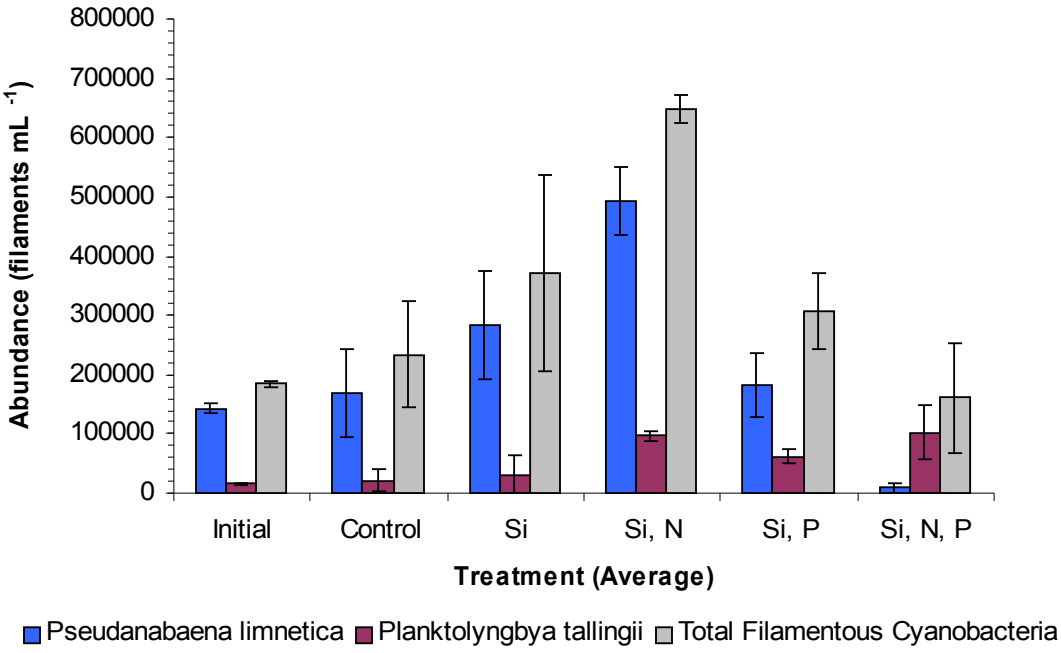

Fig. 3.2. Final average abundances of *P. limnetica*, *P. tallingii*, and total filamentous cyanobacteria (error bars represent one standard deviation)

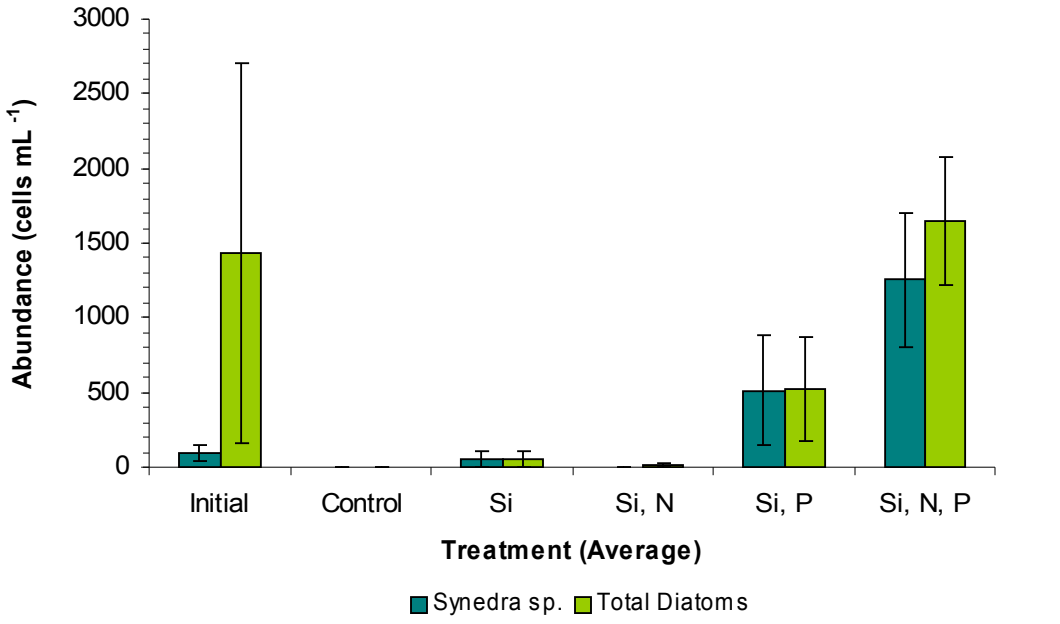

Fig. 3.3. Final average abundances of *Synedra* and total diatoms (error bars represent one standard deviation)

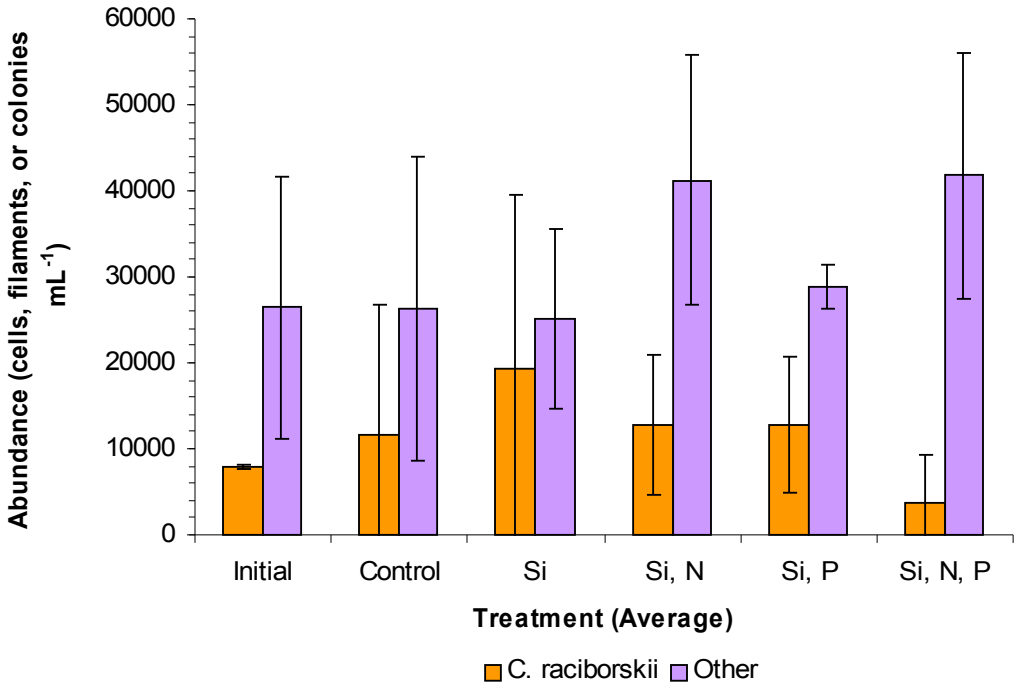

Fig. 3.4. Final average abundances of *C. raciborskii* and "other" algae that are not filamentous cyanobacteria or diatoms (mainly green algae) (error bars represent one standard deviation)

The process of incubation in the growth chamber had a couple effects that are apparent when comparing the initial phytoplankton community in the lake water with the final control community (Figs. 3.2, 3.3, and 3.4). Perhaps the most obvious difference is the disappearance of the somewhat abundant diatom community (which was almost entirely comprised of the genus *Chaetoceros* in the initial community) during incubation (Fig. 3.3). Less obvious is that the incubation process (with two separate duplicates) increased the variability in the phytoplankton community (note the larger standard deviations for most of the control populations relative to the initial populations, Figs. 3.2,

3.3, and 3.4). Other treatments could be expected to respond to the incubation process in a similar way to the control.

Final average compositions and abundances of the control and Si treatments were similar to those of the initial samples (Figs. 3.1, 3.2, and 3.3) with the phytoplankton community dominated by filamentous cyanobacteria (mainly *P. limnetica*) at moderate abundances of between 200,000 and 400,000 filaments mL^{-1} and few large diatoms. All the diatoms found for the Si treatment were *Synedra* (Fig. 3.3). No diatoms were found in the control treatment (Fig. 3.3). *Chaetoceros* was not found in the final assemblages of these or any other treatments.

Abundances and relative abundances of filamentous cyanobacteria were also similar to those in the initial lake water for the Si + P treatment. Total filament abundances for this treatment averaged 306,000 filaments mL^{-1} (Fig. 3.2), which is between the averages for the control and Si treatments. There appears to be a slight difference in the phytoplankton community in that *P. limnetica* was somewhat less prominent and *P. tallingii* more prominent in the final assemblages for this treatment than those for the initial, control, or Si treatments (Fig. 3.2). However the differences between the relative proportions of *P. limnetica* and *P. tallingii* with respect to the total abundances of filamentous cyanobacteria were not found to be statistically significant for these treatments. In contrast, a rather notable difference between the Si + P treatment and the previously mentioned treatments is that large diatoms, particularly *Synedra*, were considerably more abundant in samples of this treatment than in initial, control, or Si treatment samples with total diatom abundances averaging 526 cells mL^{-1} (Fig. 3.3).

Samples where both Si and N were added had final communities that were notably different than those of initial and control samples and the addition of P for one of

these treatments changed the final community completely. Differences between final phytoplankton assemblages in these two treatments versus the other treatments were statistically significant (see the end of this section for further explanation). The addition of N without P (Si + N treatment) resulted in a large increase in the abundances of filamentous cyanobacteria relative to control and initial samples. As with the the Si + P treatment, *P. limnetica* was somewhat of a less prominent component and *P. tallingii* a more prominent component of the filamentous cyanobacteria than in initial and control samples (Fig. 3.2), but this difference was not significant. Average filament abundances for the Si + N treatment were 493,000 ± 24,800 filaments mL^{-1}. Diatom abundances averaged only 16 cells mL^{-1} (approximately one third the final diatom abundance for the Si treatment) (Fig. 3.3). Only centric diatoms were found in final samples for this treatment.

While filamentous cyanobacteria became more abundant and large diatoms were scarce with *Synedra* apparently absent in the Si + N treatment samples, the opposite was true where P was added in addition to N and Si (i.e., Si + N + P treatment). *P. tallingii* abundances were higher in both treatments where N was added than in any other treatment (Fig. 3.2), but the abundances of *P. limnetica*, and, consequently, of total filamentous cyanobacteria, plummeted during incubation for both duplicates of the Si + N + P treatment (Fig. 3.2). This brought average total filament abundances much nearer to those of the control (162,000± 93,000 filaments mL^{-1}) and changed the composition of the filamentous cyanobacteria from being dominated by *P. limnetica* to being comprised almost entirely of the small coils of *P. tallingii*. Large diatom abundances, *Synedra* in particular, increased considerably so that abundances of large diatoms were notably higher than those for any other treatment. Only the *Chaetoceros*

abundances of the initial samples were similar to total diatom abundances found for the Si + N + P treatment (Fig. 3.3). Total diatom abundances averaged 1,646 ± 426 cells mL^{-1} and were dominated by *Synedra* (Fig. 3.3). Centric diatoms were also a large component, with an average abundance of approximately 378 cells mL^{-1}. There are a couple of additional observations that may be of interest. First of all, no mature *C. raciborskii* filaments were found in the final sample for one of the Si + N + P duplicates, yet they were found in every other sample from every other treatment. Also, a few *Nitzschia acicularis* were found for this treatment, though at much lower abundances than *Synedra* or centric diatoms. For all the treatments, where differences were found relative to initial and control samples, they were primarily due to shifts in the abundances and relative abundances of *P. limnetica* and diatoms (especially *Synedra*) (Figs. 3.2 and 3.3).

Results from ANOVAs and Tukey's Method (Table 3.1) show that there were some significant differences in the compositions of some of the treatments. The Si + N + P treatment had significantly different abundances of the diatom *Synedra* when compared to the control, Si, and Si + N treatments. This treatment also had significantly different abundances of *P. limnetica* when compared to the Si and Si + N treatments and was significantly different from every other treatment in terms of total diatom abundance. Furthermore, the Si + N + P treatment is the only treatment to differ significantly from all other treatments in the relative proportions of *P. limnetica* and *P. tallingii* with respect to total abundances of filamentous cyanobacteria. The Si + N treatment had significantly different *P. limnetica* abundances when compared to the control, Si + P, and Si + N + P treatments and differed significantly from the control and Si + N + P treatements in its average abundance of filamentous cyanobacteria. These results support those shown in

Figures 3.2 and 3.3 suggesting that the addition of N alone causes a shift in the phytoplankton community toward increased abundance of and dominance by filamentous cyanobacteria (particularly *P. limnetica*) while the addition of P stimulates large diatoms (*Synedra* in particular) to abundances greater than those achievable without its addition.

Table 3.1. Summary of ANOVA and Tukey's test results comparing final abundances. Treatments with different letters have significantly different abundances at α=0.05. Treatments with the same letter are not significantly different at α=0.05.

	Treatment				
Phytoplankton Group	**Control**	**Si**	**Si + N**	**Si + P**	**Si + N + P**
Psuedanabaena limnetica	a	b	b	a	a
Total Filamentous Cyanobacteria	b	a	a	a	b
Synedra	a	a	a	b	b
Total Diatoms	a	a	a	a	b
Relative proportions of *P. limnetica* and *P. tallingii* with respect to total filamentous cyanobacteria	a	a	a	a	b

Other than the differences mentioned above, the addition of nutrients was not found to have a significant effect on the other components of the phytoplankton community (i.e. *C. raciborskii* and phytoplankters that are neither filamentous cyanobacteria nor diatoms). Of the other phytoplankters found, only one genus of green algae, *Chlamydomonas*, showed a significant change in abundances with nutrient additions. Abundances increased with increasing nutrient additions (response to Si + P greater than that for Si + N) so that there was a significant difference between final abundances in the Si + N + P treatment and those of the control (not shown).

Nutrient addition also did not have any significant affects on the diversity of the phytoplankton community, though the process of incubation apparently increased green algae diversity (Fig. 3.5). The green algae communities tended to be more diverse than the cyanobacteria communities (Fig. 3.5), but, since green algae had such low abundances relative to cyanobacteria (Figs. 3.2 and 3.4 (green algae are the main component of the "other" group)), the diversities of the entire phytoplankton communities closely reflected those of the cyanobacteria communities and were generally not significantly altered as a result of nutrient additions (Fig. 3.5).

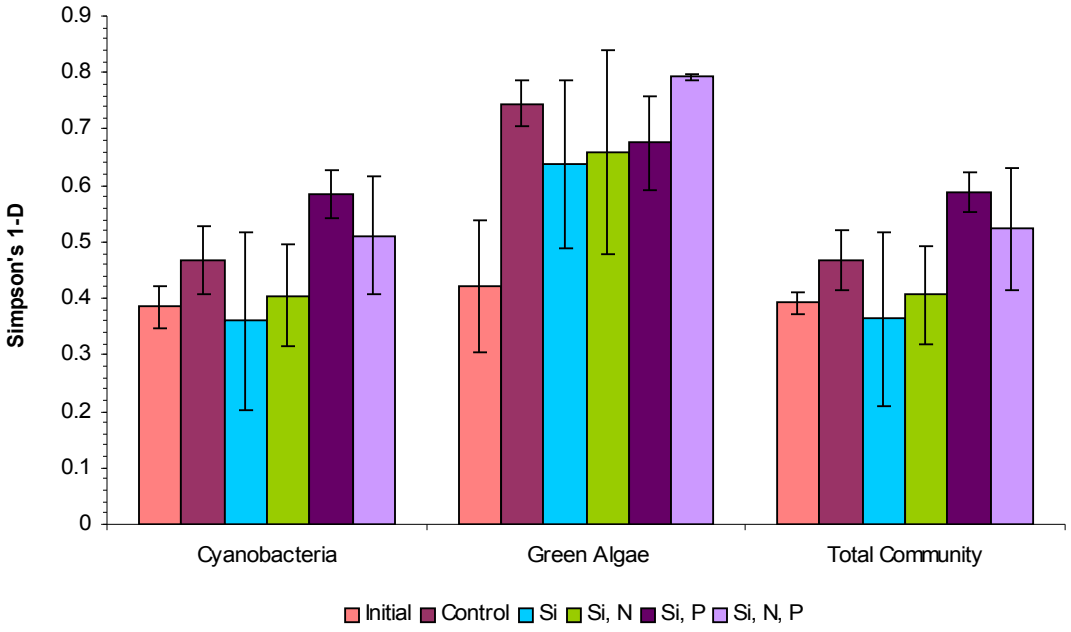

Fig. 3.5. Final diversities of the cyanobacteria community, the green algae community, and the total phytoplankton community as expressed as the Simpson's 1-D (error bars represent one standard deviation)

DISCUSSION AND CONCLUSIONS

Based on the results to this study, both chlorophyll fluorescence measurements and final abundances for treatment duplicates, it appears that phytoplankton under laboratory conditions with an optimal light regime are generally N-limited. The addition of N to treatments resulted in the highest chlorophyll fluorescence and the greatest abundances of filamentous cyanobacteria or large diatoms, depending on the treatement. *P. limnetica* seems to be the best competitor when nutrients, especially P, are low as it attained the highest final abundances when no P was added and became somewhat less dominant (lower relative abundance) when either N or, especially, P was added. Even though these differences are not significant, note the increased difference between the final *P. limnetica* and total filamentous cyanobacteria abundances and/or *P. tallingii* abundances for the Si + N and Si + P treatments compared to other treatments and the scarcity of *P. limnetica* for the Si + N + P treatment (Fig. 3.2). With low P concentrations (no added P), *P. limnetica* was a superior competitor for added N, overwhelming the rest of the phytoplankton community to reach high abundances within the range of those observed during the 2010 summer bloom, particularly those in August (see chapter 2). Large diatoms, on the other hand, appear to be largely stimulated by the addition of Si and, particularly, P. *Synedra* in particular, only reached considerable concentrations when both Si and P were added, apparently suffering during the *P. limnetica* bloom when Si and N were added without P and thriving (at least relatively) when all three nutrients (Si, N, and P) were added and it could compete with the cyanobacteria for the added N. This competition with the increasing diatom population for the available N appears to be what limited *P. limnetica* growth and maintained water clarity while increasing chlorophyll fluorescence so effectively for the Si + N + P

treatment, though other types of phytoplankton that were less of a focus in this experiment likely played a role as well.

Results from this study are consistent with what may have been expected based upon the initial concentrations of P and Si. The initial concentration of total dissolved P (0.03 mg L^{-1}) was somewhat low so it is not surprising that it seemed to limit diatom production in the experiment. On the other hand, initial Si concentrations (1.99 mg L^{-1}) were relatively high and well above the 0.23 mg L^{-1} below which Wetzel (1983) suggests that low dissolved Si concentrations could adversely affect diatom populations, contributing to the dominance and increased abundance of cyanobacteria. The fact that there was no significant increase in diatom abundance relative to the control when Si was added without P is a further indication that the diatoms in Lake Elsinore (at least for the summer of 2011) are limited first and foremost by P and secondarily by Si and N.

These results are especially interesting when placed in the light of what has been observed in Lake Elsinore and another shallow eutrophic lake (Lake Okeechobee in Florida) with a similar seasonal succession and summertime phytoplankton composition. As described in chapter two of this document, the February 2010 phytoplankton assemblage of Lake Elsinore was mostly diatoms, with diatoms continuing to comprise a considerable portion of the phytoplankton community through April. Beginning in June, however, the phytoplankton was comprised nearly entirely of cyanobacteria (especially *P. limnetica*) and this dominance by *P. limnetica*, while greatest in June through August remained, to lesser degrees, through October and December 2010. While diatom abundances decreased and filamentous cyanobacteria (mainly *P. limnetica*) abundances increased from February to June 2010, average monthly P concentrations were decreasing and N concentrations increasing throughout the lake over this same time

frame (Santa Ana Regional Water Quality Control Board, 2011). Thus, although this experiment focused primarily on abundances and trends from chapter two are based on biomass, results from this experiment using a 2011 summertime phytoplankton assemblage from Lake Elsinore fit logically with trends observed during the first half of 2010. As the TN:TP ratio increases, diatom biomass (or abundances) can be expected to decrease while *P. limnetica* biomass (or abundances) increases. Results from the July 2011 draft for the Lake Elsinore and Canyon Lake Nutrient TMDL Annual Water Quality Monitoring Report and from the data presented in chapter two were used to create a plot of TN:TP versus diatom and *P. limnetica* biomass that shows these trends (Fig. 3.6). It should be noted, however, that trends for TN:TP versus biomass may not correlate well with trends for TN:TP versus abundances if the size of the dominant diatom changes considerably from one sampling to the next. This happened between February and April of 2010 in Lake Elsinore. The dominant diatom in February, *Skeletonema*, is considerably larger than the dominant diatom in April, *Cyclotella*, so that, while February diatom biomass was about twice as in April, abundances in February were actually lower. For this reason, biomass is the more appropriate variable to compare with the TN:TP ratio unless the species composition and average size of individuals do not change significantly.

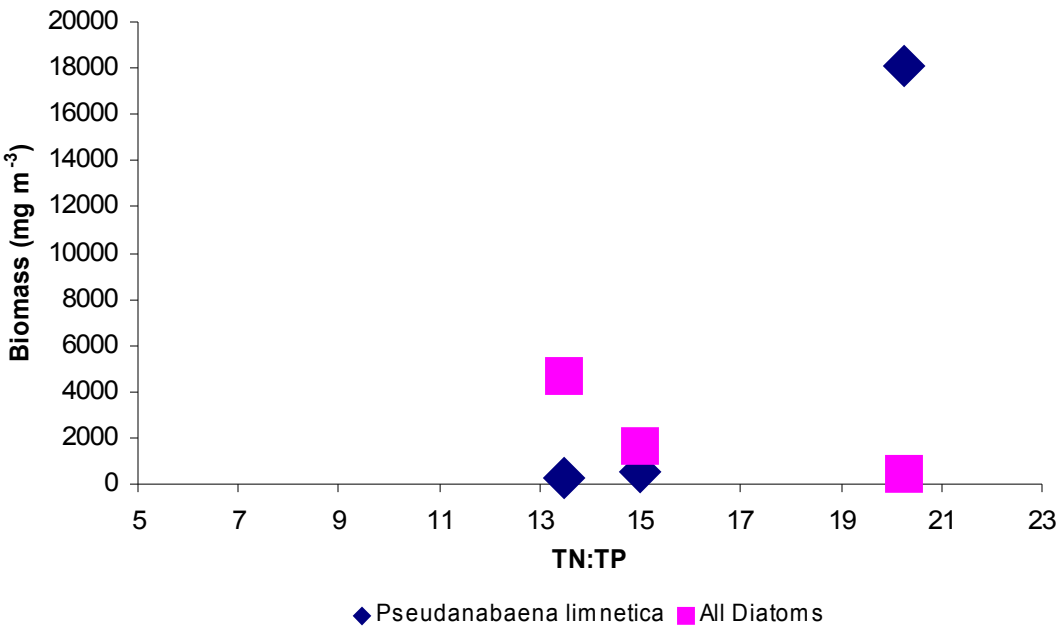

Fig. 3.6. TN:TP ratio versus average *P. limnetica* and diatom biomass for Lake Elsinore in February, April, and June 2010

The response by *P. limnetica* is one that may be expected in a lake where N is a limiting nutrient for a species that does not fix its own N using atmospheric N_2. Similar results were also observed by McCarthy et al. (2009) during a 10-year study of Lake Okeechobee. During this Florida study, there was a diatom maximum from March through May followed by a rapid increase in the proportion of cyanobacteria in June and a continuing dominance of cyanobacteria through October. *P. limnetica* was one of the dominant cyanobacteria species. Also similar to what was observed for Lake Elsinore, a higher TN:TP ratio in summer corresponded to a high cyanobacteria proportion, contrary to what would be expected for known N-fixing cyanobacteria such as *C. raciborskii*. McCarthy et al. (2009) also noted that there was a large increase in the NH_4^+:NO_X ratio

prior to the initial surge in the cyanobacteria proportion from May to June, which may have resulted because of the diatoms' affinity for NO_3^- and its depletion during the diatom max. NO_3^- was the form of N added during this nutrient addition experiment for Lake Elsinore. With these observations considered, it seems that, although N appears to be a limiting nutrient in Lake Elsinore, it is the relatively high concentrations of N relative to P that contribute to the observed summer *P. limnetica* blooms in the lake because *P. limnetica* is a superior competitor at low nutrient levels, particularly low P. Once the TN:TP ratio is reduced and the diatoms have ample phosphorus (and perhaps ample NO_3^- relative to NH_4^+), they appear to be better able to compete with *P. limnetica* for available nutrients and grow enough to keep the population of this filamentous cyanobacterium in check.

Another similarity between this and previous studies at other lakes is the considerable increase in *Synedra* abundances with the addition of nutrients to the lake water. Studies have been done of diatoms preserved in the sediments of several different lakes that have experienced accelerated anthropogenic nutrient loading. Results of these studies indicate that the early stages of eutrophication are often characterized by increased abundances of spindle-shaped araphidinate diatom species (including *Synedra*) (Brugam 1978; Smol and Dickman 1981; Smol et al. 1983; Stockner and Benson 1967; Zeeb et al., 1994). Therefore, results of this study are consistent with what may be expected after the experimental addition of nutrients to lake water.

Considering the above, the results of this study suggest that phosphorus inputs in runoff stimulate the diatoms in Lake Elsinore when dissolved Si concentrations are high enough and that it is the high inputs and lake-wide concentrations of N relative to P that drive the formation of the summer *P. limnetica* bloom. Results also suggest that, at

least during *P. limnetica* blooms, nutrient loading does not have an appreciable effect on algae other than diatoms and a couple species of filamentous cyanobacteria (*P. limnetica* and *P.tallingii*) and that it does not significantly affect the diversity of the phytoplankton community. Since green algae were found to be more diverse than cyanobacteria in the study and diversity values during the 2010 in-lake study were only high at times when green algae and/or diatoms were at relatively high biomasses (see chapter 2), diversity will probably not change considerably unless the effects of nutrient loading and other factors combine to break the dominance of the filamentous cyanobacteria. This was not found to occur for the conditions of this experiment. Therefore, it seems unlikely that nutrient manipulations and control will have an appreciable effect towards increasing the abundance and variety of favorable food items for zooplankton once a cyanobacterial bloom begins.

It should be noted that, results from this study only show what has happened under a specific set of conditions with a specific phytoplankton community and may not be indicative of what will definitely happen should Si, N, and P concentrations increase in the lake due, for example, to increased recycled water additions. For example, increased P in the water column, without an increase in N concentrations, decreases the TN:TN ratio and may favor the growth of bloom-forming, N-fixing cyanobacteria as N becomes more limiting relative to P (Koski-Vahala et al., 2001). Though the potentially harmful N-fixing cyanobacterium *C. raciborskii* does not appear to have been notably stimulated in this particular study, it could be stimulated with P loading under a different set of conditions. Furthermore, green algae and other genera of diatoms besides *Syndera* may be stimulated by nutrient loading under differing conditions, such as lower temperatures, which may be less favorable to *P.* limnetica. Nonetheless, results from

this study do provide new insight into factors affecting the summer phytoplankton community in Lake Elsinore that could be useful in planning future actions to reduce the intensity of cyanobacterial blooms through careful management of nutrient inputs.

REFERENCES

Brugam, R.B. 1978. Human disturbance and historical development of Linsley Pond, Ecology 59:19-36.

Clesceri, L.S., A.E. Greenberg, and A.D. Eaton (Eds.). 1998. Standard Methods for the Examination of Water and Wastewater, 20th Edition, American Public Health Association, pg. 10-9.

DeMott, W.R. 1999. Foraging strategies and growth inhibition in five daphnids feeding on mixtures of a toxic cyanobacterium and a green algae, Freshwater Biology 42:263-274.

Gliwicz, Z.M. 1990. Why do cladocerans fail to control algal blooms, Hydrobiologia 200:83-97.

Hanazato, T. 1996. Toxic cyanobacteria and the zooplankton community, from Toxic Microcystis, Chapter 5, editors Watanabe, M.F., K. Harada, W.W. Carmichael, and H. Fujiki, pgs. 79-98.

Kirk, K.L., and J.J. Gilbert. 1992. Variation in herbivore response to chemical defenses – zooplankton foraging on toxic cyanobacteria, Ecology 73:2208-2217.

McCarthy, M.J. et al. 2009. Nutrient ratios and phytoplankton community structure in the large, shallow, eutrophic, subtropical Lakes Okeechobee (Florida, USA) and Taihu (China), Limnology 10:215-227.

Ooms-Wilms, A.L. 1997. Are bacteria an important food source for rotifers in eutrophic lakes?, Journal of Plankton Research 19:1124-1141.

Pereira, R., A.M.V.M. Soares, R. Ribeiro, F. Goncalves. 2002. Assessing the trophic state of Linhos lake: a first step towards ecological rehabilitation, Journal of Environmental Management 64:285-297.

PhycoTech – Raising the Standard in Aquatic Analysis. 2011. 18 August 2011 <http://www.phycotech.com/index.html>

Sahuquillo, M., M.G.G. Melao, M.R. Miracle. 2007. Low filtering rates of *Daphnia magna* in a hypereutrophic lake: Laboratory and in situ experiments using synthetic microspheres, Hydrobiologia 594:141-152.

Sakamoto, M. 1966. Primary production by phytoplankton community in some Japanese lakes and its dependence on lake depth, Arch. Hydrobiol. 62:1-28.

Santa Ana Regional Water Quality Control Board. 2011. *Lake Elsinore and Canyon Lake Nutrient TMDL Annual Water Quality Report: June 2011*. Draft Report for the Santa Ana Regional Water Quality Board. pg. 3-6.

Smol, J.P. and M.D. Dickman. 1981. The recent histories of 3 Canadian shield lakes – A paleolimnological experiment, Archiv Fur Hydrobiologie 93:83-108.

Smol, J.P., S.R. Brown, and R.N. McNeely. 1983. Cultural disturbances and trophic history of a small meromictic lake from central Canada, Hydrobiologia 103:125-130.

Standard Methods for the Examination of Water and Wastewater. Twentieth Edition. American Public Health Association, 1998.

Starkweather, P.L. 1981. Trophic relationships between the rotifer *Brachionus calyciflorus* and the blue-green alga *Anabaena flos-aquae*, Verh. Int. Ver. Limnol. 21:1507-1514.

Starkweather, P.L., J.J. Gilbert, T.M. Frost. 1979. Bacterial feeding by the rotifer *Brachionus calyciflorus*: clearance and ingestion rates, behavior and population dynamics, Oecologia 44:26-30.

Starkweather, P.L., and P.E. Kellar. 1983. Utilization of cyanobacteria by *Brachionus-calyciflorus – Anabaena flos-aquae* (NRC-44-1) as a sole or complementary food source, Hydrobiologia 104:373-377.

Stockner, J.G., and W.W. Benson. 1967. The succession of diatom assemblages in the recent sediments of Lake Washington, Limnol. Oceanogr. 5:195-211.

Vakkilainen et al. 2004. Response of zooplankton to nutrient enrichment and fish in shallow lakes: a pan-European mesocosm experiment, Freshwater Biology 49:1619-1632.

Walz, N. 1995. Rotifer populations in plankton communities – energetics and life-history strategies, Experientia 51:437-453.

Wetzel, R. 1983. Limnology, 2nd ed. Philadelphia PA: Saunders College Publishing. pgs. 373-375.

Wetzel, R.G. 2001. Limnology: Lake and River Ecosystems: Third Edition, Elsevier Academic Press.

Zeeb, B.A. et al. 1994. Responses of diatom and chrysophyte assemblages in Lake-227 sediments to experimental eutrophication, Canadian Journal of Fisheries and Aquatic Sciences 51:2300-2311.

CHAPTER FOUR
SUMMARY & CONCLUSIONS

Traditional monitoring of physical and chemical aspects of a water body ignore the organisms that are influenced by them and that could, potentially, influence them in return. Aquatic organisms can, however, significantly affect water quality though their influences on the composition of lake water. Nutrients and contaminants may be taken into and stored in the food web as they are taken into organism's bodies and the length of storage in biological material and extent of recycling back into the water column through excretion or death and decomposition influence concentrations of these chemicals. Furthermore, organisms may put nutrients or toxins into an aquatic system. A good example of this is cyanobacteria, which may fix atmospheric N_2 gas to NH_4^+, which is then available to other organisms in the environment and they may product toxins that could harm humans and animals. Through photosynthesis and respiration, plants and animals affect the concentrations of dissolved oxygen in the water column and sediments. This alters the redox state of the system and the form different elements are in. Therefore, monitoring of aquatic communities provides significant information about a body of water that could assist managers in improving or maintaining water quality. For this reason, biological monitoring is particularly important for lakes that are widely used by humans and animals.

One such lake that is utilized as both a wildlife habitat and recreation site is Lake Elsinore. Lake Elsinore is a shallow, polymictic lake located in the southwest corner of Riverside County in Southern California (33° 39.5' latitude and 117° 21.0' longitude). The lake has historically been plagued by poor water quality, including algal blooms, low dissolved oxygen (DO) concentrations, and periodic fish kills. As a result, Lake Elsinore

has been listed on the State of California's 303d list, and is considered "impaired" due to nutrients and other factors. The Santa Ana Regional Water Quality Control Board (RWQCB) subsequently developed a nutrient TMDL for the lake. A number of restoration activities have been implemented in the lake to improve water quality, including carp removal, stocking of hybrid striped bass, and installation of axial flow pumps and diffused aeration systems. While water column measurements and nutrient concentrations have been closely monitored since 2000, comparatively less is known about the biological condition of the lake. As a result, a 1-yr intensive biological monitoring study was conducted to quantify the abundance, diversity, and richness of phytoplankton, zooplankton and benthic invertebrates in Lake Elsinore. This thesis summarizes results from this research.

The phytoplankton community in Lake Elsinore varied strongly over time. Total phytoplankton biomass increased from about 5,000 mg m^{-3} in the winter to over 20,000 mg m^{-3} in August 2010, before declining sharply in October. The community was dominated by diatoms in the winter, comprising about 80% of the biomass in February 2010, with much smaller numbers of blue-green algae (cyanophytes), euglenoids and green algae (chlorophytes). The winter community was also found to have high species richness (n=36) and high diversity (0.79). Total phytoplankton biomass was similar in April 2010, although the composition of the community shifted to one with lower species richness (n=29) and somewhat lower diversity (0.71). Diatom abundance decreased, while blue-green algae, green algae and euglenoid biomasses all increased. The blue-green algae had already become the dominant group by April, comprising about 45% of the total algal biomass. Blue-green algal biomass increased dramatically in June 2010, to almost 20,000 mg m^{-3}, with <1,000 mg m^{-3} (<5%) of the total phytoplankton biomass in

other groups. Species richness (n=12) and diversity (0.2) both dropped sharply as well. Similar results were found in the August sampling, although diversity did edge up slightly (0.34). The blue-green algae were comprised principally of *Pseudanabaena limnetica* (formerly *Oscillatoria limnetica*), which accounted for 80-90% of the total phytoplankton biomass in the summer. The results are thus similar to observations made in the lake by Oza (2003). Total phytoplankton biomass (especially that of blue-green algae) decreased strongly in the fall. Although species richness remained comparatively low, diversity within the community did increase, returning to about 0.8 by December 2010.

Strong changes in the zooplankton community were also observed through the year. Total zooplankton abundance ranged from quite low values in June and December 2010 (approximately 60 individuals L^{-1}) to about 200 individuals L^{-1} in April and 400 individuals L^{-1} in October. In general terms, there appeared to be an increase in the overall density of the zooplankton community through the winter and spring, a strong decline in the summer when temperatures were high and very high levels of *Pseudanabaena limnetica* were present, and a large but short-lived increase in zooplankton (principally *Brachionus*, a rotifer) in the fall. As observed by Veiga-Nascimento (2004) under broadly similar conditions, cladocerans were absent through much of the year, although small numbers of *Daphnia lumnholtzi* and other *Daphnia* species were found in the spring, perhaps delivered with the winter-spring rains and runoff into the lake (*D. lumholtzi* has previously been found in Canyon Lake in moderate abundances). Warm temperatures, poor food quality, possible salinity impairments, and predation by threadfin shad are thought to have constrained development of a large *Daphnia* population in Lake Elsinore.

Benthic invertebrates in the lake were restricted to chironomids (commonly referred to as bloodworms), *Chaoborus* (midge larvae), and *Ilyocryptus* (a benthic cladoceran). Densities of these benthic invertebrates varied strongly across the 4 sampling sites as well as over time. The shallower sites with higher DO and lower organic C concentrations in the sediments had consistently higher numbers of organisms than the deeper sampling sites. The chironomids as well as the benthic cladoceran all possessed a significant amount of hemoglobin, allowing for binding and transfer of oxygen; this reflects the low DO concentrations near and within the bottom sediments in the lake. The very low abundances of benthic invertebrates over much of the lake bottom due to anoxic conditions through much of the year suggest that they would constitute a relatively minor part of the diet of the fish in the lake, resulting in greater grazing pressure on the zooplankton community (especially large-bodied *Daphnia* and copepods). This would be consistent with the generally small size of the zooplankton, since threadfin shad and other zooplanktivorous fish selectively graze upon larger zooplankton.

The ecology and food web in Lake Elsinore is thus consistent with a eutrophic lake with a phytoplankton community with low species richness, low diversity and strong blue-green algae production (especially *Pseudanabaena limnetica*) during much of the year. At the same time, intense grazing pressure by threadfin shad and other effects (e.g., low DO, poor food quality) constrains the development of a beneficial zooplankton community, and thus limits grazing on phytoplankton and natural control of algal levels in the lake. Low dissolved oxygen levels may also be limiting the capacity for benthic invertebrate production in the lake.

Results from the nutrient addition experiment indicate that the phytoplankton overall are N-limited (especially *P. limnetica*), but that diatoms may be more limited primarily by P and then secondarily by N and Si. Results also suggest that it is an excess of N relative to P that contributes to the formation of the summer *Pseudanabaena limnetica* bloom. The results are logical based upon recent observations in Lake Elsinore. During the first half of 2010, the diatom population biomass in the lake decreased as P concentrations decreased and *P. limnetica* biomass increased and it began to bloom as N concentrations increased and P became scarcer and scarcer in the lake. However, these results only apply to the specific conditions during the experiment and may not be replicable in the lake itself.

Through examination of the communities in Lake Elsinore and some of the factors that influence them, this study has contributed valuable new insight into a lake that is widely used by both people and animals. Some of the factors driving problems that affect water and ecological quality (including the crash of the diatom community and formation of cyanobacterial blooms and the absence of a permanent population of large cladocerans capable of keeping the phytoplankton community in check) are now much more apparent. Thus, this study has established a reference point for future lake monitoring and management and providing an ecological benchmark for future studies.