Influencing factors on plankton populations in a desert man-made lake

Steven Weber
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INFLUENCING FACTORS ON PLANKTON POPULATIONS
IN A DESERT MAN-MADE LAKE

by

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Bachelor of Science
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1992

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ABSTRACT

Influencing Factors On Plankton Populations in a Nevada Man-made Lake

by

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Dr. Shawn Gerstenberger, Examination Committee Chair
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The purpose of this study was to examine and identify the factors that have influenced changes in phytoplankton and zooplankton assemblages during the initial thirteen years of ecological development of Lake Las Vegas, Clark County, Nevada. Univariate and multivariate statistical methods were used to determine the level of influence lake water chemistry, lake physical characteristics and planktonic groups will have on overall plankton community dynamics. Water quality and plankton population data have been collected since 1991 when the lake began filling through the period of study ending December, 2003.

Results indicate the zooplankton biomass was most influenced by conductance, total dissolved solids, water temperature, chlorophyll a and pH. Phytoplankton biomass was most related to TN, nitrogen : P ratio, total dissolved solids, conductivity and water temperature. During the first thirteen years external influences have caused the reservoir plankton populations to change annually due to random disturbances.
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CHAPTER 1

LITERATURE REVIEW

Plankton Ecology

Many factors influence the growth of phytoplankton and zooplankton in freshwater lakes; however, many of these factors are inter-dependent and cannot be adequately addressed independently. Plankton have a wide range of physiological requirements and vary in response to physical and chemical characteristics such as light, temperature, and nutrient requirements (Wetzel 2001). Researchers have considered numerous limnological characteristics when examining factors influencing plankton populations. Despite the intellectual benefits of comparative ecology, several factors complicate its application. One of the primary constraints on comparative ecology is identification of a conceptual or analytical framework that can be applied in analysis of diverse ecosystems and ecosystem types (Downing 1991). This study will explore the influence of salinity, nutrients, and biological influences on the assemblages of plankton in a man-made lake.

When evaluating the evolution of a juvenile man-made lake, one must first establish the time scale being considered. Biomass, a principle measurement of succession, can be produced and accumulate in time periods ranging from days to millennia. The composition of phytoplankton communities and associated relative abundance of individual species undergo continuous change in response to various environmental variables, and the effect of these changes can be observed from days to years or longer. This study will focus on the shorter successional time scale, specifically monthly, seasonal and annual changes that can be measured and adapted to present and future resource management planning. Limnologists and lake managers focus on seasonal fluctuations of biomass. High or excessive phytoplankton and plant biomass in many reservoirs can impede recreation, domestic water use and aesthetics in reservoirs and lakes for the public. Lake and reservoir biomass is comprised of plants and animals, both live and decaying. Animal biomass is present as fish, invertebrates, crustaceans and reptiles.
In addition, waterfowl that rely on these aquatic ecosystems could be included. Plants are represented by the more commonly associated, higher life forms, including primarily the vascular macrophytes and macroalgae. In this study I focused on changes associated with the lower life plant and animal forms, pelagic plankton, as they comprise significant portion of an aquatic ecosystem’s standing crop.

In respect to aquatic systems, similar studies provided sentinel concepts and theories since the early 1900’s. Early ecologists such as Birge, Juday, Hutchinson, Margalef, and Lindeman pioneered a better understanding of aquatic ecosystems.

Lindeman (1942) adapted previous views of ecosystems and incorporated the concept of trophic categories that included producers, consumers, and decomposers. These categories were instrumental in describing the energy flow within lakes and reservoirs. Lindeman (1942) provided a means to mathematically articulate the basic successional process of biomass accumulation that previous investigators had founded, while Clements’ (1936) concept of ecosystem climax is not easily applied to lakes and reservoirs. Margalef (1958) suggested that the concept of climax in plankton communities be dropped and use the terms of “less” and “more” mature ecosystems. Again we need to understand the importance of defining “what is our time scale of study?” There is agreement that natural or anthropogenic disturbance within any ecosystem can halt or accelerate the succession or evolutionary process, but what types of disturbance are considered normal and which are extraordinary? Normal disturbances such as seasonal temperature variations, annual rainfall, and wind can influence seasonal variability; while major watershed changes attributed to natural phenomenon (forest fires, earthquakes, and floods) can have a significant impact on lake and reservoir ecosystems, often completely affecting the long-term species dynamic previously established. Each type of disturbance can influence the time scale of a lake’s succession or ontogeny.

Species diversity is often included in the analysis of successional state. Hutchinson (1961) and Wetzel (1983) found that the co-existence of a number of phytoplankton species is a feature of fresh waters. Although generally a few species dominate a phytoplanktonic assemblage, a number of less dominant species also co-exist. Margalef (1958) found that by examining an ecosystems species diversity, fewer species will be observed in lake systems with high concentrations of nutrients.
Factors Influencing Phytoplankton

Morris (1980) outlined that plankton populations are influenced by both allogenic and autogenic factors. Allogenic factors, those environmental conditions where the organism does not have any controls, include; salinity, light, temperature, turbulence, and anthropogenic sources. Autogenic sources are those environmental conditions that are regulated to a significant degree by the plankton or other trophic levels and include; life cycles, nutrients, water quality, predation, and ectocrines (antibiotics). Those considered in this study are: light, temperature, water chemistry to include plant nutrients and salinity, zooplankton grazing, and planktivorous fish.

Turbulence

Hutchinson (1961) proposed the “Paradox of the Plankton”, a hypothesis that multiple phytoplankton species can inhabit the epilimnion of a lake (in violation of Hardin’s (1960) “principle of competitive exclusion”) by existing in a relatively isotropic environment and all compete for the same resources. Hutchinson hypothesized that the chances of plankton remaining static within the epilimnion of a lake was unlikely due to the presence of ever changing light gradients and effects of vertical and horizontal turbulence. Hutchinson (1961) has a great effect on how ecologists consider the presence and absence of plankton in all aquatic ecosystems.

Richerson, et al. (1970) investigated plankton from Castle Lake, California in an attempt to further understand Hutchinson’s theory. This group of researchers found that physical, chemical, and biological characteristics of the epilimnion can change rapidly ranging in hours to days. This unstable environment led the researchers to report that the epilimnion may contain various niche habitats that allow for multiple phytoplankton assemblages and dominance to occur at frequent and random intervals. The presence of these temporary niches favor the maintenance of several species where growth conditions were different and permit niche diversity.

Light and Water Color

Light and temperature synergistically affect photosynthesis. The behavior of light in water has important water quality implications because it regulates visual aesthetics when combined with nutrients and temperature, light strongly influences phytoplankton growth (Effler et al. 1998). Although photosynthetic rate and algal growth are directly related to irradiance intensity, the response to light intensity, especially at light saturation, is temperature dependant and variable among species (Wetzel
Two attenuating processes regulate the extent of light penetration: absorption and scattering (Effler et al. 1998). A considerable degree of adaptation to changing light intensities occurs among phytoplankton, often by regulation of pigment concentrations per unit biomass (Wetzel 2001). The vertical distribution of photosynthesis of many phytoplankton is strongly related to available light. A near-exponential decline with increasing depth underlies a surficial zone of maximum photosynthesis (Wetzel 2001). As photosynthetic rate per unit volume of water increase in nutrient enriched waters, the biogenic turbidity resulting from dense phytoplankton populations constricts the thickness of the trophogenic zone towards the surface (Wetzel 2001). Prediction with optics models demonstrated substantial improvements in light penetration could be achieved through systematic reduction in phytoplankton biomass (e.g., reduction of external nutrient loading) and/or inorganic tripton (e.g., erosion control) (Effler et al. 1998).

**Temperature**

Phytoplankton have a definite temperature optima and tolerance ranges that interact with other characteristics to cause season succession (Wetzel 2001). Growth of a population under conditions of adequate light and temperature is often limited by a single nutrient. Limitation can shift rapidly from nutrient to nutrient as their availability changes on a diurnal, daily, and season basis (Wetzel 2001).

Temperature also plays an important role in lake chemical solutes and increased phytoplankton standing crops (Schindler et al. 1990). In laboratory studies, lake warming processes have demonstrated that small increases in temperature influence the growth, development, and feeding rates of many individual aquatic organisms (Wooton et al. 1980; McKee and Ebert 1996; Santamaria and van Vierssen 1997). Temperature influence is usually positive on growth and feeding rates below normally occurring optima. On the other hand, in simple microcosm communities increased temperature may be destabilizing, reducing the period of cyclicity of populations and altering trophic relationships (Petchey et al. 1999; Grover et al. 2000). Under field conditions, temperature increases associated with moderate thermal pollution often resulted in changed, macrophyte community composition, increased productivity, and accelerated life cycles (Haag and Gorham 1977; Svenson and Wigen-Svenson 1992; Taylor and Helwig 1995).

Carpenter et al. (1992) and Schneider and Root (1996) have identified that increased temperatures did show a tendency to intensify the water chemistry processes involved with eutrophication. Temperature increase
could be problematic because ecosystems already pushed close to the threshold change may be especially susceptible to further, unpredictable stress events.

**Water Chemistry**

Peterson (1975) expanded the original concept of Hutchinson Paradox of the Plankton concept with his model of nutrient-limited growth, based on Michaelis-Menten kinetics. Peterson's considered that the fewer limiting factors that are in operation, the fewer is the available number of niches, and in turn the fewer the number of plankton species in co-existence. The following sections will discuss the influence of water chemistry to include in great part, nutrients influence on the plankton.

**Alkalinity and pH**

A comparison of the species composition of microcrustacean zooplankton in many Northwest European, North American, and temperate Asian lakes has shown that the number of species present is strongly correlated with pH, with species diversity highest in lakes with a pH ranging from 6.8 to 7.2 (Ivanova 1987). Thus, it seems that the abundance and presence of many zooplankton species are negatively affected by both low and high pH (Vijverberk et al. 1996). Field and laboratory experiments suggest that most cladoceran species have an upper survival pH limit in the range of 10.5 to 11.5 (O'Brien and DeNoyelles 1972; Hansen et al. 1991).

As an example, direct effects that may play a role at high pH are the toxic effects of non-ionized ammonia (NH₃) on *Daphnia* and disruption of ion exchange in *Daphnia* (Vijverberk et al. 1996). In some cyclopoid, copepod, and cladoceran species, the physiological effects of high pH affect the sodium balance (Potts and Fryer 1979; Nilssen et al. 1984). Copepods usually exhibited normal sodium balance up to ~ pH 9.5, but above this pH they exhibited a net sodium loss (Nilssen et al. 1984). In Vijverberk et al. (1996), a laboratory experiment was developed where a NaHCO₃-NaOH buffer was used that resulted in a concentration of Na⁺ in the *Daphnia* medium of approximately 3.4 mmol liter⁻¹, which is high compared with natural lake water and makes it less likely that sodium became limited at higher pH levels. It is conceivable; however, that other metals became limited at high pH levels due to changes in ion/non-ionized associations. Results of culture experiments by Elendt and Bias (1990) suggest that high pH levels caused selenium deficiency in culture media and may cause egg abortions and neonate mortality in *Daphnia*. Vijverberg et al. (1996) observed exactly the same phenomena at high pH so it is tempting to regard Selenium limitation as the possible cause of increased abortions.
Deleterious effects of important abiotic influences such as pH or toxic substances are often stronger at low food levels because these effects usually act via the inability of the organism to keep food intake and assimilation high enough to pay for increased respiration (Reinikainen et al. 1994). Vijverberk et al. (1996) demonstrated that high pH (>10.0) can substantially reduce the egg viability and fitness of microcrustacean zooplankton. A pH > 10.0 is commonly found in many eutrophic and hypereutrophic lakes. Therefore, the effect of high pH on the population dynamics and community composition of microcrustacean zooplankton is probably much more important than has been assumed.

**Nutrients**

The chemical characteristics of a body of water may influence the structure of phytoplankton communities by direct mediation of competitive interactions or by several indirect routes determined by the relationships among the members of the community (Lane and Levins 1977). Consequently, while the response of a phytoplankton community to enrichment is often dramatic and predictable, the mechanisms promoting the response are rarely known (Lynch and Shapiro 1981). Phosphorus (P) and nitrogen (N) commonly limit algae growth in lakes and oceans. The biomass and species compositions of plankton are regulated by the availability of nutrients, principally phosphorus (McQueen et al. 1986; Smith 1983). In freshwaters, phosphorus is the most frequent limiting factor (Schindler 1977), although transitions between phosphorus and nitrogen limitation often occur seasonally and in anthropogenically eutrophic lakes (Wetzel 1983; USEPA 1988). Limitation can shift rapidly from nutrient to nutrient as their availability changes on a diurnal, daily, and seasonal basis (Wetzel 2001).

Theoretical and empirical evidence is accumulating to indicate that food-web structure can indeed be a powerful factor in determining levels of algal abundance and productivity in a reservoir, within the constraints of its overall rate of nutrient loading (Benndorf 1988; Carpenter 1988). McQueen et al. (1986) on the basis of comparative analyses and experimental studies suggested that zooplankton effects on phytoplankton will be greatest in oligotrophic systems where zooplankton can reduce nutrient-limited phytoplankton assemblages.

**Resource Ratios**

Researchers have used resource ratios to explain primary production in freshwater and marine environments. This approach focuses on the relative abundance of critical elements (such as carbon, nitrogen, and phosphorus) during ecological interactions as a means for insight into diverse phenomena.
such as foraging behavior of individuals, population regulation, resource competition, and nutrient limitation of primary production (Sterner et al. 1992). Ecological stoichiometry may be especially appropriate for comparative ecology, because all biotic and most abiotic components of ecosystems can be characterized with respect to ratios of elements such as carbon, nitrogen, and phosphorus (Hassett et al. 1997).

Goldman et al. (1979) took the widespread occurrences of a sestonic stoichiometry near carbon:nitrogen:phosphorous (C:N:P) = 106:16:1 (Redfield ratio) as evidence for nutrient-saturated production of phytoplankton in the oligotrophic ocean, despite extremely low ambient nutrient concentrations. Harris (1986) extended this claim to all kinds of seas and lakes.

In addition, three limiting resource ratios along which phytoplankton species could be sorted: TN:TP, TN:light, TP:light. The ratios were calculated stoichiometrically (mol DIN:mol SRP) (Sommer 1989). Rothhaupt (1995) found that the type of phytoplankton growth limitation is determined by the nutrient that is in minimum supply relative to other nutrients or to light. There are, however, two key interactive factors that potentially complicate the predictive power of N:P ratio analyses. Physically stratified systems yield variable ratios of biologically available or total N:P when comparing epilimnia and hypolimnia. Hence, organisms able to migrate freely could meet requirements for both nutrients without the need for N\textsubscript{2} fixation (Paerl 1988).

Elser et al. (1988) found that N:P ratios of zooplankton nutritional requirements are generally lower than that for phytoplankton. Zooplankton would then inherently recycle nitrogen in greater amounts relative to phosphorus, which would be expressed as changes in relative availability of nitrogen and phosphorus as either zooplankton biomass or size distribution changed (owing to allometric effects). However, certain species of zooplankton could have the same TN:TP requirements as phytoplankton. But if the N:P ratio of excreta exceeded that of egesta, recycling would be higher for nitrogen than for phosphorus, as the products of excretion are available for uptake immediately while the availability of egested nutrients may be delayed (Elser et al. 1988).

Excretion of phosphorus by zooplankton represents recycling of existing nutrients within the water column and not a mechanism of supply (Hargrave et al. 1968). Elser and Goldman (1991) determined that nutrient recycling from zooplankton was important in a number of lakes along a trophic gradient and from the nature of likely limiting factors in oligotrophic and eutrophic systems. During the stratified season in
many lakes, nutrient recycling by zooplankton can satisfy the nutrient requirements of a rapidly growing phytoplankton assemblage (Lehman 1980 and Elser et al. 1988) and can significantly alter its nutritional status in the epilimnion (Bergquist and Carpenter 1986; Elser et al. 1986).

Sedimentation is an important regulating factor for the phytoplankton standing crop and may exceed losses through zooplankton grazing at certain times (Reynolds 1984; Scavia and Fahrenstiel 1987). Sedimentation is the sinking of particles (silt, algae, animal feces, and dead organisms) through the lake water column and their deposition on the lake bottom. These detritus particles are degraded in the water column and in the bottom sediments through oxygen-consuming decomposition processes. Organic matter decomposition, a collective term for the net conversion of organic material back to inorganic compound, occurs through the respiratory activities of all organisms, including bacteria, fungi, and other microbes (USEPA 1988). The sedimentation of the dead plankton is influenced by thermal or chemical density gradients with lake depth (Wetzel 2001). In highly productive systems, sedimentation of plankton can contribute to oxygen depletion within the hypolimnion.

Lehman (1980) found that when nutritional demands of phytoplankton in-situ are estimated from primary production data or nutrient uptake, the cells are using inorganic nutrients at a rate far greater than those at which the substances are supplied from external sources. It is also known that the phosphorus of dead zooplankton is rapidly returned to solution (Cooper 1935; Gardiner 1937) and it has been postulated that phosphates released from living and dead zooplankton liberate inorganic phosphate from the soluble organic phosphorus compounds in the water (Steiner 1938; Margalef 1951). Previous work has shown that for Daphnia, PO$_4^{3-}$ (Peters and Lean 1973) and NH$_3$ (Jacobsen and Comita 1976) are the predominant released forms of phosphorus and nitrogen. Abundant evidence from other sources suggests that the same chemical species are the main release product of copepods as well (Conover and Corner 1968; Butler et al. 1969; Corner and Newell 1967; Jawed 1973).

Tracer studies of the phosphorus circulation in small lakes demonstrated the extreme mobility of inorganic phosphorus (Rigler 1956). There is a rapid removal of phosphate from solution by plankton and an equally rapid release of PO$_4^{3-}$ into the water by plankton. Measurements showing that the concentration of inorganic phosphate in the epilimnion remain constant over a few hours or days, indicate that the rate of removal is in homeostatic balance with regeneration. Temporal changes in phosphate concentrations can be caused by a very slight difference between the rates of removal and release by plankton (Rigler 1961).
An increase in the amount of inorganic phosphorus in water containing high concentrations of zooplankton has been observed and has been interpreted by Gardiner (1937), Cushing (1954) and Steele (1959) as indicating directed excretion of inorganic phosphorus by zooplankton. Margalef (1951) has shown that living zooplankton secrete phosphate into the water. Therefore, the increases of phosphate observed, might have been caused, at least in part, by the hydrolysis of soluble organic phosphorus compounds. Another possible source of error causing phosphorus concentrations to be low would be the uptake of $\text{PO}_4^{3-}$ by suspended and epizootic bacteria. Conover (1961), in experiments using algae that had ingested radioactively tagged phosphorus-32, estimated an excretion rate by *Calanus finmarchicus* of 10% daily, but suggested that the excretion rate of smaller plankton might be higher. Pomeroy et al. (1963) observations confirmed this finding, since they found excretion of phosphorus on the order of 100% daily.

A general tendency for higher N:P ratio in recycled nutrients from zooplankton communities dominated by daphnids than from copepods-dominated communities can explain the results of Elser et al. (1988), who found distinct shifts between nitrogen and phosphorus limited phytoplankton growth accompanying changes in zooplankton community structure. One hypothesis (Schnidler 1977; Niemi 1979) suggests that low TN:TP supply ratios should result in nitrogen limitation of phytoplankton growth and should therefore be associated with blooms of nitrogen fixing cyanobacteria (bluegreen algae) in lakes and estuaries.

Salinity

In arid regions, saline lakes are common and often the dominant aquatic habitats. In many cases diversions of freshwater inputs for irrigation or other human uses have resulted in diminished lake sizes and increased salinity (Jellison 1996). Notable examples of this phenomenon include the Aral Sea in SW Kazakhstan and NW Uzbekistan (Micklin 1988), Mono Lake in Nevada (Patten et al. 1987), and Pyramid Lake in Nevada (Galat et al. 1981). Since most saline lakes exist in hydrologically closed basins and the balance between inputs of freshwater and surface evaporation determines their bathymetric characteristics and salinity levels, they are particularly sensitive to regional climate changes (Jellison 1996).

For limnologists, salinity is the sum of all ions. A conventional salinity value, now widely accepted as the upper limit for healthy freshwater ecosystems is 3,000 mg/L (Mandaville 2000). This value has some physiochemical and biological basis. This salinity concentration is near: 1) the calcite branch point, 2) the lowest points between modes when the frequency distribution of salinity of all lakes over 100
sq. km area is plotted logarithmically, 3) the salinity at which most humans first begin to taste salt, and 4) the point when freshwater biota begins to disappear or not extend (Mandaville 2000).

Three natural mechanisms are said to control the salinity of lakes and rivers: atmospheric precipitation, mineral/soil composition, and the evaporation-crystallization process (Clarke 1924; Gibbs 1970). One can evaluate the relative importance of each mechanism by plotting the weight ratio Na\(^+\)/(Na\(^+\) + Ca\(^{2+}\)) vs. salinity for selected waters. Kilham (1990) found that when total dissolved solid concentrations are greater than 1,000 mg/L, Ca\(^{2+}\) is removed from solution.

Additionally, salinity is often expressed as a measure of the water’s electrical conductance or the sum of the total dissolved solids. In this study salinity was measured and tested as specific conductance and total dissolved solids. In studies of freshwater environments, conductance and total dissolved solids are commonly used to represent salinity, while in marine environments it is measured by the total concentration of all cations in water, calcium, magnesium, sodium, and potassium, and their associated anions, bicarbonate, sulfate, and chloride (Wetzel 1983).

Conductance is defined “as an aqueous solutions ability to carry electric current or the reciprocal of electrical resistance. This ability depends on the presence of ions and their respective concentration, mobility, valence, and relative concentrations, and on the temperature of measurement” (APHA 1985), while total dissolved solids are defined as the organic and inorganic residue that remain after a 60 mL water sample is filtered through a 40 – 60 μm glass-fiber filter disk and evaporated in a crucible at a constant drying temperature of 180 °C (APHA 1985).

Salinity bioassays on phytoplankton (Melack et al. 1985; Herbst and Castenholz 1994) and invertebrate species (Herbst et al. 1988; Dana et al. 1993) demonstrated significant declines in individual measures of productivity with increasing salinity (Jellison 1996). Evans et al. (1996) identified that there were changes in phytoplankton taxa associated with increases in lake water specific conductance (as a measure of salinity). Increases in specific conductance at the lower end of the salinity gradient were generally accompanied by decreases in the frequency and magnitude of phytoplankton blooms (as measured by Chlorophyll a) on a year-to-year basis, although this relationship is poorly defined within a given growing season. The changes in lake phytoplankton populations are greatest in those with the lowest salinity, either due to some direct effect of salinity or as a consequence of salinity induced changes in the numbers and species of grazing zooplankton (Evans et al. 1996).
Biological interactions, such as grazing, may be of greater importance in determining the species composition of the fauna at lower conductivities (<2,000 mmhos cm$^{-1}$) (LaBarbera et al. 1974). Dodson (1970), Sprules (1972), and Zaret (1972) and have shown that these interactions are important in determining the distribution and abundance of certain zooplankton.

**Nitrogen and Salinity**

Studies of marine (Howarth 1988), coastal pond (Caraco et al. 1987), and prairie saline lake systems (Bierhuizen and Prepas 1985; Campbell and Prepas 1986) indicate that nitrogen availability becomes increasingly important in limiting phytoplankton biomass as salinity increases. In most situations where both nitrogen sources are present, ammonium is preferentially assimilated and the nitrate reductase activity is consequently low (Morris and Syrett 1963; Eppley et al. 1969).

**Phosphorus and Salinity**

The relationship between phosphorus and phytoplankton biomass and productivity has been documented in many studies of freshwater systems (Dillon and Rigler 1974; Smith 1979). In saline systems the relationship between phosphorus and phytoplankton growth is not as robust (Evans et al. 1997). Berman et al. (1995) proposed that resource limitation of biologically available phosphorus has been the major factor responsible for restraining increases in primary production and phytoplankton in Lake Kinneret (Israel). It is consistent with empirical analysis of Toetz and McFarlend (1987), however, and with the conclusion of other investigators who have suggested that absolute phosphorus concentrations rather than N:P ratios may be the dominant factor influencing the success of planktonic cyanobacteria (Doremus 1982; Reynolds 1986; Pick and Lean 1987; Trimbee and Prepas 1987; Garnier and Montesanto 1988).

Shapiro (1979) found considerable variability in published phosphorus loading-chlorophyll relationships. The observation that some variations in the total phosphorus-chlorophyll relationship can be attributed to aspects of the zooplankton community (Hrpacek et al. 1978; Pace 1984) indicate that further study of food-web effects on phytoplankton nutrient interactions is warranted (Elser et al. 1988). Sakamoto (1966) noted that the chlorophyll yield in Japanese lakes was logarithmic function of both TP and TN. He concluded that over the range $10 \leq \text{TN} : \text{TP} \leq 17$ by weight, chlorophyll yield was very nearly balanced with respect to both TP and TN but that chlorophyll was dependent only on TN when $\text{TN} : \text{TP} < 10$, and
only on TP when the TN : TP ratio was > 17. Smith (1984) found that the optimal N: P ratio at which switching from nitrogen limitation (N:P <20) to phosphorus (N:P >20) occurs.

**Biologic Interactions**

**Plankton**

**Phytoplankton Inter-actions**

Co-existence of a number of phytoplankton species is a feature of fresh water ecosystems. Although a few species commonly dominate a phytoplanktonic assemblage, a number of less dominant species co-exist among the dominant species. Many differences in physiological characteristics, nutrient requirements, and tolerances, as well as season and spatial variations in environmental parameters, permit the apparent multi-species equilibrium to exist for short periods (Wetzel 1985). Sommer (1981) compared seasonal variability. LaBounty & Burns (2005) provided a summary of zooplankton and phytoplankton species found in Lake Mead, Nevada during the 2000 through 2004. Their grouping also supports Sommer's hypothesis regarding the annual cycle of groupings by size and growth class in phytoplankton to the morphology and performance of individual species. He found that species of similar size and growth class tended to associate with each other during the annual cycle. The smaller, fast growing species were observed during the spring where the large, slow growing species were present in mid-summer. Sommer attributed this phytoplankton ranking in dominance and time to reproductive strategy difference between r and K section. As nutrients become less available in the summer months, populations trend toward K-strategist.

A number of phytoplankton and zooplankton indices have been developed (Thunmark 1945; Nygaard 1949; Elster et al. 1970; and Einsle 1983)). These indices have attempted to identify, with limited success, the relationships of phytoplankton assemblages (Wetzel 1985).

A compound index of all algal groups against Bacillariophyceae was developed and a general relationship was observed. Low index value indicated that the water body exhibited more oligotrophic characteristics. Inversely, as the compound index increases, the water body was more eutrophic. Nygaard (1949) interpreted compound quotients, which appeared to be the most reliable indicators of trophic status, of less than 1.0 to indicate oligotrophic lake conditions, values of 0.0-0.3 suggesting dystrophy. Values greater than 1.0 probably indicate eutrophy, and those between 5 and 20 indicate a high degree of eutrophy.
with possible contamination by cattle excrement. More recent studies have shown a poor correlation between modern measurements of productivity and Nygaard’s index (Wetzel 1985).

An inverse relationship generally exists between phytoplankton biomass and productivity per unit biomass in the phytoplankton. Often small species of relatively minor contribution to the phytoplankton community biomass have short generation times and contribute more to total primary productivity than do large species (Wetzel 1985).

Lampert (1977) introduced the concept of “threshold” food levels, distinguishing between two types. The population threshold is the amount of food necessary for reproduction to exactly offset mortality, resulting in zero change in population size. Maintenance metabolism primarily requires energy, while growth requires many other essential building blocks: elements, essential biomolecules, etc. As a general rule, it might be the case that expressions of food that are closely related to energy, such as carbon, are sufficient at low food density. During growth, however, energy alone does not appear to be sufficient in all circumstances, and other variables must be taken into account. Body nutrient contents relate to ecological processes such as those described in the study of consumer-resource stoichiometry, but such nutrient contents relate to body growth and not to maintaining a given biomass in maintenance metabolism (Sterner et al. 1994).

A distinct periodicity in the biomass of phytoplankton is observed in fresh water ecosystems. LaBounty and Burns described this phenomenon observed in neighboring Lake Mead, Nevada. Growth is greatly reduced or negligible during the winter period and normally increases in the spring period under improved sunlight conditions. A distinct period of clear water is typical of the spring algal succession in mesotrophic and eutrophic lakes of temperate zones (Sommer et al. 1986). An early peak of small, rapidly growing phytoplankton (flagellates and Bacillariophyceae), which often represent the highest biomass concentration of the annual cycle, is followed by a short period of very clear water and high Secchi transparency (Lampert et al. 1986; Gaedke 1992; Vanni et al. 1990). The clear water phase usually coincides with a spring peak of filter feeding zooplankton and the increase in water clarity has sometimes been attributed to grazing activity (Lampert and Schober 1978, LaBounty and Burns 2005). Alternatively nutrient depletion, climatic events, or parasitism have been suggested as causes for the phytoplankton crash (Reynolds 1984). In most instances this clear-water occurs because herbivorous zooplankton become abundant and graze phytoplankton populations to low levels. Lampert et al. (1986) clearly demonstrated

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that the clear water phase resulted from a season maximum of *Daphnia* in late spring. Water transparency subsequently declined as *Daphnia* populations decreased precipitously in early summer, in spite of high *Daphnia* abundance and increased water clarity being linked in other studies (Shapiro and Wright 1984; Benndorf 1987; Carpenter et al. 1987; LaBounty and Burns 2005).

Lampert et al. (1986) stated that two conditions must be met to produce a clear water phase: the phytoplankton standing stock must consist of small, edible cells, and the biomass of filter feeding zooplankton must reach high levels. A phytoplankton community dominated by small cells is typical for spring, whereas large, bizarrely shaped or gelatinous (inedible) algae develop during the summer, probably in response to grazing (Porter 1977). Sommer et al. (1986) found that cladocerans were primary filter feeders in their study. The timing of their increase is probably determined by temperature and supported by hatching ephippia. In other lakes, however, there may also be a component of invertebrate predation that delays the population increase (Lamper and Schober 1978).

The ability of large-bodied daphnids to reduce phytoplankton assemblages typical of eutrophic lakes (Mills et al. 1987; Vanni 1987) appears to allow food-web manipulation to be considered a viable management tool for improving the summer clarity of eutrophic lakes. Lundstedt et al. (1991) suggested that the mid-summer decline of the cladoceran community could be explained by qualitative changes in the phytoplankton community, whereas it previously had been ascribed to predation. The importance of predation in summer has probably been overestimated, which suggests that the availability of suitable resources may be more important in determining population dynamics and interactions between zooplankton species than previously considered. In many eutrophic lakes, the summer decline is short lived and blue-green algae (cyanophyta) and dinoflagellates will bloom and persist until fall disruption of thermal stratification. Some lakes can experience a secondary fall maximum that is predominately comprised of Bacillariophyceae and is generally not as pronounced as the spring event (Vanni et al. 1990).

Zooplankton populations will typically peak during late spring near the time of turnover and will then decrease following the onset of thermal stratification (Wetzel 2001). In many eutrophic lakes, the summer decline is short lived and blue-green algae (cyanophyta) and dinoflagellates will bloom and persist until fall disruption of thermal stratification. Some lakes can experience a secondary fall maximum that is predominately comprised of Bacillariophyceae and is generally not as pronounced as the spring event (Vanni et al. 1990).
Zooplankton are likely to experience long-term (seasonal) shifts in phytoplankton resources over their lifespan, some of which cause food limitation of reproduction. Short-term changes in resources will also be encountered if predation forces zooplankton to migrate vertically to depths where food quality and quantity are reduced (Williamson et al. 1996). Gliwicz (1969) found a relationship between the trophic state of a lake, the size distribution of the zooplankton food, and the zooplankton species occurring in the lake. Gliwicz implied that, within the limits set by the trophic condition of a lake, the zooplankton species determine the size distribution of their food.

Inter-annual changes, in particular, remain one of the least investigated areas in limnology. The need for further characterization and understanding of this annual variability is important for several reasons. Aside from their intrinsic interest, the variance associated with inter-annual fluctuations obscure the understanding of this phenomena observed over longer and shorter time scales. Long-term trends, for example, can be masked by inter-annual variability (Likens 1983; Goldman 1988). Similarly, the results of short-term, whole lake experiments may be subject to misinterpretation if the experiments are performed in anomalous years (Schindler 1987).

### Zooplankton and Phytoplankton Relationships

Zooplankton species may affect phytoplankton size-frequency distributions (via size-selective feeding) even though phytoplankton density does not appear to determine herbivore species distribution (Dodson et al. 1976). Becker et al. (2003) identified four factors that determine the nutritional quality of phytoplankton for zooplankton; 1) the size and the morphology of the algae (Brendelberger 1991), 2) toxicity of phytoplankton owing to certain compounds (Turner and Tester 1997), 3) the mineral composition of the phytoplankton (Turner and Tester 1997), and 4) the biochemical features of the phytoplankton such as the fatty acid content (Muller-Navarra 1995). Becker et al. (2003) summarized that different aspects of food quality not only changed in importance depending on the severity of the limitations present, but also that they might play a role in different phases of an animal’s life. Phosphorus was found to be the overriding limitation. If phosphorus is not found in suitable concentrations, the other limiting factors were not important.

As several investigations have revealed, the zooplankton-phytoplankton interaction is complex, with algal responses to zooplankton being species specific (Lehman and Sadgren 1985; Elser et al. 1987)
and often non-linear, owing to the negative grazing mortality and positive growth stimulation because of nutrient recycling effects of zooplankton on phytoplankton (Bergquist and Carpenter 1986; Sterner 1986). Elser et al. (1991) generalized that the pattern of zooplankton effects on phytoplankton observed in their experiments support certain views of the nature of algal-grazer interactions as a function of lake trophic status. Changes in phytoplankton community composition were dynamic and many changes were associated with changes in the zooplankton community (Elser et al. 1987; Elser and Carpenter 1988).

The weak effects of grazers, such as *Daphnia pulicaria*, on the large-sized blue-green dominated alga assemblages in eutrophic Clear Lake, California confirm the view that grazer impacts should weaken in eutrophic and hypereutrophic systems (McQueen et al. 1986; Sommer et al. 1986; Carney and Elser 1990). In such nutrient rich systems, colonial and other large algal taxa can dominate, lessening the ability of crustacean zooplankton to graze them significantly (Elser et al. 1991). Copepods, rotifers and some cladocera with the exception of *Daphnia pulex* may utilize blue-green algae to a great degree (Wright 1958). Blazka (1966) reported successful growth and reproduction of feeding on a bloom of blue-green algae; it is possible that the bacteria present were also an important factor.

Gliwicz (1969; 1975), Porter (1973), and Nadin-Hurley and Duncan (1979) found that zooplankton feed selectively according to the size, taste, and morphology of their prey. The ability of zooplankton to ingest various sizes of phytoplankton has also been thought to determine zooplankton community structure in lakes of different trophic status (Brooks and Dodson 1965; Makarewicz and Likens 1979; McCauley and Kalff 1981; McCauley 1983). *Daphnia*'s dominance in lakes is attributed to their foraging mode and ability to consume a broader range of particles than other rotifers, copepods, and other smaller cladocerans (Tessier et al. 2001).

Carney and Elser (1990) also proposed that the importance of macro-zooplankton grazing would be greatest in lakes of intermediate productivity. Lehman (1976) even suggested that a model that predicts the behavior of filter feeders is incomplete if it ignores the size-selective ingestion of food particles. Food particle size selection by zooplankton has been found in laboratory experiments (e.g. Mullin 1963; Gliwicz 1970; McQueen 1970; Arnold 1971; DeMott 1982), and Peters and Downing (1984) have tried to quantify the general effect of food particle size on grazing rate. However there have been few attempts to measure *in situ* particle size selection. Brett et al. (2000) suggest that zooplankton growth will be limited by the food quality of phytoplankton communities whenever these communities are not strongly dominated by
Bacillariophyceae, cryptophytes, or other highly nutritious phytoplankton. According to Ramos-Rodriguez et al. (2003), low phosphorus or nitrogen concentrations in the medium increase the quality of *Cryptomonas* as a food resource for *Kertella*. Zooplankton populations could depend not only on the algal size as determinant of edibility but also on the taxonomic features of the phytoplankton species and their specific nutrient requirements.

Several field examples suggest that the ratio of diaptomids to daphnids is an inverse function of lake productivity (Elster et al. 1970; Einsle 1983; Patalas 1972; Pace 1986; George et al. 1990). Rotifer species are more susceptible than *Daphnia* or copepods to nutrient limitation, especially phosphorus limitation (Morales-Baquero and Conde-Porcuna 2000). Cond-Porcuna et al. (2002) observed that the abundance of some rotifer species were not correlated with food availability but showed a strong dependence on phosphorus availability in a reservoir. Different susceptibilities of zooplankton species to nutrient limitation could be important in explaining the dynamics of these organisms in natural situations. Rothhaupt (1995) and Conde-Porcuna (2000) showed that phosphorus limitation significantly reduces the growth rates of the rotifer *Brachionus* and *Anraeopsis*.

It is often difficult to define the precise ecological requirements of a species. In fact, it is generally recognized that such ecological details are poorly known for zooplankton. Frequently if two species of cyclopoids are found co-existing in a lake or reservoir they are almost invariably of different genera. Similarly, when two limnetic calanoids are found together, it is very seldom that they are both species of *Diaptomus* (Pennak 1957). In the works of Carl (1940) on Canadian lakes, strong evidence for the fact that a genus of copepods or cladocerans is seldom represented by more than one species in limnetic samples.

Pennak (1957) found that a particular species attains numerical dominance only in a transient sense, since it characteristically is cyclic in its seasonal occurrence and may be abundant at one time and represented by only a few individuals a short time previously or subsequently. Rarely does the most abundant species of zooplankton account for less than fifty (50) percent of all of the individual zooplankton taxa present in vertical tow net and vertical series trap samples.

Herbivorous zooplankton can be limited by the amount and quality of their food resources (Lampert 1985). Planktonic rotifers and cladocerans can be very sensitive to toxic cyanobacteria (Fulton et al. 1987; Gilbert 1990; DeMott 1991; Smith et al. 1995). Therefore, the occurrence of such blue-green
algae in plankton communities has the potential to directly alter the population dynamics of susceptible zooplankton and to shift the species structure of zooplankton communities in favor of species that either do not ingest the blue-green algae, or are unaffected by the toxins of the blue-green algae. The community changes may then affect organisms at lower and higher trophic levels (Gilbert 1996). Tezuka (1971) and O’Brien et al. (1972) found that severe inhibition of filtering rate after blooms of blue-green algae is most likely not a result of secreted toxins by a physiological response of the zooplankton to high pH levels by increases in photosynthesis. The presence of Oscillatoria could diminish the usefulness of a food population that otherwise would permit successful growth. The effect reduces the ability of Daphnia to compete with Diaptomus and other organisms that do not have the rejection reaction. The inhibitory mechanism is not general for all blue-green algae nor for all filamentous organisms. Daphnia can thrive in the presence of large colonies of Anabaena, Microcystis, or Aphnizomenon, which can be larger than the Daphnia itself (Lynch 1980) Berman et al. (1992) showed that phytoplankton in Lake Kinneret, Israel seemed to be relatively resistant to environmental changes and that there had been no extreme long-term increase in the static phytoplankton characteristics.

**Planktivorous Fish**

Temperature, light, turbulence, and nutrient concentrations are usually considered the main factors determining the growth rates of plankton, whereas the influence of biological interactions has traditionally received less interest. Introduction of an intensely planktivorous fish, such as crappies, into a lake tends to cause a decrease in the size of zooplankton species. This in turn may result in an increase in the density of large phytoplankton, such as are often associated with “nuisance” conditions in a lake (Stross 1973). If a lake is well supplied with nutrients, fish introductions could lead to a significant decrease in water quality (Dodson et al. 1976; Vanni 1987; Gliwicz and Pijanowska 1989).

Planktonic community structure can also be affected through predation by planktivorous fish (Lazzaro 1987; Northcote 1988). McQueen et al. (1986) hypothesized that regulation of planktonic communities by fish changes with lake trophic state. Specifically, they suggest that suppression of zooplankton by fish has a more intense effect on phytoplankton in oligotrophic lakes than in eutrophic or hypereutrophic systems. In statistical terms they are hypothesizing that an interaction effect exists between fish effects and lake trophic effects (Drenner et al. 1989). Berman et al. (1995) found that there was a steady decline in zooplankton (mainly cladocera and copepods) attributed to increased predation by fish.

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Freshwater zooplankton communities typically are dominated by either large- or small-bodied species. Mueller (1955) observed that shad populations in Lake Mead had a significant influence on seasonal distribution and abundance of *Daphnia pulex*.

Lakes containing abundant planktivorous fish contain mostly small species of zooplankton, the result of the elimination of large species by size-selective fish predation (Hrbacek 1962; Brooks and Dodson 1965; Lynch 1979). Luecke, et al. (1990) indicated that reductions in planktivorous fish abundance would have little impact on peak daphnid abundances in spring, but will likely cause an increase in mean summer biomass of daphnids.

Many studies have considered nutrient release by fish to be an important source of nutrients to phytoplankton (e.g., Brabrand et al. 1990; Carpenter et al. 1992; Schindler et al. 1993), and a few studies have provided experimental evidence that direct nutrient recycling by fish affects phytoplankton community structure (Reinersten et al. 1986; Vanni and Findlay 1990; Schindler 1992; Vanni and Layne 1997; Attayde and Hanson 1999). Persson (1997) found that phytoplankton biomass was significantly enhanced when fish predation and excretion acted together but not when they acted alone and suggested that both are important mechanisms by which fish affect phytoplankton. Vanni and Layne (1997) provided experimental evidence suggesting that nutrient excretion by fish is an important mechanism controlling phytoplankton communities, but it is not clear from their results whether the effects of excretion by fish were more important than the effects of predation on zooplankton.

It has been demonstrated, in lakes where planktivorous fish are abundant that an increase in concentrations of TP and TN in the water column occurs (Schindler and Eby 1997; Vanni et al. 1997). The relative importance of fish predation and excretion should change along the trophic gradient. Nutrient excretion by fish may be more important in lakes with low phosphorus inputs and a relatively high biomass of planktivorous fish, which may arise when piscivorous fish are absent or rare and when planktivorous fish consume considerable amounts of littoral/benthic resources (Vanni and Layne 1997).
CHAPTER 2

INTRODUCTION

As outlined in the literature review, plankton species may be influenced by many environmental factors. Categorically, those factors that can influence phytoplankton succession can be described as physical, chemical, and biological processes. In many cases not one single variable can predict or describe changes in a population, but various combinations may cause unique plankton assemblages to occur. In many freshwater ecosystems, succession can be measured in various time scales. Species change and diversity can occur on a daily, weekly, monthly, seasonal basis as well as annual and geologic time. In addition, seasonal succession changes are often influenced by changes in lake trophic state, which may be a result of natural or anthropogenic phenomenon.

Purpose

The purpose of this study is to examine and identify the environmental factors and management decisions that have influenced changes in phytoplankton and zooplankton assemblages during the initial thirteen years of Lake Las Vegas, Clark County, Nevada.

In April 1989, Transcontinental Corporation began construction of a 130 ha reservoir adjacent to the Lake Mead National Recreation Area (LMNRA) boundary. This reservoir is the focal point of a 1,337 ha master planned destination community called “Lake Las Vegas Resort” (hereafter referred to as “Resort”) located in Henderson, Nevada (Figure 1). Unique to this Resort is that the reservoir was constructed within the Las Vegas Wash channel by constructing a 1.6 km long, 45.8 meter high earthen dam, but the Wash base flows (approximately 300 c.f.s.) are bypassed under the reservoir via two 213.4 cm diameter concrete pipes. Only in storm events of greater than three-year frequencies does the Las Vegas Wash water flow into the reservoir. Construction of the dam and its other three spillways were completed in May 1991 and the process of filling the reservoir commenced.
At full pool elevation the reservoir is 3.2 km in length and one mile wide with 19.8 km of shoreline. Normal reservoir operating levels fluctuate two to four feet annually and have an average storage capacity of 13,050,218 m$^3$. The maximum depth of the reservoir is 43 m with an average depth of 10 m. Untreated Lake Mead water is the primary source of makeup water and is provided by the City of Henderson via the Basic Water Company raw water delivery system. Annually the reservoir loses an average of approximately 2 m of reservoir elevation to evaporation, which is equivalent approximately to 2,466,960 m$^3$ of water. This rate of evaporate loss is consistent with the evaporative rate observed at Lake Mead.

The reservoir provides a recreational amenity to the master-planned Resort’s property owners and hotel guests. In addition, the reservoir serves as an irrigation source for the Resort’s three existing golf courses.
Lake Las Vegas is a warm monomiotic reservoir that exhibits the characteristics of a mesotrophic system.

Typically seventy-five percent of the shoreline is at a 4:1 slope or steeper inhibiting the growth of vascular plants. Emergent aquatic plants are limited to *Typha latifolia*, *Typha domingensis*, and *Phragmites australis*, while a limited submergent community of *Najas marina* and *Potamogeton sp* are also present. Upon completion of the surrounding resort development, the reservoir's shoreline will be one hundred percent improved.

The reservoir's management plan does not include the use of pesticides to control nuisance plants or animals. Biologic control is encourages by selective stocking of fish species to minimize insect development and limit zooplankton grazing.

Ambient air temperatures range between a low of -1.1°C in the winter and a high of 47.2°C in the summer (National Weather Service). Corresponding reservoir water temperatures range between a low of 7.3°C in the winter and a high of 28.2°C in the summer.

Questions and Hypotheses

The following five research questions and hypotheses were developed to assist in and identifying the factors that have influenced changes in phytoplankton and zooplankton assemblages in Lake Las Vegas.

**Question 1**

Is there a relationship between zooplankton assemblages and available phytoplankton assemblages as food sources?

**Hypothesis 1**

As a food source phytoplankton do not influence zooplankton assemblages.

**Method 1**

Plankton were analyzed by taxonomic divisions using univariate correlation analysis. Cluster analysis was used to verify seasonal similarities by grouping monthly surface water temperatures. These seasonal clusters or groups were used to calculate seasonal plankton biomass estimates. These estimates were then analyzed using univariate correlation, autoregression, and partial least squares analysis to determine what plankton attributes cause assemblage presence between zooplankton and phytoplankton.

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**Question 2a.**

Is there a relationship between annual changes in lake water phosphorus species, nitrogen species, nitrogen and phosphorus ratios, major ions, and pH and zooplankton assemblages?

Zooplankton assemblage fluctuations are related to changes in phytoplankton biomass, not lake water nitrogen, phosphorus, nitrogen and phosphorus ratios, pH, or salinity.

**Method 2a.**

Annual and Seasonal water quality variables were estimated using the same clustering protocol used in method 1. Zooplankton divisions established in question one (1) were analyzed against the water quality characteristics to determine what relationships exist. Univariate correlation and partial least squares analysis were used to establish statistical relationships.

Autoregression analysis was used to determine time lag relationship between seasonal and environmental changes in water quality and food source availability for zooplankton grazing.

Partial least squares regression (PLS) was used to provide a multivariate model with the capability to analysis multiple dependent and independent variables simultaneously. PLS was chosen due the models strengths in accommodating smaller data sets without respect to normality and its ability to account for and correct autocorrelations.

**Question 2b.**

What correlation is there between lake water phosphorus species, nitrogen species, nitrogen and phosphorus ratios, major ions, and pH and phytoplankton biomass and assemblages?

**Hypothesis 2b.**

Phytoplankton dominance is not influenced by lake water nitrogen, phosphorus, nitrogen and phosphorus ratios, pH, or salinity.

**Method 2b**

Annual and Seasonal water quality variables were estimated using the same clustering protocol used in method 1. Phytoplankton divisions from method 1 were analyzed against the water quality characteristics to determine what relationships exist. Univariate correlation and partial least squares analysis was used to establish statistical relationships.
Autoregression analysis was used to determine time lag relationship between seasonal and environmental changes in water quality and food source availability for phytoplankton grazing.

Partial least squares regression (PLS) was used to provide a multivariate model with the capability to analysis multiple dependent and independent variables simultaneously. PLS was chosen due the models strengths in accommodating smaller data sets without respect to normality and its ability to account for and correct autocorrelations.

Question 2c.
Do zooplankton and phytoplankton assemblages follow seasonal limnological trends?

Hypothesis 2c.
Plankton assemblages do not follow seasonal limnological trends.

Method 2c.
Plankton assemblages from method 1 were analyzed against time as determined in method 2b. Univariate correlation, partial least squares, and multivariate regression were used to establish statistical relationships and predictive models.

Autoregression analysis will be conducted to determine lag time between seasonal and environmental changes in water quality and food source availability for zooplankton grazing. Upon completion, trend analysis was completed to determine what population trends exist between plankton communities.

Partial least squares regression (PLS) was used to provide a multivariate model with the capability to analysis multiple dependent and independent variables simultaneously. PLS was chosen due the models strengths in accommodating smaller data sets without respect to normality and its ability to account for and correct autocorrelations.

Question 3
What effect does changing salinity (total dissolved solids and conductivity) concentrations have on Daphnia pulex populations?

Hypothesis 3
Daphnia pulex populations will not change as lake water salinity changes.

Method 3
Univariate correlation and multivariate partial least squares analysis were conducted to determine whether salinity stated as a measurement of total dissolved solids and specific conductance has an impact on *Daphnia pulex*. 
CHAPTER 3

METHODS

Lake Las Vegas Sampling Locations

In 1991, Lake Las Vegas Resort began a water quality monitoring program to fulfill its United States Army Corps of Engineers permit obligations. This permit required that a special amendment to the states 208 Water Quality Management Plan be made for the creation of the reservoir. As part of this amendment, a long term water quality monitoring program was developed and implemented. The water quality monitoring program would be conducted on Lake Las Vegas monthly in January, February, November, and December, and/or biweekly during March and October, and weekly during April through September. Water quality monitoring was conducted at the four (4) stations shown in Figure 2, at fixed points along the historical center channel in the deepest areas of the reservoir. Spatial characteristics for each of the sites are summarized in Table 1. Table 1 provides the physical characteristics of each of the monitoring stations. This study considered the physical, chemical and biological data collected at stations LLV-1 and LLV-1A from 1991 to 2003 (Table 2).
Figure 2. Location of water quality monitoring stations at Lake Las Vegas, Clark County, Nevada.

Table 1. Monitoring Stations Characteristics for Lake Las Vegas, Clark County, Nevada.

<table>
<thead>
<tr>
<th>Station Identification</th>
<th>Depth (m)</th>
<th>Latitude</th>
<th>Longitude</th>
</tr>
</thead>
<tbody>
<tr>
<td>LLV-1</td>
<td>30</td>
<td>N 36° 7' 157&quot;</td>
<td>W 114° 54' 797&quot;</td>
</tr>
<tr>
<td>LLV-1A</td>
<td>30</td>
<td>N 36° 6' 961&quot;</td>
<td>W 114° 55' 074&quot;</td>
</tr>
<tr>
<td>LLV-2</td>
<td>15</td>
<td>N 36° 6' 713&quot;</td>
<td>W 114° 55' 241&quot;</td>
</tr>
<tr>
<td>LLV-3</td>
<td>9</td>
<td>N 36° 36' 337&quot;</td>
<td>W 114° 55' 590&quot;</td>
</tr>
</tbody>
</table>
Physical Measurements

Physical water quality data were collected with a Hydrolab Surveyor Model III Water Quality Analyzer or a YSI Water Quality Analyzer. Each device uses a multi-probe sonde that measures temperature, dissolved oxygen, pH, and specific conductance. The probe is lowered through the reservoir’s vertical profile via a calibrated cable and measurements were recorded at two (2.0) meter intervals from one (1.0) meter below the reservoir’s surface to approximately one (1.0) meter above the reservoir bottom at each of the sampling stations.

Transparency or lake water clarity was measured at each monitoring station by lowering a standard bi-colored 30.5 cm Secchi disc into the water via a calibrated chain on the shaded side of the boat. The depth that the disk is no longer visible is known as the Secchi depth.

Chemical Measurements

Depth integrated water samples were collected from zero (0) to two and one half (2.5) meters at the monitoring stations (Figure 2) for the following: nitrite + nitrate (NO₂-N+NO₃-N), ammonia (NH₄-N), total kjeldal nitrogen, ortho-phosphorus (PO₄-P), TP, total dissolved solids, total suspended solids and chlorophyll a. Quarterly samples were collected and analyzed from LLV-1A for calcium, bicarbonate, sodium, magnesium, potassium, chloride, and sulfate (Table 2). Samples were collected with an integrated sampling device, manufactured and sold under the trade name of “Sludge Judge” that is made of 3.18 cm diameter clear PVC pipe that has a small one-way valve installed on the bottom and is similar to a well bailer. This allows the sample to remain in the sampler when removed from the water and also permits the collector to view the column of water and identify if there are any visible differences in the sample. The sampler was rinsed with lake water at each station prior to collecting the sample and all field equipment is acid washed and rinsed with distilled water on a monthly basis. Samples were transferred into Nalgene sample bottles were stored in an ice cooler until delivered to the laboratory for analysis. Each bottle was marked with sample identification criteria to include: date, sample station, analysis requested, and who obtained the sample. Field duplicates were obtained on a frequency of ten (10) percent of samples collected.
A State of Nevada certified laboratory performed chemical and biological analyses using Environmental Protection Agency (EPA) approved methods unless otherwise noted (Table 2). Table 2 outlines the analytical characteristics and methods.

Phytoplankton

Over the thirteen-year period phytoplankton were collected at a minimum quarterly and as frequently as weekly from the surface waters of Lake Las Vegas (0 - 2.5 m) at site LLV-1A. Samples were collected with the previously discussed integrated sampling device. In this study only the samples collected near the 15th of each month were included. This was done since phytoplankton and zooplankton were collected at monthly frequencies.

Samples were collected by Lake Las Vegas field crews and transferred to 250 mL Nalgene bottles and fixed with a Lugols solution. Samples were shipped to Janik, Inc. of Davis, California for phytoplankton identification and enumeration.

Zooplankton

Over the thirteen-year period monthly zooplankton samples were collected at station LLV-1 in a vertical tow from 0-15 m with an 80 μm Wisconsin plankton net. Samples were transferred to 250 mL Nalgene bottles and shipped to Janik, Inc. of Davis, California for zooplankton identification and enumeration.
Table 2. Lake Las Vegas Water Quality Sampling Measurements identifying depth, frequency, and methods.

<table>
<thead>
<tr>
<th>Sampling Program</th>
<th>Type</th>
<th>Variable</th>
<th>Depth (m)</th>
<th>Method</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Physical</td>
<td>Temperature (°C)</td>
<td>2.0 meter intervals</td>
<td>Electronic multimeter and sonde</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Dissolved Oxygen (mg/L)</td>
<td>surface to 1.0 m above bottom</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>pH (std. units)</td>
<td>Integrated 0 – 2.5 m</td>
<td>EPA 180.1</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Conductivity (µS/m)</td>
<td>Integrated 0 – 2.5 m</td>
<td>EPA 160.1</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Turbidity (NTU)</td>
<td>Integrated 0 – 2.5 m</td>
<td>EPA 160.2</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Total Suspended Solids (mg/L)</td>
<td>Variable</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>Total Dissolved Solids (mg/L)</td>
<td>Variable</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Secchi (m)</td>
<td>Variable</td>
<td>Variable</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Chemical</td>
<td>Total Nitrogen (TN) (µg/L)</td>
<td>Integrated 0 – 2.5 m</td>
<td>EPA 365.2</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Ammonia (NH₄-N) (µg/L)</td>
<td>Integrated 0 – 2.5 m</td>
<td>EPA 350.2</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Nitrite + Nitrate (NO₂-N + NO₃-N) (µg/L)</td>
<td>Integrated 0 – 2.5 m</td>
<td>EPA 350.2</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Total Phosphorus (TP) (µg/L)</td>
<td>Integrated 0 – 2.5 m</td>
<td>EPA 365.2</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Ortho-Phosphorus (PO₄-P) (µg/L)</td>
<td>Integrated 0 – 2.5 m</td>
<td>EPA 200.7</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Major Anions/Cations (mg/L)</td>
<td>Variable</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Biological</td>
<td>Chlorophyll a (µg/L)</td>
<td>Integrated 0 – 2.5 m</td>
<td>APHA (1995)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Phytoplankton (mg/m³)</td>
<td>Integrated 0 – 2.5 m</td>
<td>APHA (1995)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Zooplankton (mg/m³)</td>
<td>0 – 15 m Vertical Tow</td>
<td>APHA (1995)</td>
</tr>
</tbody>
</table>

Phytoplankton

Counting Procedure

The inverted-microscope method or Utermohl method (Utermohl 1958, Kellar et al. 1980, 1984) was used for enumeration and identification of phytoplankton samples. The procedure incorporates a stratified design using at least three (x 78, 280, 560) magnifications (Janik 1984). The rational for this approach is that phytoplankton in most lakes have greatest axial linear dimension (GALD) that span three orders of magnitude from 1-2 µm to 1000 µm or more for filamentous taxa.
Sample Sedimentation of Phytoplankton

Wild™ and Hydro-Bios™ combined plate chambers consisting of a top cylinder (Sedimentation cylinder) of 10 mL capacity and a bottom-plate chamber (base plate) were used. The bottom diameter of the base chamber is 25.5 mm. Volumes sedimented range from 2.0 – 10.0 mL depending of algal density.

Biovolumes

Cell volumes are calculated based on the measurements of at least 20 individuals of each species and the geometrical formulae, which most closely approximates the cell shape (Lund et al. 1958). Cell sizes are measured at x 560 with a calibrated ocular micrometer. For most organisms the measurements are taken from outside cell wall to outside cell wall.

Zooplankton Sample Preparation and Counting Procedure

Samples were analyzed with a Wild M40 inverted phase contrast microscope (Wetzel and Likens, 1979). Samples will be counted at: x 78, higher magnification of x 280, and 560 are available to facilitate identifications.

The zooplankton sample is mixed by gently inverting the sample bottle for 30 seconds. A wide-bore automatic pipette is used to withdraw 2.9 mL of sample and fill a Hydro-Bios combination plate chamber. A cover slip is then placed on top of the chamber and allowed to settle for 15 minutes before counting. A second chamber is then prepared for a total of 5.8 mL for each sample. The entire 510 mm² plate chamber is counted in continuous strips.

Statistical Analysis

Samples collected during the study period were analyzed and the corresponding data were categorized and stored in Microsoft Access databases. An individual database was developed for each category of data collected: physical water quality data, water chemistry data, and plankton. In this study, data were compiled, sorted, graphed, and analyzed using the following software programs: Microsoft Access 2000, Microsoft Excel 2000, Microsoft Word 2000, SPSS 7.0 for Windows, SAS version 8 statistical software, Sigma Stat version 3.0, and Sigma Plot version 9.0.1.
As previously stated, the period of study for this project spanned from July 1991 to December 2003 and data were analyzed on data collected on or near the fifteenth day of each month. Data were evaluated in two fashions, first on a whole to include all data and second in grouping of similar temporal characteristic. To determine what logical groupings made sense, hierarchical cluster analysis was implemented and statistical analysis was completed using SPSS 7.0 for Windows. Cluster analysis was used to verify seasonal similarities by considering monthly surface water temperatures. These seasonal clusters were then used to calculate seasonal plankton biomass estimates and water chemistry characteristics.

A hierarchical approach (Figure 3) to statistical analysis was used where all data sets were first described and compared for unusual trends or data outliers. Next, data sets were analyzed using univariate correlation to determine what correlations may exist between the variables samples and if a positive or negative influence is observed.

The correlation findings were then used to develop a stepwise autoregression model that tested whether a time lag occurred between seasonal and environmental changes in water quality and food source availability for both zooplankton and phytoplankton. The stepwise autoregression method was selected since it has the ability to eliminate many autoregressive lags and then sequentially removes autoregressive parameters until all remaining autoregressive parameters have significant t-tests (SAS, 1999). Using SAS statistical software (version 8), stepwise autoregressive process was performed using the Yule-Walker method. In addition, to insure that error terms were independent a Durbin-Watson test was the method used of testing and identifying autocorrelated variables.

Partial least squares regression (PLS), a multivariate model, was selected due its capability to analysis multiple dependent and independent variables simultaneously. PLS is a robust model that accommodates smaller data sets, without respect to normality, and will take into account autocorrelation variables. In addition, PLS will test all variables and explain any covariance that may occur within the data set. In this study PLS models were developed to study the relationships between phytoplankton, zooplankton and water chemistry.
Figure 3. Provides a graphical depiction of the statistical procedures used in this study. In most cases the statistical test overlapped between research questions due to the inter-related nature of variables studied.
CHAPTER 4

RESULTS

Lake Fill and Level

Upon completion of the reservoir in 1991, Resort management began the filling process. Figure 4 illustrates the sequence and timing of the filling of the reservoir and illustrates that full pool elevation was first obtained in January 1996. Since 1997, the reservoir has maintained an average elevation of 1,402 feet based on North American Vertical Datum 1988 (NAVD88) plus or minus six (6) feet.

Figure 4. Reservoir elevation for Lake Las Vegas, Clark County, Nevada for the period of 1991 through 2003. All elevations are referenced to North American Vertical Datum 88.
Source water is derived from two sources: Lake Mead and storm water resulting from rainfall in the Las Vegas Valley watershed. Since 1991, approximately 20,420,000 m$^3$ of Lake Mead water has been pumped into the reservoir via the Basic Management pipeline. An additional 30,840,000 m$^3$ of storm water has filled the reservoir during this period (Figure 4). The Resort has appropriated state water rights to collect storm water from the Las Vegas Wash in the amount of 2,503,000 m$^3$ per year.

Descriptive Statistics

In reviewing the descriptive statistics, the variability of the water quality characteristics varies due to both natural and anthropogenic influences. During the period of study these influences had a profound impact on water quality. Table 3 outlines those chronological events of significance.

Appendix I and II of this paper provide the descriptive statistics of all the samples collected during the study period of July 1991 to December 2003. Sample size (n), mean ($\bar{x}$), standard deviation (SD), coefficient of variation (CV) and range are reported on an annual and study period basis. The CV is calculated by dividing the standard deviation by the mean of the population or data set and multiplied by 100. The CV is an expression of variability to the mean and is useful for comparisons because it is a utilities measure.
<table>
<thead>
<tr>
<th>Year</th>
<th>Event</th>
<th>Environmental Impact</th>
</tr>
</thead>
<tbody>
<tr>
<td>1991</td>
<td>Reservoir began filling. 4,193,832 m³ of Lake Mead water added to reservoir.</td>
<td>Lake ontogeny began</td>
</tr>
<tr>
<td>1992</td>
<td>4,853,743.8 m³ of storm water spilled into the reservoir. Elevation increased 20 ft.</td>
<td></td>
</tr>
<tr>
<td>1993</td>
<td>183,788.52 m³ of storm water spilled into reservoir. Zooplankton harvested from Lake Mead stocked in reservoir. No Lake Mead water added.</td>
<td>Lake clarity measured an average of 6.5 meters.</td>
</tr>
<tr>
<td>1994</td>
<td>86,343.6 m³ of storm water spilled into reservoir. Largemouth Bass, channel catfish, and bluegill stocked in reservoir. 1,549,250.8 m³ of Lake Mead water added.</td>
<td>Total Dissolved Solids exceeds 2,000 mg/L.</td>
</tr>
<tr>
<td>1995</td>
<td>2,523,700 m³ of storm water spilled into reservoir. Large storm event in January raised reservoir elevation from 1,397.6 to 1,403.5 ft. 754,889.76 m³ of Lake Mead water added.</td>
<td>Large nutrient input from Las Vegas Wash.</td>
</tr>
<tr>
<td>1996</td>
<td>376,211.4 m³ of storm water spilled into reservoir. SouthShore Golf Course completed. 2,537,268.3 m³ of Lake Mead water added.</td>
<td>Ponds overflow into lake. Monitoring program began to study impacts to reservoir. Total Dissolved Solids exceeds 2,200 mg/L.</td>
</tr>
<tr>
<td>1997</td>
<td>204,757.68 m³ of storm water spilled into reservoir. No Lake Mead water added.</td>
<td></td>
</tr>
<tr>
<td>1998</td>
<td>10,191,011 m³ of storm water spilled into reservoir (classified El Nino year). Reflection Bay Golf Course completed. 3,970,572.1 m³ of Lake Mead water added.</td>
<td>Four fairways constructed adjacent to reservoir. Runoff monitored.</td>
</tr>
<tr>
<td>1999</td>
<td>Summer Flood of Record occurred July 9th. 17,000 cfs entered lake passing approx. 6,104,492.5 m³ of storm water through spillways. Fertilizer runoff into reservoir from Golf course due to rain fall immediately following application. No Lake Mead water added.</td>
<td>Localized cyanophyta bloom due to change in nutrient balance. High BOD load and poor lake water clarity for remainder of year. Total Dissolved Solids exceeds 2,500 mg/L.</td>
</tr>
<tr>
<td>2000</td>
<td>3,368,633.8 m³ of storm water spilled into reservoir. 108,546.24 m³ of Lake Mead water added.</td>
<td></td>
</tr>
<tr>
<td>2001</td>
<td>312,070.44 m³ of storm water spilled into reservoir. Large scale dredging operation removed 244,672 m³ from west end of reservoir. 2,516,299.2 m³ of Lake Mead water added.</td>
<td>Upon removal of silt curtains from work area, large amounts of nutrients were dispersed through the reservoir causing phytoplankton blooms for two growing seasons. Largest annual population of Chlorophyta in reservoir.</td>
</tr>
<tr>
<td>2002</td>
<td>3,119,470.9 m³ of Lake Mead water added.</td>
<td>Largest annual population of Cyanophyta, Chloromonadophyta and Pyrrophyta in reservoir.</td>
</tr>
<tr>
<td>2003</td>
<td>225,726.84 m³ of storm water spilled into reservoir. 2,681,585.5 m³ of storm water spilled into reservoir. 1,107,665 m³ of Lake Mead water added.</td>
<td>Total Dissolved Solids exceeds 2,600 mg/L.</td>
</tr>
</tbody>
</table>
Seasonal Analysis

Lake water chemistry and biological activities are typically influenced by temperature and in some cases salinity. In this study lake water surface temperature (0 meters) was used as the variable to cluster. Surface water temperature was selected over other measurements, since water temperature is relatively consistent year-to-year and temperature has a strong influence over seasonal lake turnover and mixing. Figure 5 illustrates the months that share similar water temperatures. Three distinct groups present themselves: summer (June, July, August, September), spring/fall (March, April, May, October, November), and winter (December, January, February). These three seasonal grouping were used repeatedly through this study to understand the seasonal effects on plankton populations. To further support this grouping, Hutchinson in his second volume of *A Treatise on Limnology*, defined the similar seasonal groupings based on his research of thermal maxima and minima (Hutchinson 1967).

![Figure 5. Rescaled Distance Cluster Combine. Seasonal groupings as determined by a hierarchical cluster analysis using surface water temperature as the independent variable. The analysis used a nearest neighbor linkage and a squared Euclidean distance interval.](image-url)
The seasonal distribution of zooplankton was evaluated and Figure 6 illustrates the relative frequency of the three primary groups of zooplankton found in the reservoir. Copepods dominated the population during the three seasonal groups outlined as winter (69%), spring/fall (64%) and summer (67%); followed by cladoceran (29%, 34%, 23%) and then rotifers (2%, 2%, 10%), respectively. These data were compiled from the entire study period and does not accentuate the seasonal fluctuation that has occurred in individual sampling years (Figure 7). During the summer periods a notable decline in cladoceran frequency was observed and conversely a notable increase in rotifers was observed. This observation is well documented and was discussed earlier in the literature review. Annual zooplankton relative frequency fluctuated when copepods and cladocerans dominated the population (Figure 7). Annual variability can be linked to environmental influences such as storm events that result in periodic shifts in reservoir water chemistry.
Figure 6. Zooplankton relative frequency (%) found in Lake Las Vegas, Clark County, Nevada for the study period of 1991 through 2003.
Seasonal phytoplankton population relative frequencies were dominated by those taxa included in the division Chlorophyta (winter 18.7%, spring/fall 72.6%, summer 41.7%), while populations during the spring/fall period had notable increases in taxa from the division Cyanophyta (winter 2%, spring/fall 26%, summer 10%), and the summer period populations were dominated by taxa from Chlorophyta (44%) and Chrysophyta, of the class Bacillariophyceae, that consists of those algae known as diatoms (Figure 8). Annual phytoplankton relative frequency varied between study years with 2000-2002 exhibiting unusually high concentrations of Chlorophyta. This is attributed to the species, *Pyramachlamys dissecta*. During the late winter of 1999 and early 2000, *Pyramachlamys dissecta* dominated all phytoplankton populations during all seasons during this period and skewed the relative importance of Chlorophyta in many of my analyses.
Figure 8. Phytoplankton relative frequency (%) found in Lake Las Vegas, Clark County, Nevada for the period of 1991 through 2003.
Figure 9. Annual relative frequency of phytoplankton populations in Lake Las Vegas, Clark County for the period of 1991 through 2003.

In Table 4, the relative frequency and number of species present during the three seasonal groups are outlined. Additionally included in the parenthesis are the relative frequency and number of species calculated excluding the 2000-2002 sampling years. The 2000-2002 sampling years represented a period when *Pyramachlamys dissecta* dominated the reservoir's phytoplankton biomass. Approximately a 10% increase in relative frequency for Chlorophyta was observed during these years. Bacillariophyceae importance increased by 5% percent and Pyrrhophyta decreased by 15%. A similar phenomenon was observed at Lake Mead during this same period (LaBounty and Burns 2005). Many scientists have attempted to describe the cause of the *Pyramachlamys dissecta* bloom during this period, but have not been able to establish a universally accepted hypothesis. Most of the theories involve the temporal addition of
phosphorus from the Las Vegas Wash, storm water and higher than normal water temperatures caused by mild winter weather patterns.

Table 4. Effects of increased water temperature and dredging during 2000 through 2002 on phytoplankton species count and relative frequency for Lake Las Vegas, Clark County, Nevada for the period of 1991 through 2003. Numbers are expressed as percentages.

<table>
<thead>
<tr>
<th>Division</th>
<th>Summer Spec. (#)</th>
<th>Spring/Fall Spec. (#)</th>
<th>Winter Spec. (#)</th>
<th>Summer RFREQ (%)</th>
<th>Spring/Fall RFREQ (%)</th>
<th>Winter RFREQ (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Bacillariophyceae (Diatoms)</td>
<td>12, 9</td>
<td>12, 6</td>
<td>10, 7</td>
<td>23, 16</td>
<td>4, 3</td>
<td>9, 15</td>
</tr>
<tr>
<td>Chlorophyta (Greens)</td>
<td>30, 21</td>
<td>21, 18</td>
<td>12, 12</td>
<td>31, 42</td>
<td>64, 73</td>
<td>74, 19</td>
</tr>
<tr>
<td>Chrysophyta (Goldens)</td>
<td>6, 6</td>
<td>6, 5</td>
<td>4, 3</td>
<td>2, 6</td>
<td>0, 2</td>
<td>4, 17</td>
</tr>
<tr>
<td>Crytophyta (Cryomonads)</td>
<td>7, 6</td>
<td>7, 7</td>
<td>7, 6</td>
<td>1, 2</td>
<td>2, 7</td>
<td>4, 16</td>
</tr>
<tr>
<td>Cyanobacteria (Bluegreens)</td>
<td>28, 17</td>
<td>21, 11</td>
<td>11, 6</td>
<td>17, 25</td>
<td>17, 9</td>
<td>5, 18</td>
</tr>
<tr>
<td>Pyrrhophyta (Dinoflagellates)</td>
<td>8, 0</td>
<td>7, 6</td>
<td>3, 2</td>
<td>20, 5</td>
<td>10, 1</td>
<td>0, 1</td>
</tr>
<tr>
<td>Euglenophyta</td>
<td>2, 1</td>
<td>1, 0</td>
<td>0, 0</td>
<td>0, 0</td>
<td>0, 0</td>
<td>0, 0</td>
</tr>
<tr>
<td>Haptophyta</td>
<td>1, 1</td>
<td>1, 1</td>
<td>1, 1</td>
<td>6, 5</td>
<td>2, 6</td>
<td>5, 14</td>
</tr>
<tr>
<td>Chloromonadophyta</td>
<td>0, 0</td>
<td>1, 0</td>
<td>0, 0</td>
<td>0, 0</td>
<td>1, 0</td>
<td>0, 0</td>
</tr>
<tr>
<td>Total Species</td>
<td>94, 61</td>
<td>77, 54</td>
<td>48, 37</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Average GALD</td>
<td>148.8</td>
<td>116.5</td>
<td>126.5</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>AWG (Average Weighted GALD)</td>
<td>32.4</td>
<td>38.4</td>
<td>16.3</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Maximum GALD</td>
<td>1038.0</td>
<td>905.0</td>
<td>1038.0</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Minimum GALD</td>
<td>3</td>
<td>3.0</td>
<td>3.0</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>


Tables 5, 6, and 7 reflect the species assemblages observed during the study period for each of the three seasonal groups. A total of two hundred and nineteen (219) species were observed in the reservoir over the study period. Of these species, ninety-four (94) were observed over the summer months, dominated by *Anomoeoneis vitrea* (8.3%), *Cyclostella sp.* (5.8%), *Cylotella maneghiniana* (5.3%), *Pyramichlamys dissecta* (18.6%), *Gloeocystis ampla* (9.8%), *Peridinium penardiforme* (19.0%), *Chrysochoromulina parva* (6.1%). Seventy-seven (77) species were observed during the spring/fall months, dominated by *Pyramichlamys dissecta* (53.9%), *Gloeocystis ampla* (8.4%), *Planktothrix rubescens* (11%), *Peridinium sp.* (5.6%). Forty-eight (48) species were observed during the winter months, dominated by *Cyclostella bodanica* (4.5%), *Pyramichlamys dissecta* (69.5%), and *Chrysochoromulina parva* (4.9%).
Measurements of the phytoplankton's greatest axial linear dimensions (GALD) were measured and reported in Tables 5, 6, and 7. GALD is a common measurement used to categorize phytoplankton using a standardized method. In general, those species of smaller growth form were observed during the winter when nutrients were readily available and when the reservoir is not thermally stratified. This is consistent with the observations of Sommer (1981) as discussed in the literature review. In consideration of Sommer's findings related to the presence of r-strategist in the fall, winter, and spring and k-strategist in the summer to test Sommer's theory of seasonal influence on rank strategist phytoplankton dominance, \( \bar{x} \) weighted GALD average (AWG) was calculated for each seasonal group adjusting GALD by individual species relative frequency (Table 4). AWG for the winter groups was 16.3 and 39.4 and 32.4 for spring/fall and summer respectively. Figure 10 illustrates those months that share similar GALD values. It is evident that species seasonal size (GALD) distribution does not accurately account for all the variability in phytoplankton assemblages. I will elaborate on why this may be in the discussion chapter.
Figure 10. Rescaled Distance Cluster Combine. Hierarchical cluster analysis of average monthly GALD values for Lake Las Vegas, Clark County, Nevada. The analysis used a nearest neighbor linkage and a squared Euclidean distance interval.
Table 5. Winter phytoplankton biomass, frequency, relative frequency, and GALD summary for Lake Las Vegas, Clark County for the period of 1995 through 2003.

<table>
<thead>
<tr>
<th>Division</th>
<th>Genus/Species</th>
<th>Winter 1995-2003</th>
<th>Total Biomass (mg/m$^3$)</th>
<th>Freq (%)</th>
<th>Rfreq (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Bacillariophyceae</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Cyclotella bodanica</td>
<td>15.6</td>
<td>478.4</td>
<td>50.9</td>
<td>4.5</td>
</tr>
<tr>
<td></td>
<td>Cyclotella sp.</td>
<td>7.2</td>
<td>286.4</td>
<td>30.5</td>
<td>2.7</td>
</tr>
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Table 5 (continued). Winter phytoplankton biomass, frequency, relative frequency, and GALD summary for Lake Las Vegas, Clark County for the period of 1995 through 2003.

<table>
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<tr>
<th>Division</th>
<th>Genus/Species</th>
<th>GALD</th>
<th>Total Biomass (mg/m$^3$)</th>
<th>Freq (%)</th>
<th>Rfreq (%)</th>
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Table 6. Spring/Fall phytoplankton biomass, frequency, relative frequency, and GALD summary for Lake Las Vegas, Clark County for the period of 1995 through 2003.

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<th>Genus/Species</th>
<th>GALD</th>
<th>Total Biomass (mg/m²)</th>
<th>Freq (%)</th>
<th>Rfreq (%)</th>
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<td>Freq (%)</td>
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Table 7. Summer phytoplankton biomass, frequency, relative frequency, and GALD summary for Lake Las Vegas, Clark County for the period of 1995 through 2003.

<table>
<thead>
<tr>
<th>Division</th>
<th>Genus/Species</th>
<th>Total Biomass (mg/m³)</th>
<th>Freq (%)</th>
<th>Rfreq (%)</th>
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<tr>
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<td>Bacillariophyceae</td>
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Table 7 (continued). Summer phytoplankton biomass, frequency, relative frequency, and GALD summary for Lake Las Vegas, Clark County for the period of 1995 through 2003.

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<th>Genus/Species</th>
<th>GALD</th>
<th>Total Biomass (mg/m$^3$)</th>
<th>Freq (%)</th>
<th>Rfreq (%)</th>
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<td>Coccoid Blue-Greens</td>
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<td>12.8</td>
<td>2.1</td>
</tr>
<tr>
<td></td>
<td>Cyanobacterium sp</td>
<td>3.6</td>
<td>991.5</td>
<td>12.8</td>
<td>2.1</td>
</tr>
<tr>
<td></td>
<td>Anabaena sp.</td>
<td>101.1</td>
<td>671.4</td>
<td>8.6</td>
<td>1.5</td>
</tr>
<tr>
<td></td>
<td>Anabaena aphanizemenoide</td>
<td>193.2</td>
<td>1,034.2</td>
<td>13.3</td>
<td>2.2</td>
</tr>
<tr>
<td></td>
<td>Pseudanabaena limnetica</td>
<td>73.6</td>
<td>1,585.7</td>
<td>20.4</td>
<td>3.4</td>
</tr>
<tr>
<td></td>
<td>Planktolyngbya limnetica</td>
<td>49</td>
<td>264.9</td>
<td>3.4</td>
<td>0.6</td>
</tr>
<tr>
<td></td>
<td>Aphanoacapsa delicatissima</td>
<td>11.3</td>
<td>158.5</td>
<td>2.0</td>
<td>0.3</td>
</tr>
<tr>
<td></td>
<td>Aphanizomenon flos aquae</td>
<td>133.5</td>
<td>90.9</td>
<td>1.2</td>
<td>0.2</td>
</tr>
<tr>
<td></td>
<td>Pseudanabaena galeata</td>
<td>59.4</td>
<td>86.2</td>
<td>1.1</td>
<td>0.2</td>
</tr>
<tr>
<td></td>
<td>Oscillatoria sp.</td>
<td>1038</td>
<td>77.9</td>
<td>1.0</td>
<td>0.2</td>
</tr>
<tr>
<td></td>
<td>Lyngbya lagereimii f. minor</td>
<td>45</td>
<td>59.9</td>
<td>0.8</td>
<td>0.1</td>
</tr>
<tr>
<td></td>
<td>Aphanocapsa elachista</td>
<td>36</td>
<td>53.0</td>
<td>0.7</td>
<td>0.1</td>
</tr>
<tr>
<td></td>
<td>Merismopedia tenissima</td>
<td>1.3</td>
<td>39.7</td>
<td>0.5</td>
<td>0.1</td>
</tr>
<tr>
<td></td>
<td>Lyngbya Birgei</td>
<td>505</td>
<td>28.4</td>
<td>0.4</td>
<td>0.1</td>
</tr>
<tr>
<td></td>
<td>Spirulina subsalsa</td>
<td>205.5</td>
<td>28.4</td>
<td>0.4</td>
<td>0.1</td>
</tr>
<tr>
<td></td>
<td>Aphanothece nidulans</td>
<td>1.6</td>
<td>26.1</td>
<td>0.3</td>
<td>0.1</td>
</tr>
<tr>
<td></td>
<td>Synechocystis aequatilis</td>
<td>3.1</td>
<td>24.4</td>
<td>0.3</td>
<td>0.1</td>
</tr>
<tr>
<td></td>
<td>Pseudanabaena sp.</td>
<td>56</td>
<td>23.3</td>
<td>0.3</td>
<td>0.1</td>
</tr>
<tr>
<td></td>
<td>Synechocystis sp.</td>
<td>3</td>
<td>21.2</td>
<td>0.3</td>
<td>0.0</td>
</tr>
<tr>
<td></td>
<td>Oscillatoria angustissima**</td>
<td>125</td>
<td>20.1</td>
<td>0.3</td>
<td>0.0</td>
</tr>
<tr>
<td></td>
<td>Planktolyngbya contorta</td>
<td>36</td>
<td>18.7</td>
<td>0.2</td>
<td>0.0</td>
</tr>
<tr>
<td></td>
<td>Coelosphaerium pallidum</td>
<td>32.5</td>
<td>4.8</td>
<td>0.1</td>
<td>0.0</td>
</tr>
<tr>
<td></td>
<td>Chroococcus dispersus</td>
<td>0.9</td>
<td>3.2</td>
<td>0.0</td>
<td>0.0</td>
</tr>
<tr>
<td></td>
<td>Merismopedia minima</td>
<td>12</td>
<td>2.2</td>
<td>0.0</td>
<td>0.0</td>
</tr>
<tr>
<td></td>
<td>Dactylococcus irregul.</td>
<td>22</td>
<td>0.2</td>
<td>0.0</td>
<td>0.0</td>
</tr>
<tr>
<td></td>
<td>TOTAL CYANOBACTERIA</td>
<td></td>
<td>7,773.6</td>
<td>100.0</td>
<td>16.8</td>
</tr>
</tbody>
</table>
Table 7 (continued). Summer phytoplankton biomass, frequency, relative frequency, and GALD summary for Lake Las Vegas, Clark County for the period of 1995 through 2003.

<table>
<thead>
<tr>
<th>Division</th>
<th>Genus/Species</th>
<th>GALD</th>
<th>Total Biomass (mg/m²)</th>
<th>Freq (%)</th>
<th>Rfreq (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Pyrrhophyta</td>
<td><em>Peridinium penardiforme</em></td>
<td>37.9</td>
<td>8,795.9</td>
<td>96.7</td>
<td>19.0</td>
</tr>
<tr>
<td></td>
<td><em>Gymnodinium sp.</em></td>
<td>12</td>
<td>98.4</td>
<td>1.1</td>
<td>0.2</td>
</tr>
<tr>
<td></td>
<td><em>Glenodinium sp.</em></td>
<td>18.8</td>
<td>78.8</td>
<td>0.9</td>
<td>0.2</td>
</tr>
<tr>
<td></td>
<td><em>Glenodinium pulvisculus</em></td>
<td>18</td>
<td>53.8</td>
<td>0.6</td>
<td>0.1</td>
</tr>
<tr>
<td></td>
<td><em>Peridinium sp.</em></td>
<td>37.6</td>
<td>37.1</td>
<td>0.4</td>
<td>0.1</td>
</tr>
<tr>
<td></td>
<td><em>Microflagellates 3-10 um</em></td>
<td>6</td>
<td>13.4</td>
<td>0.1</td>
<td>0.0</td>
</tr>
<tr>
<td></td>
<td><em>Glenodinium armatum</em></td>
<td>16.8</td>
<td>8.3</td>
<td>0.1</td>
<td>0.0</td>
</tr>
<tr>
<td></td>
<td><em>Peridinium wisconsinense</em></td>
<td>37.8</td>
<td>5.8</td>
<td>0.1</td>
<td>0.0</td>
</tr>
<tr>
<td></td>
<td>TOTAL PYRROPHYTA</td>
<td></td>
<td>9,091.5</td>
<td>100.0</td>
<td>19.6</td>
</tr>
<tr>
<td>Euglenophyta</td>
<td><em>Trachelomonas sp.</em></td>
<td>17.1</td>
<td>31.1</td>
<td>84.3</td>
<td>0.1</td>
</tr>
<tr>
<td></td>
<td><em>Euglena sp.</em></td>
<td>35</td>
<td>5.8</td>
<td>15.7</td>
<td>0.0</td>
</tr>
<tr>
<td></td>
<td>TOTAL EUGLENOPHYTA</td>
<td></td>
<td>36.9</td>
<td>100.0</td>
<td>0.1</td>
</tr>
<tr>
<td>Haptophyta</td>
<td><em>Chrysochromulina parva</em></td>
<td>3</td>
<td>2,841.6</td>
<td>100.0</td>
<td>6.1</td>
</tr>
<tr>
<td></td>
<td>TOTAL HAPTOPHYTA</td>
<td></td>
<td>2,841.6</td>
<td>100.0</td>
<td>6.1</td>
</tr>
<tr>
<td></td>
<td>SUMMER TOTAL</td>
<td></td>
<td>46,273.7</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Tables 8, 9, and 10 provide the descriptive statistics for the three seasonal groupings, winter, spring/fall, and summer. Each Table provides the sample size (n), mean, standard deviation (SD), Coefficient of Variance (CV), and the range. Surprisingly, the mean for the variables measured did not vary greatly between seasons. As would be expected, summer average chlorophyll $a$ concentrations (8.3 µg/L) were greater than winter and spring/fall concentrations; respectively 4.6 µg/L and 3.1 µg/L.
Table 8. Winter water quality descriptive statistics by season for Lake Las Vegas, Clark County, Nevada for the period of 1991 through 2003.

<table>
<thead>
<tr>
<th>Variable</th>
<th>n</th>
<th>Mean</th>
<th>SD</th>
<th>CV (%)</th>
<th>Range</th>
</tr>
</thead>
<tbody>
<tr>
<td>Chlorophyll <em>a</em> (µg/L)</td>
<td>37</td>
<td>4.6</td>
<td>6.5</td>
<td>141.5</td>
<td>[1.0, 30.0]</td>
</tr>
<tr>
<td>pH (S.U.)</td>
<td>35</td>
<td>8.1</td>
<td>0.3</td>
<td>3.4</td>
<td>[7.4, 8.8]</td>
</tr>
<tr>
<td>Temperature (°C)</td>
<td>36</td>
<td>18.4</td>
<td>6.2</td>
<td>33.9</td>
<td>[7.3, 26.9]</td>
</tr>
<tr>
<td>Secchi (m)</td>
<td>34</td>
<td>4.3</td>
<td>2.6</td>
<td>60.5</td>
<td>[0.8, 11.0]</td>
</tr>
<tr>
<td>Conductivity (µS/m²)</td>
<td>37</td>
<td>2942.8</td>
<td>503.9</td>
<td>17.1</td>
<td>[1860.0, 3950.0]</td>
</tr>
<tr>
<td>Total Dissolved Solids (mg/L)</td>
<td>37</td>
<td>2247.6</td>
<td>362.3</td>
<td>16.1</td>
<td>[1338.0, 2754.0]</td>
</tr>
<tr>
<td>PO₄-P (µg/L)</td>
<td>37</td>
<td>3.4</td>
<td>4.5</td>
<td>133.5</td>
<td>[1.0, 26.0]</td>
</tr>
<tr>
<td>TP (µg/L)</td>
<td>37</td>
<td>19.4</td>
<td>11.2</td>
<td>57.5</td>
<td>[8.0, 58.0]</td>
</tr>
<tr>
<td>NO₂-N+NO₃-N (µg/L)</td>
<td>37</td>
<td>631.2</td>
<td>409.4</td>
<td>64.9</td>
<td>[18.0, 1540.0]</td>
</tr>
<tr>
<td>NH₄-N (µg/L)</td>
<td>37</td>
<td>87.9</td>
<td>94.4</td>
<td>107.3</td>
<td>[6.0, 470.0]</td>
</tr>
<tr>
<td>Total Kjedahl Nitrogen (µg/L)</td>
<td>37</td>
<td>844.9</td>
<td>540.6</td>
<td>64.0</td>
<td>[130.0, 3200.0]</td>
</tr>
<tr>
<td>TN (µg/L)</td>
<td>37</td>
<td>1476.2</td>
<td>816.0</td>
<td>55.3</td>
<td>[173.0, 3990.0]</td>
</tr>
<tr>
<td>AN:OP</td>
<td>37</td>
<td>382.7</td>
<td>288.2</td>
<td>75.3</td>
<td>[12.0, 942.0]</td>
</tr>
<tr>
<td>TN:TP</td>
<td>37</td>
<td>90.4</td>
<td>55.8</td>
<td>61.7</td>
<td>[9.0, 235.0]</td>
</tr>
<tr>
<td>Total Phytoplankton</td>
<td>37</td>
<td>550.4</td>
<td>1045.8</td>
<td>190</td>
<td>[0.0, 4446.0]</td>
</tr>
<tr>
<td>Total Zooplankton</td>
<td>37</td>
<td>216.0</td>
<td>619.3</td>
<td>286.7</td>
<td>[0.0, 4012.0]</td>
</tr>
</tbody>
</table>

Table 9. Spring and Fall water quality descriptive statistics by season for Lake Las Vegas, Clark County, Nevada for the period of 1991 through 2003.

<table>
<thead>
<tr>
<th>Variable</th>
<th>n</th>
<th>Mean</th>
<th>SD</th>
<th>CV (%)</th>
<th>Range</th>
</tr>
</thead>
<tbody>
<tr>
<td>Chlorophyll <em>a</em> (µg/L)</td>
<td>61</td>
<td>3.1</td>
<td>4.0</td>
<td>127.8</td>
<td>[1.0, 20.0]</td>
</tr>
<tr>
<td>pH (S.U.)</td>
<td>60</td>
<td>8.0</td>
<td>0.2</td>
<td>3.0</td>
<td>[7.3, 8.7]</td>
</tr>
<tr>
<td>Temperature (°C)</td>
<td>62</td>
<td>18.6</td>
<td>7.0</td>
<td>37.4</td>
<td>[7.7, 29.2]</td>
</tr>
<tr>
<td>Secchi (m)</td>
<td>60</td>
<td>4.3</td>
<td>2.1</td>
<td>49.1</td>
<td>[1.0, 9.3]</td>
</tr>
<tr>
<td>Conductivity (µS/m)</td>
<td>62</td>
<td>2858.7</td>
<td>399.2</td>
<td>14.0</td>
<td>[1860.0, 3950.0]</td>
</tr>
<tr>
<td>Total Dissolved Solids (mg/L)</td>
<td>62</td>
<td>2255.0</td>
<td>351.8</td>
<td>15.6</td>
<td>[1320.0, 2776.0]</td>
</tr>
<tr>
<td>PO₄-P (µg/L)</td>
<td>62</td>
<td>3.2</td>
<td>3.6</td>
<td>112.8</td>
<td>[1.0, 27.0]</td>
</tr>
<tr>
<td>TP (µg/L)</td>
<td>62</td>
<td>19.0</td>
<td>10.3</td>
<td>54.2</td>
<td>[5.0, 49.0]</td>
</tr>
<tr>
<td>NO₂-N+NO₃-N (µg/L)</td>
<td>62</td>
<td>637.7</td>
<td>408.6</td>
<td>64.1</td>
<td>[6.0, 1570.0]</td>
</tr>
<tr>
<td>NH₄-N (µg/L)</td>
<td>60</td>
<td>65.3</td>
<td>67.2</td>
<td>103.0</td>
<td>[4.0, 342.0]</td>
</tr>
<tr>
<td>Total Kjedahl Nitrogen (µg/L)</td>
<td>62</td>
<td>680.4</td>
<td>309.0</td>
<td>45.4</td>
<td>[122.0, 1456.0]</td>
</tr>
<tr>
<td>TN (µg/L)</td>
<td>62</td>
<td>1318.1</td>
<td>600.8</td>
<td>45.6</td>
<td>[262.0, 2516.0]</td>
</tr>
<tr>
<td>AN:OP</td>
<td>60</td>
<td>354.4</td>
<td>299.3</td>
<td>84.5</td>
<td>[9.0, 1405.0]</td>
</tr>
<tr>
<td>TN:TP</td>
<td>62</td>
<td>89.3</td>
<td>66.8</td>
<td>74.8</td>
<td>[10.0, 354.0]</td>
</tr>
<tr>
<td>Total Phytoplankton</td>
<td>62</td>
<td>216.0</td>
<td>619.3</td>
<td>286.7</td>
<td>[0.0, 4012]</td>
</tr>
<tr>
<td>Total Zooplankton</td>
<td>62</td>
<td>40124.9</td>
<td>346320.0</td>
<td>863.1</td>
<td>[0.0, 148411.0]</td>
</tr>
</tbody>
</table>

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Table 10. Summer water quality descriptive statistics by season for Lake Las Vegas, Clark County, Nevada for the period of 1991 through 2003.

<table>
<thead>
<tr>
<th>Variable</th>
<th>N</th>
<th>Mean</th>
<th>SD</th>
<th>CV (%)</th>
<th>Range</th>
</tr>
</thead>
<tbody>
<tr>
<td>Chlorophyll a (µg/L)</td>
<td>51</td>
<td>8.3</td>
<td>29.1</td>
<td>349.5</td>
<td>[1.0, 183.0]</td>
</tr>
<tr>
<td>pH (S.U.)</td>
<td>46</td>
<td>8.0</td>
<td>0.3</td>
<td>4.2</td>
<td>[6.7, 8.8]</td>
</tr>
<tr>
<td>Temperature (°C)</td>
<td>51</td>
<td>18.2</td>
<td>6.3</td>
<td>34.8</td>
<td>[7.9, 28.2]</td>
</tr>
<tr>
<td>Secchi (m)</td>
<td>50</td>
<td>4.3</td>
<td>2.5</td>
<td>58.0</td>
<td>[0.5, 10.5]</td>
</tr>
<tr>
<td>Conductivity (µS/m)</td>
<td>51</td>
<td>2823.7</td>
<td>385.1</td>
<td>13.6</td>
<td>[1760.0, 3310.0]</td>
</tr>
<tr>
<td>Total Dissolved Solids (mg/L)</td>
<td>51</td>
<td>2270.3</td>
<td>400.1</td>
<td>17.6</td>
<td>[1216.0, 2836.0]</td>
</tr>
<tr>
<td>PO₄-P (µg/L)</td>
<td>51</td>
<td>3.5</td>
<td>3.7</td>
<td>104.3</td>
<td>[1.0, 22.0]</td>
</tr>
<tr>
<td>TP (µg/L)</td>
<td>51</td>
<td>20.6</td>
<td>18.6</td>
<td>90.2</td>
<td>[5.0, 114.0]</td>
</tr>
<tr>
<td>NO₂-N+NO₃-N (µg/L)</td>
<td>51</td>
<td>766.6</td>
<td>562.5</td>
<td>73.4</td>
<td>[13.0, 1822.0]</td>
</tr>
<tr>
<td>NH₄-N (µg/L)</td>
<td>48</td>
<td>58.2</td>
<td>65.9</td>
<td>113.3</td>
<td>[3.0, 310.0]</td>
</tr>
<tr>
<td>Total Kjedahl Nitrogen (µg/L)</td>
<td>51</td>
<td>867.3</td>
<td>542.7</td>
<td>62.6</td>
<td>[140.0, 2270.0]</td>
</tr>
<tr>
<td>TN (µg/L)</td>
<td>51</td>
<td>1633.9</td>
<td>1012.6</td>
<td>62.0</td>
<td>[164.0, 4092.0]</td>
</tr>
<tr>
<td>AN:OP</td>
<td>48</td>
<td>323.5</td>
<td>340.3</td>
<td>105.2</td>
<td>[11.0, 1836.0]</td>
</tr>
<tr>
<td>TN:TP</td>
<td>51</td>
<td>116.3</td>
<td>127.3</td>
<td>109.4</td>
<td>[10.0, 644.0]</td>
</tr>
<tr>
<td>Total Phytoplankton</td>
<td>51</td>
<td>381.9</td>
<td>1054.0</td>
<td>276</td>
<td>[0.0, 6638.0]</td>
</tr>
<tr>
<td>Total Zooplankton</td>
<td>51</td>
<td>36037.1</td>
<td>3.13E + 08</td>
<td>1.7E + 05</td>
<td>[0.0, 83401]</td>
</tr>
</tbody>
</table>

Figure 11 provides a graphical summary of the coefficient of variation (CV) of the physical water chemistry variables observed during the three seasonal groups. Of the five variables considered, only water clarity, reported as Secchi, had any noticeable difference in variance between the groups. Surface samples collected at zero (0) meters for water pH, temperature, specific conductance, and surface integrated samples (0 – 2.5 m) for total dissolved solids were virtually identical through the entire year and did not appear to vary considerably by season when the mean is adjusted by the standard deviation.
In Figure 12 the CV of chlorophyll α, PO₄-P, TP, and the nitrogen species, the ratios of available nitrogen to PO₄-P and TN to TP were compared. Clearly chlorophyll α exhibited large variability between the three seasonal groups. In contrast the nitrogen variables did not show considerable variability between the seasons. PO₄-P concentrations varied more during the winter than the other two seasons, and TP had a greater observed variability during the summer. Both AN:OP and TN:TP exhibited greater CVs during the summer months when primary production is the greatest.
Figure 12. Nutrient water chemistry C.V. (%) for Lake Las Vegas, Clark County, Nevada.

Relationship Analysis

In this study, the research design implemented a hierarchal approach to understand and explain the variables that influence plankton biomass. As stated previously in the methods, univariate correlation, stepwise autoregression and partial least squares (PLS) were used to characterize the relationships and to account for sample size, normality and autocorrelation.

Table 11 provides a summary of the multiple univariate correlation analyses conducted on four data sets that include winter, spring/fall, summer and all data collected from 1991-2003. In this test all variables are tested to determine if any positive or negative correlation relationships exist. Selection criteria were based on an $r \geq 0.50$ and a $p \leq 0.10$. As shown in Table 11, phosphorus did not consistently influence the presence of phytoplankton biomass, whereas conductance and total dissolved solids appeared to have a strong influence over the presence of cladocerans, rotifers, Chrysophyta and Chryptophyta. Nitrogen appears to have various positive or negative influences over both phytoplankton and zooplankton biomass.
Table 11. Correlation summary by season and all years for plankton and physical and chemical characteristics collected in Lake Las Vegas, Clark County, Nevada.

<table>
<thead>
<tr>
<th>Variable</th>
<th>Copepods</th>
<th>Cladocerans</th>
<th>Rotifers</th>
<th>Diatom</th>
<th>Chlorophyta</th>
<th>Chrysophyta</th>
<th>Chrypophyta</th>
<th>Cyanophyta</th>
<th>Pyrophyta</th>
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<td>1,4</td>
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<td></td>
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</tr>
</tbody>
</table>

Legend:
1 - Winter (December, January, February)  *
2 - Spring/Fall (March, April, May, October, November)  *
3 - Summer (June, July, August, September)  *
4 - 1991 – 003

* Bold items were found significant at p ≤ 0.10
* Selection criteria: r ≥ 0.50
* Direction of correlation: +

In an attempt to further summarize the variables considered in the correlation analysis, the same data groups were analyzed using stepwise autoregression. This model provided two valuable tools. First, this method tested for autocorrelation between the independent variables and, second allowed for the consideration of time on the dependent variables. Selection criteria for entry was p < 0.15 and stay criteria of p > 0.3 as suggested by Mickey and Greenland (1989).
As observed in both the correlation analysis both specific conductance and total dissolved solids influenced plankton biomass (Table 11 and Table 12). Chlorophyll a was added as an independent variable to this test to determine the relevance of this analytical procedures accuracy in estimating the phytoplankton biomass and providing a better understanding of the factors influencing the plankton. When evaluating the influence of the measured independent variables on the dependent variables in Table 12, it is apparent that annual cycling is more defined than seasonal cycling.

Based on the strong relationship between phosphorus and chlorophyll a, a sub-study was conducted to look at the effect of time on changes in lake water chemistry on chlorophyll a concentrations. In this study chlorophyll a was the dependent variable and a total of five time lags were considered. Each time lag represented a period of one (1) month of the independent variables water temperature, conductance, total dissolved solids, TP, TN and the TN:TP ratio. Only TP and PO4-P were the two independent variables that best predicted chlorophyll a concentrations (F3, 145 = 43.41, p < 0.0001). These results closely reflect the widely accepted influence of phosphorus on chlorophyll a. Figure 13 provides the annual TN:TP ratios for the reservoir during the study period against annual phytoplankton biomass. In general, phytoplankton biomass increases and decreases with respect to TN:TP increases and decreases. However, it is evident that there are other factors that contribute to phytoplankton biomass.

The influence of time was considered and was found important only out to one time lag (1 month), suggesting that phytoplankton biomass is only influenced by water quality additions or subtractions to the reservoir that occurred over the previous thirty to forty-five day period. This is consistent with the findings of Sommer (1993), who found that resource lags ranged from zero (0) to six (6) weeks.
Table 12. Stepwise autoregression summary showing which independent variables were most important for predicting plankton biomass.

<table>
<thead>
<tr>
<th>Independent Variable</th>
<th>Chlorophyll</th>
<th>Copepod</th>
<th>Cladoceran</th>
<th>Rotifers</th>
<th>Bacillariophyceae</th>
<th>Chlorophyta</th>
<th>Chrysophyta</th>
<th>Chryrophyta</th>
<th>Cyanophyta</th>
<th>Pyrrophyta</th>
<th>Haptophyta</th>
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<td>1,2,3</td>
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</tbody>
</table>

Legend:
1 – Winter (December, January, February)  
2 - Spring/Fall (March, April, May, October, November)  
3 – Summer (June, July, August, September)  
4 - 1991 -2003  

• Entry Criteria: p < 0.15  
• Stay Criteria: p > 0.3
Table 13 summarizes the results of three partial least squares models that were developed to evaluate the influence of various predictor variables on the biomass of the response variables.

The first model used eight water chemistry variables to predict or weight the importance of each to the response variable. In this model, the response variables were the six (6) phytoplankton divisions’ cumulative biomass. The model output is reflected in the variable of importance for predictions (VIP) values. Those predictor variables with a VIP value \( \geq 1.0 \) are considered of significance in predicting the response variables. In the case of the first model, TN, TN : TP, total dissolved solids, conductance, water temperature and pH strongly influenced phytoplankton biomass.

In the second model the same predictor variables were modeled against the total zooplankton biomass from the three (3) divisions and conductance, water temperature, total dissolved solids, chlorophyll \( a \) and pH VIP values were found to strongly predict the increase in zooplankton biomass. The nutrient predictor variables were not found to predict the abundance of zooplankton.
The third model explains the relationship between zooplankton and phytoplankton. In this model the phytoplankton divisions of Chlorophyta, Chrysophyta and Chrytrphyta predicted a positive change in zooplankton biomass. The division Chlorophyta (green algae) had the highest VIP value of 2.21. This is a positive observation that we would hope to observe, since this division is considered a high quality food source for zooplankton, especially cladocerans. As observed in Tables 5, 6, and 7, these divisions also represent the highest biomass observed.
Table 13. Partial Least Squares summary showing variables of importance for prediction (VIP) values for three models looking at relationship between phytoplankton, zooplankton, and water chemistry for Lake Las Vegas, Clark County, Nevada. Value of VIP > 1 are considered significant.

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<th>Model</th>
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<td>Zooplankton v. Chemistry</td>
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<td>Temperature</td>
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<td>Total Dissolved Solids</td>
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<td>pH</td>
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<td>TN:TP</td>
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</tr>
<tr>
<td>Response Variables</td>
<td>Copepods, Cladocerans, Daphnia, Rotifers</td>
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<td>Zooplankton v. Phytoplankton</td>
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INTRODUCTION
Discussion and Summary

As discussed in the literature review, there are many factors that can influence the distribution and assemblage of plankton. In this study, five research questions were developed to determine if various environmental measurements collected over a twelve-year period significantly influenced plankton assemblages, ontogeny and abundance.

**Question 1.** Is there a relationship between zooplankton assemblages and available phytoplankton assemblages as food sources?

**Question 2a.** Is there a relationship between annual changes in lake water phosphorus species, nitrogen species, nitrogen and phosphorus ratios, major ions, and pH and zooplankton assemblages?

**Question 2b.** What correlation is there between lake water phosphorus species, nitrogen species, nitrogen and phosphorus ratios, major ions, and pH and phytoplankton biomass and assemblages?

**Question 2c.** Do zooplankton and phytoplankton assemblages follow seasonal limnological trends?

**Question 3.** What effect do changing salinity (total dissolved solids) concentrations have on *Daphnia pulex* populations?

These research questions lend themselves to five discussion themes: 1) Influence of seasonality on plankton, 2) Key water quality factors influencing plankton, 3) Interactions between zooplankton and phytoplankton, and 4) Influence of seasonality, water quality, and plankton interactions on lake management.
Influence of Seasonality

Phytoplankton

The concept of seasonal influences on plankton is not a new concept in limnology. Hutchinson (1967) in his seminal work, *Treatise on Limnology*, was one of the first to synthesize the existing literature. Others like Sommer and Goldman devoted careers to studying the producer/consumer relationships of Lake Constance and Lake Tahoe (Goldman 1988), respectively. Hutchinson (1967) proposed three groups of factors that influence seasonal variation in phytoplankton populations and component species: independent physical factors (temperature, illumination, and turbulence), interdependent chemical factors (inorganic nutrients, accessory organic compounds, and antibiotics), and biological factors (parasitism, predation, and competition).

In this study we used monthly surface water temperatures as our field-measured characteristic to define the seasonal groupings in Lake Las Vegas (Figure 5). Hutchinson (1967) also used temperature to separate the seasons based on thermal maxima and minima. He also included illumination as a characteristic variable and defined his seasons as follows:

- **Winter** (January – February) as cold with relatively low light
- **Spring** (April – May) as cold with relatively high light
- **Summer** (July – August) as warm with relatively high light
- **Fall** (October – November) as warm with relatively low light

Hutchinson did not include the months of March, June and December. Illumination was not measured as part of this study and was not included as a defining variable.

Unlike lakes located in most temperate regions that have periods of ice cover, lake and reservoirs found in desert regions do not experience large seasonal fluctuations in temperatures and illuminations. Table 14 outlines the seasonal variation between those variables that the literature suggests can influence plankton assemblage and succession.

The chosen method to group the seasons by surface temperature did not yield a large variance (Table 14). The fact that summer average surface temperature was one (1) degree Celsius lower than the fall average would appear to defy logic. In future studies, I would recommend considering a group scenario that would divide the samples into periods of thermal stratification and non-stratification. In spite
of our classical limnology training and desire to replicate historic studies of temporal lakes, desert lakes do not experience thermal extremes and ice on phenomenon of most northern hemisphere lakes and reservoirs.

Table 14. Seasonal characteristics of Lake Las Vegas, Clark County, Nevada for the period of 1991 through 2003.

<table>
<thead>
<tr>
<th>Season Months</th>
<th>Winter (December – February)</th>
<th>Fall &amp; Spring (October – November to May)</th>
<th>Summer (June – September)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Surface Water Temperature °C (average, min., max.)</td>
<td>18, 7, 27</td>
<td>19, 8, 29</td>
<td>18, 8, 28</td>
</tr>
<tr>
<td>Illumination (% sunny days) (1)</td>
<td>Low (79%)</td>
<td>Medium (85%)</td>
<td>High (95%)</td>
</tr>
<tr>
<td>Chlorophyll a µg/L (average, min., max.)</td>
<td>5, 1, 3</td>
<td>3, 1, 20</td>
<td>8, 1, 183</td>
</tr>
<tr>
<td>TN:TP (average, min., max.)</td>
<td>90, 9, 235</td>
<td>89, 10, 354</td>
<td>116, 10, 644</td>
</tr>
</tbody>
</table>

1 - Data were obtained from the National Weather Service’s Las Vegas Climate Book

Sommer (1981) studied the impact of repetitive nutrient cycle on phytoplankton populations in Lake Constance. Sommer (1985) and Sterner (1989) later explained that nutrient availability varied with seasonal change. Increased nutrient availability was often found in the fall following thermal destratification, and remained available for biomass production through the spring. After lake stratification in the spring, nutrient availability declines and the epilimnion is depleted of biologically available nutrients by plankton and fish. They found that each season has a repeated succession of phytoplankton species that may be observed from year-to-year. Pioneering species that exhibit population growth traits that follow the characteristics of r-strategists dominate the plankton after fall lake destratification. These species are small in size, have high reproductive rates, can rapidly assimilate the readily available nutrients, reach maturity quickly, and their populations can crash and reestablish quickly. Fall and winter populations are often dominated by species found within the divisions of Cryptophyta, Chlorophyta, Chrysophyta, Pyrophyta, and to a lesser extent the Bacillariophyceae. As water temperatures and illumination increase in the spring, phytoplankton populations shift towards diatom dominance due to their ability to out compete other species (Hutchinson 1967). Bluegreen algae (Cyanophyta) also dominate during this season due to their ability to
shade the photic zone and to reduce the growth of other pelagic species. Following the spring maximum plankton (immediately following thermal stratification), a period of low nutrients and high visibility is observed (Sommer et al. 1986). Bacillariophyceae population declines and random dominance by species found in the divisions of Pyrrophyta, Chlorophyta, and Cyanophyta are observed during the summer months (Sommer 1981). Often the species are colonial and reflect a more advanced cellular structure. These species tend to exhibit population growth characteristics of a k-strategist. Cells are typically larger, and they are less mobile, and adapted to survival in an environment that may be nutrient limited because of their ability to store resources for future needs (Sommer 1981).

In Lake Las Vegas similar nutrient patterns of phytoplankton response were observed. A total of 219 species were identified between 1991 and 2003 of which 94 species were observed during the summer months, 77 species were observed during the spring/fall months and 48 species were found during the winter months. Figure 8 illustrates that the Chlorophyta dominated all seasonal groups. Bacillariophyceae, Chrysophyta, and Chryptophyta were commonly found during the winter, while Cyanophyta, Chryptophyta, and Pyrrophyta were present during the spring and fall months. Bacillariophyceae represented a large portion of the summer months along with all other divisions which were equally represented to a lesser extent.

In consideration of Sommer’s (1981) findings related to the presence of r-strategist in the fall, winter, and spring and k-strategist in the summer, individual species GALD measurements were collected and analyzed. Based on the finding reported in Table 4 and Figure 10, the use of GALD measurements to predict the seasonal species assemblages is not a consistent predictor of species presence or absence. Unlike Sommer’s work on Lake Constance, Lake Las Vegas is a relatively young reservoir located in an urban environment, while Lake Constance is a very old, established lake. It is for this reason that we must use caution when attempting to draw linear comparisons between different lake types and use great caution when making comparative predictions based on their associated phytoplankton communities. We must consider what species are present in each of these groups and their physiologic characteristics. Many species that we would consider k-strategist have the ability to conserve and store resources for future periods of resource deficiency. In Sommer’s work he categorized the dinoflagelletes as summer dominant species, while in Lake Las Vegas many large bodied dinoflagelletes can contribute a major portion of the biomass and abundance during the winter (Table 5).
Two phenomena have a biased influence on the effect of seasonality in this study. First, the disproportional influence of *Pyramidichlamys dissecta* blooms during 2000 through 2003 and second, the influence of natural (floods) and anthropogenic (fertilization) related events on the reservoir will be discussed in the water quality discussion.

In Lake Las Vegas, two events can be linked to the *Pyramidichlamys dissecta*. This is not to say that other factors did not contribute. During the winter of 1999-2000, Lake Las Vegas experienced higher than normal water temperature by three to four degrees Celsius. In addition, Lake Las Vegas was in the middle of a dredging project that removed approximately 2,485,000 m$^3$ of sediment from the shallow west end of the reservoir. In spite of the installation of a surface-to-sediment silt curtain, the combination of hydraulic dredging and mechanical removal by a barge mounted backhoe, a large amount of sediment stored nutrients were released. Upon completion, the silt curtain was removed and an immediate response from the phytoplankton community was observed due to the nutrient influx. The reservoir assimilated these nutrients back into biomass and sediment stores over the following two year period.

An additional concept that should be mentioned is that the reservoir is regularly subjected to the introduction of new species from the reservoirs two water sources, Lake Mead and storm water runoff from the Las Vegas Valley. During the period of high biomass of *Pyramidichlamys dissecta* in Lake Las Vegas, *Pyramidichlamys dissecta* was very abundant in Lake Mead. Owing to a short historical record, this trend cannot be substantiated, but cursory observations have shown that phytoplankton assemblages and dominance changed when large volumes of water were introduced into the reservoir from these two sources (Table 3).

### Zooplankton

When considering the seasonal nature of zooplankton assemblages one must recognize that many of the factors that influence phytoplankton populations and their succession also have a profound influence on zooplankton. The producer/consumer relationship that occurs between the plankton is easily linked to previous discussions. Phytoplankton serves as the primary prey for filter-feeding zooplankton, but not all phytoplankton are equally desirable as a food source for zooplankton. Many zooplankton are filter feeders, daphnids being an example, and limited to prey that can be ingested. Large-celled and colonial species of phytoplankton, often associated with bluegreen algae and dinoflagellates, are difficult for filter feeders to consume.
Additionally, we need to take into account the other factors that influence zooplankton populations such as zooplankton competition, predation by fish or other zooplankton (Lazzaro 1987; Northcote 1988) and allogenic factors influencing zooplankton biomass and succession (Morris 1980). This study did not focus on species level changes, but divisional changes by those environmental variables measured during the study period. Berman et al. (1995) found a steady decline in zooplankton (mainly cladocerans and copepods) owing to increased predation by fish. Zooplankton predation by fish predation can have an adverse effect on the size and distribution of zooplankton species. These changes often manifest themselves in a change in the abundance of phytoplankton that are not easily assimilated into higher trophic level biomass (Lazzaro 1987; Northcote 1988). In Lake Las Vegas, zooplankton populations tend to follow changes in phytoplankton populations. Care was taken when the reservoir was initially stocked with fish to avoid the selection of species that favored zooplankton as their primary food source.

Interaction Among Plankton

In making the comparison between zooplankton and phytoplankton biomass, we must recognize that the sampling protocol for each were different. Zooplankton were collected by means of 0 – 15m vertical tow and phytoplankton were collected from the upper epilimnion (0 – 2.5m) by an integrated sampler. An argument can be make that the discrepancies in sampling method has created an apples to oranges comparison, however a number of other variables were not controlled either. This study did not take spatial variability into account or the influence of daily vertical migration of plankton. The assumptions adopted in this study accepted these short comings since samples were collected consistently over time and from the same location for the duration of the sampling period.

In an attempt to determine what zooplankton and phytoplankton relationships may exist between the major divisions, univariate correlation, autogression and partial least squares regression were used to evaluate the data. Each statistical method served to isolate the variables that had a higher level of influence on the predictor (dependent) variables. Three conditions will be discussed in this section, zooplankton relationships, phytoplankton relationships, and zooplankton and plankton relationships.
Zooplankton Relationships

A number of observed relationships were found among the zooplankton divisions. Copepods were commonly present when cladocerans and rotifers were present, but if rotifers were present cladocerans were not (Table 15). This can be explained by the fact that cladoceran populations typically decline during the summer months due to food availability and rotifers and copepods are able to survive on the phytoplankton present during the summer months. When rotifers were present, their overall abundance and biomass was quite low. One possible explanation for this is that rotifer populations may have been underestimated due to the use of an 80 μm tow net. Many of the rotifer species are smaller than 80 μm.

Table 15. Summary of zooplankton relationships in Lake Las Vegas, Clark County, Nevada during 1999 through 2003.

<table>
<thead>
<tr>
<th>Independent Variables</th>
<th>Copepods</th>
<th>Cladocerans</th>
<th>Rotifers</th>
</tr>
</thead>
<tbody>
<tr>
<td>Copepods</td>
<td></td>
<td>C</td>
<td></td>
</tr>
<tr>
<td>Cladocerans</td>
<td></td>
<td>C,R</td>
<td></td>
</tr>
<tr>
<td>Rotifers</td>
<td></td>
<td>C</td>
<td></td>
</tr>
</tbody>
</table>

C = Correlation, R = Stepwise Regression

Phytoplankton Relationships

When Bacillariophyceae biomass was measurable Chlorophyta, and Chytrophyta were present. In general, phytoplankton species of the divisions Chlorophyta, Chrysophyta, and Chytrophyta were most frequently observed in the reservoir. Table 16 presents a summary of phytoplankton intra-relationships found in Tables 11, 12, and 13. It must be noted that these results should be interpreted cautiously, as correlative relationships mask the importance of physiological difference among these species.
Table 16. Summary of phytoplankton biomass relationships in Lake Las Vegas, Clark County, Nevada during the period 1991 through 2003.

<table>
<thead>
<tr>
<th>Independent Variables</th>
<th>Bacillariophyceae</th>
<th>Chlorophyta</th>
<th>Chrysophyta</th>
<th>Cryptophyta</th>
<th>Cyanophyta</th>
<th>Pyrophyta</th>
<th>Haptophyta</th>
</tr>
</thead>
<tbody>
<tr>
<td>Bacillariophyceae</td>
<td>C,R</td>
<td>C</td>
<td>C,R</td>
<td>C,R</td>
<td>R</td>
<td>R</td>
<td>C,R</td>
</tr>
<tr>
<td>Chrysophyta</td>
<td>C</td>
<td>C,R</td>
<td>C,R</td>
<td>C,R</td>
<td>C</td>
<td>R</td>
<td>R</td>
</tr>
<tr>
<td>Cryptophyta</td>
<td>C,R</td>
<td>C</td>
<td>C,R</td>
<td>C,R</td>
<td>C,R</td>
<td>R</td>
<td>R</td>
</tr>
<tr>
<td>Cyanophyta</td>
<td>R</td>
<td>C,R</td>
<td>C,R</td>
<td>C,R</td>
<td>C,R</td>
<td>R</td>
<td>R</td>
</tr>
<tr>
<td>Pyrophyta</td>
<td>R</td>
<td>C</td>
<td>C</td>
<td>C</td>
<td>C,R</td>
<td>R</td>
<td>R</td>
</tr>
<tr>
<td>Haptophyta</td>
<td>C,R</td>
<td>R</td>
<td>R</td>
<td>R</td>
<td>R</td>
<td>R</td>
<td>R</td>
</tr>
</tbody>
</table>

C = Correlation, R = Stepwise Regression

**Plankton Relationships**

Table 17 presents a summary of those relationships between zooplankton and phytoplankton that contribute to increased zooplankton biomass. Partial least squares was used to determine if a multivariate relationship exists and the model predicts that the divisions of Chlorophyta, Chrysophyta, and Cryptophyta have the greatest likelihood to influence zooplankton biomass. It is apparent that the cladocerans have the best chance of population prediction by the presence of Bacillariophyceae, Chrysophytes, and Cryptophytes. Rotifer biomass was most related to changes in phytoplankton biomass in the divisions of Chlorophyta, Cryptophyta and Haptophyta. Copepods were the least influenced by phytoplankton, but were related to the presence of those species in the division Chrysophyta. This observation may be attributed to the fact the copepod species observed are not herbivorous. In a study of this type where landscape level changes are of interest, it is my opinion that the use of partial least squares is desirable. PLS is very robust and corrects for multicollinearity and autocorrelation.
Table 17. Summary of zooplankton and phytoplankton biomass relationships in Lake Las Vegas, Clark County, Nevada during the period 1991 through 2003.

<table>
<thead>
<tr>
<th>Group</th>
<th>Independent Variables</th>
<th>All Zooplankton</th>
<th>Copepods</th>
<th>Cladocerans</th>
<th>Rotifers</th>
</tr>
</thead>
<tbody>
<tr>
<td>Phytoplankton</td>
<td>Bacillariophyceae</td>
<td>P</td>
<td></td>
<td>(C),R</td>
<td>(C)</td>
</tr>
<tr>
<td></td>
<td>Chlorophyta</td>
<td>P</td>
<td>R</td>
<td>(C),R</td>
<td>R</td>
</tr>
<tr>
<td></td>
<td>Chrysophyta</td>
<td>P</td>
<td>R</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Cryptophyta</td>
<td>P</td>
<td>R</td>
<td></td>
<td>R</td>
</tr>
<tr>
<td></td>
<td>Cyanophyta</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Pryrophyta</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Haptophyta</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

C = Correlation, R = Stepwise Regression, P = Partial Least Squares; ( ) = negative relationship

Key Water Quality Factors

Water chemistry within Lake Las Vegas is influenced by a number of sources both natural and anthropogenic. Natural influences such as flood events that periodically discharge into the reservoir have changed water chemistry and plankton communities in a very short period of time. Depending on the magnitude of these events, the observed change may be very short lived, or the effect may be observed for years when the lake’s total volume is completely replaced or flushed, as in the July 1999 flood event. Table 3 provided a chronology of those events that influenced the reservoir’s plankton populations. Clearly, the addition of nutrients by either flood events or accidental fertilizer runoff from the adjacent golf course has been documented (Table 3) and their effect observed in plankton response both visually and statistically during the course of this research study. In both the areas of lake and golf course management, measures have been put in place to reduce the likelihood of future fertilizer runoff by means of “Best Management Practices”.

Zooplankton

Correlation analysis identified that a number of relationships exist between reservoir water chemistry and plankton biomass and frequency. The most significant is that rotifers are very susceptible to changes in salinity ($r \geq 0.5$, $p \leq 0.1$) when conductance and total dissolved solids increased rotifer biomass decreased. This is observed in Figure 7 where rotifer relative frequency declined from 1994 through 2000 and began to recover from 2001 through 2002 after large volumes of low total dissolved solids storm water entered the reservoir during the summer months when rotifers are common. One precautionary note must be made, that in general rotifer biomass was likely underestimated in this study due to sampling methods.
The use of an 80 µm plankton tow net may have precluded the capture of a number of rotifers, since some species are smaller than the mesh opening of the nets. In future studies this should be considered in sampling protocol that should include grab samples. Owing to the fact that the data collected in this study are part of a long-term monitoring program, changes in sampling methodology are not favorable.

A number of zooplankton relationships were identified and are summarized in Table 18 which represents a summary of the results presented in Tables 11, 12, and 13. A number of water chemistry relationships were observed for each of the three zooplankton divisions. Changes in Rotifers biomass were related to ammonia, conductance, total dissolved solids, TN:TP, and chlorophyll $a$. All of these variables had a positive influence on biomass, but conductance and total dissolved solids had a significant negative impact on rotifer biomass ($r \geq 0.5$, $p \leq 0.1$). Copepod biomass was influenced by changes in pH, water temperature, Secchi depth, TP, NO$_2$-N+NO$_3$-N, conductance, and chlorophyll $a$. As observed with the rotifers, copepods also had a negative relationship with increases in conductance of lake water. Cladoceran biomass, in contrast to rotifers and copepods, had a positive response to increases in lake water conductance. In addition, cladocerans were positively influenced by total dissolved solids, PO$_4$-P, NO$_2$-N+NO$_3$-N, NH$_4$-N, TKN, and TN.

When looking at zooplankton total biomass, partial least squares analysis revealed that conductivity, water temperature, total dissolved solids, chlorophyll $a$ and pH are all significant (VIP > 1) in the prediction of zooplankton biomass.
Table 18. Summary of water chemistry relationships to zooplankton biomass in Lake Las Vegas, Clark County, Nevada during the period 1991 through 2003.

<table>
<thead>
<tr>
<th>Group</th>
<th>Independent Variables</th>
<th>All Zooplankton</th>
<th>Copepods</th>
<th>Cladocerans</th>
<th>Rotifers</th>
</tr>
</thead>
<tbody>
<tr>
<td>Physical</td>
<td>pH</td>
<td>P</td>
<td>P</td>
<td>(C), R</td>
<td>(C), R</td>
</tr>
<tr>
<td></td>
<td>Temperature</td>
<td>P</td>
<td>C</td>
<td>C, R</td>
<td>(C), R</td>
</tr>
<tr>
<td></td>
<td>Secchi</td>
<td>P</td>
<td>(C)</td>
<td>C, R</td>
<td>(C), R</td>
</tr>
<tr>
<td></td>
<td>Conductance</td>
<td>P</td>
<td>(C), R</td>
<td>C, R</td>
<td>(C), R</td>
</tr>
<tr>
<td></td>
<td>T. Dissolved Solids</td>
<td>P</td>
<td>(C), R</td>
<td>C, R</td>
<td>(C), R</td>
</tr>
<tr>
<td>Chemistry</td>
<td>O. Phosphorus</td>
<td></td>
<td>C</td>
<td>C</td>
<td></td>
</tr>
<tr>
<td></td>
<td>T. Phosphorus</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>NO₂-N+NO₃-N</td>
<td>C</td>
<td>C</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>NH₄-N</td>
<td></td>
<td></td>
<td></td>
<td>(C)</td>
</tr>
<tr>
<td></td>
<td>T. Kjeldahl-N</td>
<td></td>
<td></td>
<td></td>
<td>(C)</td>
</tr>
<tr>
<td></td>
<td>T. Nitrogen</td>
<td>C</td>
<td>C</td>
<td></td>
<td>R</td>
</tr>
<tr>
<td></td>
<td>TN:TP</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

| Temporal       | Time                  |                 |          | R           |          |

C = Correlation, R = Stepwise Regression, P = Partial Least Squares; ( ) = negative relationship

Phytoplankton

Table 19 summarizes the relationships that were observed for this study. It is a summary of Tables 11, 12, and 13 found in the results section of this dissertation. Bacillariophyceae biomass was related to changes in water temperature, total dissolved solids and time. Changes in Chlorophyta biomass were positively related to pH, Secchi depth, NH₄-N, nitrogen and total kjeldahl nitrogen. Changes in Chrysophyta biomass were found related to Secchi depth, conductance, and total dissolved solids. Chyrtophyta biomass increased with increases in lake water conductance and total dissolved solids. Cyanophyta biomass changes related to pH, Secchi depth, total kjeldahl nitrogen and TN. Haptophyta biomass related to pH, temperature, Secchi depth, NH₄-N, total kjeldahl nitrogen, and TN.

When considering what variables have the most influence over total phytoplankton biomass, TN, TN:TP, total dissolved solids, conductivity, and temperature were found most significant using partial least squares analysis. The fact that the correlation analysis and regression analysis did not find a strong relationship between phytoplankton biomass and TP and PO₄-P directly was not expected. In many lakes and reservoirs phosphorus is limiting and sources may be limited to internal loading from sediment stores.
One possible explanation is that phosphorus by itself is not limiting phytoplankton biomass. When considered in combination with nitrogen as a ratio, phosphorus is found to have an impact. Additionally, based on the finding expressed in Table 19, physical characteristics of lake water, such as salinity, are as important in predicting changes in phytoplankton biomass in Lake Las Vegas.

Table 19. Summary of water chemistry relationships to phytoplankton biomass in Lake Las Vegas, Clark County, Nevada during the period 1991 through 2003.

<table>
<thead>
<tr>
<th>Group</th>
<th>Independent Variables</th>
<th>All Phytoplankton</th>
<th>Bacillariophyceae</th>
<th>Chlorophyta</th>
<th>Chrysophyta</th>
<th>Chlrophyta</th>
<th>Cyanophyta</th>
<th>Pyrophyta</th>
<th>Haptophyta</th>
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</thead>
<tbody>
<tr>
<td>Physical</td>
<td>pH</td>
<td></td>
<td>C</td>
<td>C</td>
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<td>Temperature</td>
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<td>Conductance</td>
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<td>T. Dissolved Solids</td>
<td>P</td>
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<td></td>
<td></td>
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<tr>
<td>Chemistry</td>
<td>O. Phosphorus</td>
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<td>R</td>
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<td></td>
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<td></td>
<td>T. Phosphorus</td>
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<td></td>
<td>NO₂-N+NO₃-N</td>
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<td>NH₄-N</td>
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<td></td>
<td>(C)</td>
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<tr>
<td></td>
<td>T. Kjeldahl-N</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>(C)</td>
<td></td>
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<tr>
<td></td>
<td>T. Nitrogen</td>
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<td></td>
<td>TN:TP</td>
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<td></td>
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<tr>
<td>Temporal</td>
<td>Time</td>
<td></td>
<td>R</td>
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<td></td>
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<td></td>
<td></td>
</tr>
</tbody>
</table>

C = Correlation, R = Stepwise Regression, P = Partial Least Squares; ( ) = negative relationship

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Management Implications

The principal focus of this study was to consider the birth of a new small man-made urban lake and provide the reservoir's owners with an understanding of how to improve upon existing lake management strategies. Lake Las Vegas presents a unique opportunity for scientists to study the changes in the reservoir's limnology over time because of the unique foresight of the project planners, regulators and managers. Sampling protocols were established by government regulators and engineers to show that the reservoir would not have negative impacts on Lake Mead, and were not necessarily designed to provide management with an accurate account of all the reservoir's processes. An example of this is the decision only to collect water quality samples at depth on a quarterly basis. Integrated depth samples are regularly collected and in many cases are sufficient during the non-stratified periods to provide an understanding of water column interactions; but there are times during the stratified period that more detailed water quality data would be helpful in understanding the distribution and assemblage of phytoplankton species. At present, a total of thirty-eight to forty samples are collected annually and intervals vary from weekly to monthly. It is my recommendation that sampling frequency should change to bi-weekly regardless of thermal stratification and the number of sampling locations reduced from four to two.

In this analysis of the twelve (12) year data set, data from only two of the four locations were used, since not all variables were measured at every site. I would recommend that the number of sampling stations be reduced from four (4) to two (2), eliminating stations at LLV-1A and LLV-2. This would leave one station at the deep end of the reservoir (LLV-1) and one station at the shallow West end (LLV-3). It is quite suspect that statistical independence is a problem if we were to compare the data collected at all of the stations. At each location three samples should be collected: an integrated surface (0 - 2.5 m), a discrete sample from within the metalimnion, and a sample from the hypolimnion (near the bottom) and all water quality variables identified in Table 2 would be analyzed. Of these variables currently measured or analyzed for in the long term monitoring program at Lake Las Vegas (Table 2). I would recommend the following changes: continue to sample major cations and anions quarterly and sample total dissolved solids, chlorophyll a, TP, TN, zooplankton and phytoplankton every two weeks as these changes would result in approximately the same cost, but would result in a sampling design that provides a greater level of detail to better understand plankton response and, in turn, overall health of the reservoir.

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Studying plankton populations in the reservoir provides a measure of ecosystem trophic state and overall health; this revised sampling program would allow a better understanding of lake management influencing factors for fishery management, storm water management, and future dredging activities. The reservoir is actively stocked with *Pimephales promelas* (fathead minnow), *Oncorhynchus mykiss* (rainbow trout), *Lepomis macrochirus* (bluegill), and *Procambarus clarkia* (swamp crayfish) annually in an attempt to manage a trophy *Micropterus salmoides* (largemouth bass) fishery. The proposed revised sampling protocol would allow for better measurements of biological interactions to be determined between stockings influence on plankton biomass and resulting aesthetics. Large amounts of storm water containing increased nutrients and salinity levels have entered the reservoir on a somewhat regular frequency (Table 3). Active management of flood events is not a proactive strategy for the reservoir since the water quality of this resource is regulated and is the responsibility of regional agencies. It has been, and continues to be, the intent of Resort management to work with these agencies to support solutions to improve storm water quality within the watershed. In Lake Las Vegas, phosphorus is limiting, but periodic flood events provide a new source of phosphorus to react with the nitrogen rich waters of the reservoir. This phenomenon is well documented and the resulting change in TN:TP ratios encourages the propagation of phytoplankton as can be observed in Figure 13.

Changes in nutrient concentrations in the reservoir should be documented and monitored for subsequent changes in plankton dominance and assemblage. This study provides evidence that the ratio of nitrogen:phosphorus is a useful metric that will help understand plankton dominance shifts.

Dredging activities within the reservoir have the potential to have a significant impact on lake water quality and biological activities. It is recommended that detailed water quality monitoring program be implemented during the period immediately proceeding further dredging, during, and after dredging operations. Unfortunately plankton sampling during the previous dredging was not adequate to identify the daily changes that occurred. It is suspected that dredging activities contributed to the bloom of *Pyramichlamys dissecta*, but statistical verification cannot be made due to lack of data.

The use of Best Management Practices, such as the use of silt containers will provide adequate desilting prior to returning processed water to the reservoir. It is critical that segregation of the job site be maintained and that any colloidal material should be allowed to settle prior to disturbing the silt curtain.
It is recommended that the silt curtain remain in place until such time that TP concentrations are at or below twenty (20) µg/L, Secchi disk readings are ≥ two (2) meters, and chlorophyll a concentrations are ≤ ten (10) µg/L.

Additionally, I would recommend that future dredging activities be confined to the thermally non-stratified months (winter). This would promote winter phytoplankton blooms, but would possibly reduce the chances of noxious cyanophyta blooms if dredging activities ceased prior to summer stratification.

Summary of Findings

Zooplankton
1. Changes in zooplankton biomass are associated with specific conductance, total dissolved solids, water temperature, chlorophyll a, and pH (Table 13).
2. Zooplankton assemblages and frequencies are seasonally dependent. Their relationship is related to the phytoplankton biomass.
3. Phytoplankton divisions of Chlorophyta, Chrysophyta, and Chrypophyta have the greatest influence on zooplankton biomass.
4. Increases in specific conductance and total dissolved solids have a positive association with cladoceran biomass, and in the case of Lake Las Vegas, the cladocerans are represented primarily by *Daphnia pulex* (Table 11).

Phytoplankton
1. Changes in phytoplankton biomass are associated with TN, TN:TP, total dissolved solids, conductivity, and water temperature (Table 13 and Figure 13).
2. Phytoplankton assemblages and biomass are seasonally dependent. During the fall, winter, and spring are dominated by small, fast growing r-strategist phytoplankton, while the summer months are dominated by large slow growing K-strategists (Tables 5, 6 and 7).
3. *Pyramachlamys dissecta* populations were unusually high during the years of 2000-2002 and biased the results. These species out competed all other species during this period (Table 4).
Management

1. Revise lake sampling protocols to better monitor lake turnover, depth variability and plankton presence or absence.

2. Develop guidelines for future dredging activities that minimize detrimental effects on lake chemistry and plankton populations.

3. Develop guidelines for storm events that minimize detrimental effects on lake chemistry and plankton populations.

4. Measure plankton population response to fishery stocking events.
APPENDIX I

Descriptive statistics for lake water characteristics collected in Lake Las Vegas, Clark County, Nevada for the period of 1991 through 2003.
Descriptive statistics for lake water characteristics collected in Lake Las Vegas, Clark County, Nevada for the period of 1991 through 2003.

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Descriptive statistics for lake water characteristics collected in Lake Las Vegas, Clark County, Nevada for the period of 1991 through 2003. (continued)

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|                   | 1994 | 12  | 16   | 4  | 27.4  | [10, 26]|
|                   | 1995 | 12  | 22   | 13 | 61.6  | [5, 50] |
|                   | 1996 | 12  | 13   | 2  | 16.2  | [9, 16] |
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|                   | 1998 | 12  | 23   | 11 | 48.1  | [10, 45]|
|                   | 1999 | 12  | 20   | 11 | 56.7  | [8, 50] |
|                   | 2000 | 12  | 20   | 12 | 58.2  | [8, 49] |
|                   | 2001 | 12  | 39   | 30 | 76.1  | [21, 114]| |
|                   | 2002 | 12  | 21   | 11 | 53.9  | [11, 41]|
|                   | 2003 | 12  | 10   | 3  | 32.7  | [5, 14] |
|                   | 1991-2003 | 150 | 20  | 14 | 70.2  | [5, 114]|

| NO₂-N+NO₃-N (µg/L) | 1991 | 6   | 298  | 65 | 21.7  | [204, 374]|
|                   | 1992 | 12  | 606  | 248| 41.0  | [225, 957]|
|                   | 1993 | 12  | 274  | 233| 85.2  | [13, 637]|
|                   | 1994 | 12  | 44   | 29 | 64.8  | [6, 95]  |
|                   | 1995 | 12  | 632  | 179| 28.3  | [385, 890]|
|                   | 1996 | 12  | 314  | 71 | 22.4  | [224, 412]|
|                   | 1997 | 12  | 457  | 75 | 16.4  | [352, 555]|
|                   | 1998 | 12  | 1039 | 264| 25.4  | [643, 1498]|
|                   | 1999 | 12  | 1418 | 222| 15.7  | [1130, 1822]|
|                   | 2000 | 12  | 1089 | 275| 25.2  | [722, 1505]|
|                   | 2001 | 12  | 621  | 366| 59.0  | [167, 1193]|
|                   | 2002 | 12  | 581  | 394| 67.8  | [44, 1183]|
|                   | 2003 | 12  | 1275 | 306| 24.0  | [820, 1680]|
|                   | 1991-2003 | 150 | 680 | 468| 68.8  | [6, 1822]|

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Descriptive statistics for lake water characteristics collected in Lake Las Vegas, Clark County, Nevada for the period of 1991 through 2003. (continued)

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Nygaard, G. 1949. Hydrobiological studies of some Danish ponds and lakes II. The quotient hypothesis and some new or little known phytoplankton organisms. Kgl.


Wright, J.C. 1958. The limnology of Canyon Ferry Reservoir. 1. phytoplankton- zooplankton relationships in the euphotic zone during September and October, 1956. Limnology and Oceanogr. 3 (2) pp. 150-159.

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Publications:


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Committee Member, Dr. Jeff Janik, Ph.D.
Graduate Faculty Representative, Dr. Chad Cross, Ph.D.