TROPHIC CONDITIONS AND NUTRIENT LIMITATIONS IN THE HEADWATERS OF BEAVER LAKE, ARKANSAS, DURING A DRY HYDROLOGIC YEAR, 2005-2006

TROPHIC CONDITIONS AND NUTRIENT LIMITATIONS IN THE HEADWATERS OF BEAVER LAKE, ARKANSAS, DURING A DRY HYDROLOGIC YEAR, 2005-2006

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ABSTRACT

Algal nutrient limitation and trophic state were evaluated in Beaver Lake, AR from 2005 to 2006. Longitudinal Nitrogen and Phosphorus concentration gradients were observed, and the greatest concentrations were in the riverine zone. Vertical patterns were observed, were SRP showed greater concentrations in the hypolimnion. Chlorophyll-*a* concentrations showed a positive and significant linear relationship with TP and TN. The comparison of historic water quality data showed that in Beaver Lake Chl-*a* and TP concentrations have increased in the past 25 years and the trophic status of the reservoir was eutrophic for the riverine and transitional zone. The increase in Ch-*a* and TP may be attributed to a very dry season and low water levels in the reservoir. Nutrient enrichment bioassays suggest that Beaver Lake nutrient limitations shift from either N or P limitations to co-limitation.

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CHAPTER 1

INTRODUCTION

Lakes are essentially a collecting basin, with biological, chemical and physical properties, and qualities very different from those of rivers. A large diversity of aquatic life exists in lakes in a dynamic equilibrium; therefore lakes can be vulnerable to alterations. Elevated inputs of nutrients (especially nitrogen (N) and phosphorus (P)) can disrupt the equilibrium of the ecosystem causing the excessive growth of algae and aquatic weeds (USEPA, 2006).

The European Commission (EC) Urban Wastewater Treatment (UWWT) Directive defines eutrophication as "the enrichment of water by nutrients, especially nitrogen and/or phosphorus, causing an accelerated growth of algae and higher forms of plant life to produce an undesirable disturbance to the balance of organisms present in the water and to the quality of water concerned". Eutrophication causes many problems, such as loss of water clarity due to increased primary production, dissolved oxygen depletion the in hypolimnion, loss of aesthetical appeal and accessibility, and increased cost of treating drinking water (Andersen et al., 2005).

Lakes can be classified based on their trophic state which is defined as the amount of living biological material (biomass) in a water body at a specific location and time or its degree of eutrophication (Carlson, 1977). This classification includes three classes as defined by Carlson (1977):

Oligotrophic: Lakes in which plant growth is limited by low chemical concentrations and the variety of species is high but the number is low; these lakes are usually large, deep, and clear with high concentrations of dissolved oxygen.

Mesotrophic: These lakes are at an in-between trophic stage, supporting a diverse community of aquatic plant and animal life; most of the popular lakes are at this point of evolution.

Eutrophic: Lakes which have large supplies of nutrients from the catchment and bottom sediments, and elevated nutrients result in excessive growth of algae; these lakes have reduced diversity in organisms and low concentrations of dissolved oxygen in the bottom waters.

Lakes and reservoirs have structural differences and functional similarities which are important to understand. Reservoirs are man-made ecosystems created especially for water storage (e.g. flood control, water supply, irrigation, hydroelectric generation) in areas where large natural lakes are not adequate or do not exist (Thornton, 1990). The areas where reservoirs are formed are generally warmer than those where natural lakes exist, resulting in greater than average water temperature, longer growing seasons, and precipitation inputs that are closely balanced by, or less than, evaporative losses. Reservoirs have a dendritic, narrow, and elongated morphometry because they are generally formed in river valleys (Wetzel, 1990).

Reservoirs can be considered as very dynamic lakes and sometimes with biological processes similar to those in rivers. Reservoirs have greater sediment loading with more fine particles, and greater nutrient loading than lakes. Therefore, these physical characteristics determine the biological processes, especially those depending on nutrients availability and light (Wetzel, 1990). The catchment of reservoirs, which are near the mouth of the drainage basin, is often much larger than those in lakes (generally in the upper part portion of a drainage basin) (Wetzel, 1990). In addition, reservoirs have irregular water level fluctuations as a result of inflow characteristics, land use practices, channelization of primary inflows, flood control, and large and irregular water withdrawals (Wetzel, 1990).

Three zones can be identified in reservoirs and each of these has unique physical, chemical, and biological characteristics. According to Thornton (1990) and Kimmel et al. (1990) these areas and their main characteristics are:

Riverine Zone: this is a lotic environment with greater flow velocities, shorter residence time, and more available nutrients, and suspended solids. This zone is, also, characterized by high turbidity which decreases light penetration which may limit the photosynthesis process.

However, phytoplankton productivity per volume of the photic zone can be high.

Transitional Zone: in this zone the flow velocity decreases, residence time increases, silts, coarse-to-medium clays, and fine organic particulate matter settle to the reservoir bottom. Biological processes produce dissolved oxygen depletion, anaerobic conditions, and redox reactions creating anoxic conditions. This occurs during stratification when the hypolimnetic volume is small. The anoxic conditions will produce the resolubilization (especially iron, manganese, and phosphorus) adsorbed by particulate matter; increasing the concentration of dissolved constituents. This zone can be considered the most productive (high phytoplankton production) in the reservoir due to light and nutrient availability.

Lacustrine zone: this is the lower part of the reservoir, sediment is composed by fine particles (colloidal material and clay), longer residence time, lower concentration of dissolved nutrients, and a deeper photic layer (higher water transparency). Anoxic conditions develop near the dam and this conditions permit the release elements, such as iron, manganese, and phosphorus, from bottom sediments. The primary productivity per volume is also less in the photic zone.

PROBLEM STATEMENT

The Arkansas Natural Resources Commission (ANRC), formerly Arkansas Soil and Water Conservation Commission (ASWCC), has classified Beaver Lake as a priority for the implementation of watershed restoration, since Beaver Lake is the main source of drinking water for four cities and surrounding smaller cities in Northwest Arkansas (ASWCC, 2002). Beaver Lake Watershed is experiencing population growth and land use changes, which can increase the transport of sediments, nutrients (P, N, and C), and other constituents into the reservoir. The additional nutrients (particularly N and P) can potentially increase algal growth in Beaver Lake. Thus, it is important to understand the trophic gradients and the limits to manage Beaver Lake and its watershed. In particular, there is a need to determine what the trophic gradients are in the headwaters of Beaver Lake and what nutrient (N, P or both elements) are controlling or limiting algal growth in this reservoir.

OBJECTIVES

The goal of this project was to evaluate changes in trophic status in the upper reaches of Beaver Lake. To accomplish this goal the following objectives were proposed:

- Determine the trophic conditions in the headwaters of Beaver Lake in terms of chlorophyll *a* (Chl-*a*) and nutrient concentrations.
- Evaluate the variability in nutrient (particularly P) concentrations with depth in the headwaters of Beaver Lake.
- Determine the relationship between Chl-*a* and nutrient concentrations.

- Evaluate and compare the current conditions versus conditions found in previous studies conducted at Beaver Lake.
- Determine the limiting factor of algal (phytoplankton) growth at Beaver Lake.

HYPOTHESIS

Hypothesis 1

H_o: A marked longitudinal gradient does not exist in trophic conditions in Beaver Lake.

H_a: A marked longitudinal gradient in trophic conditions exists in Beaver Lake.

Hypothesis 2

H_o: Nutrient (particularly P) concentrations do not vary with depth in the transitional zone of Beaver Lake.

H_a: Nutrient (particularly P) concentrations vary with depth in the transitional zone of Beaver Lake.

Hypothesis 3

H_o: Algal (phytoplankton) growth measured via Chl-*a* concentrations is not correlated to the limiting nutrient.

H_a: Algal (phytoplankton) growth measured via Chl-*a* concentrations is correlation to the limiting nutrient.

Hypothesis 4

H_o: Algal (phytoplankton) growth is not limited by nutrients.

H_a: Algal (phytoplankton) growth is limited by nutrients.

SITE DESCRIPTION

Beaver Lake is located in Northwest Arkansas at the headwaters of the White River (Figure 1.1). The reservoir is impounded by a dam located west of Eureka Springs in Carroll County constructed in the early 1960s. Beaver Lake is the youngest reservoir in a chain of four reservoirs on the White River in Arkansas. The purpose for the construction of Beaver Lake was flood control, hydropower generation, and water supply. Today, Beaver Lake is also used for fish and wildlife habitat, recreation, and waste assimilation.

The watershed has a drainage area of 3,087 km^2 at the Beaver Lake dam, and Beaver Lake contains 2,040 million m^3 of water at the top of the current conservation pool (341.4 m above sea level). The surface area of the reservoir is 114 km^2, and the length of the reservoir is 80 km from the White River at the Highway 45 Bridge to the Beaver Lake dam. The depth of the reservoir at the dam at conservation pool elevation is about 60 m, and the average depth through the reservoir is 18 m (Haggard and Green, 2002).

The watershed extends south to north, from Crawford County to near the Arkansas-Missouri state line. The watershed extends east to west from the cities of Fayetteville, Springdale, Lowell, and Rogers to approximately 10 km east to Huntsville in Madison County. The watershed includes portions of Benton, Washington, Crawford, Franklin, Madison, and Carroll counties in Arkansas. The primary inflows into Beaver Lake are White River, War Eagle Creek and Richland Creek; inflows also include smaller creeks and tributaries. Beaver Lake is the primary source of drinking water in Northwest Arkansas and provides water for the cities of Fayetteville, Springdale, Bentonville, Rogers and other smaller cities, as well as flood control and hydropower generation. Beaver Water District, supplies an average 144,000 m^3 (BWD, 2006; Haggard and Green, 2002). The primary inflows to Beaver Lake contribute to the majority of the nutrient load; Haggard et al. (2003) reported that TP and TN loading was 71.5 and 1250 Mg yr^{-1}. Beaver Lake has a marked longitudinal gradient in trophic conditions where the riverine zone is considered eutrophic and the transitional zone is mesotrophic (Haggard et al, 1999; Galloway and Green, 2006). The fact that Beaver Lake is the water supply for Northwest Arkansas is the reason this study was conducted.

For this study, seven sites were selected on Beaver Lake with three sites located in the lacustrine zone and the other four sites located in the riverine zone. Two of the sites located in the riverine zone were on the selection of the sites was based on previous experiences, and locations via GPS coordinates are shown in Table 1.1 and Figure 1.1.

Table 1.1. Lake stations and latitude and longitude at each sampling station.

Site Name	Location	Latitud	Longitud
BLL	Beaver Lake, Lowell	36°15'33N	94°04'08W
HCK	Hickory Creek	36°14'23N	94°01'33W
WTR	White River	36°13'53N	93°59'50W
WEC	War Eagle Creek	36°11'29N	94°00'04W
WEI	War Eagle Inlet	36°09'12N	94°00'04W
FCK	Friendship Creek	36°12'45N	93°58'49W
BSP	Blue Spings	36°13'31N	93°58'31W

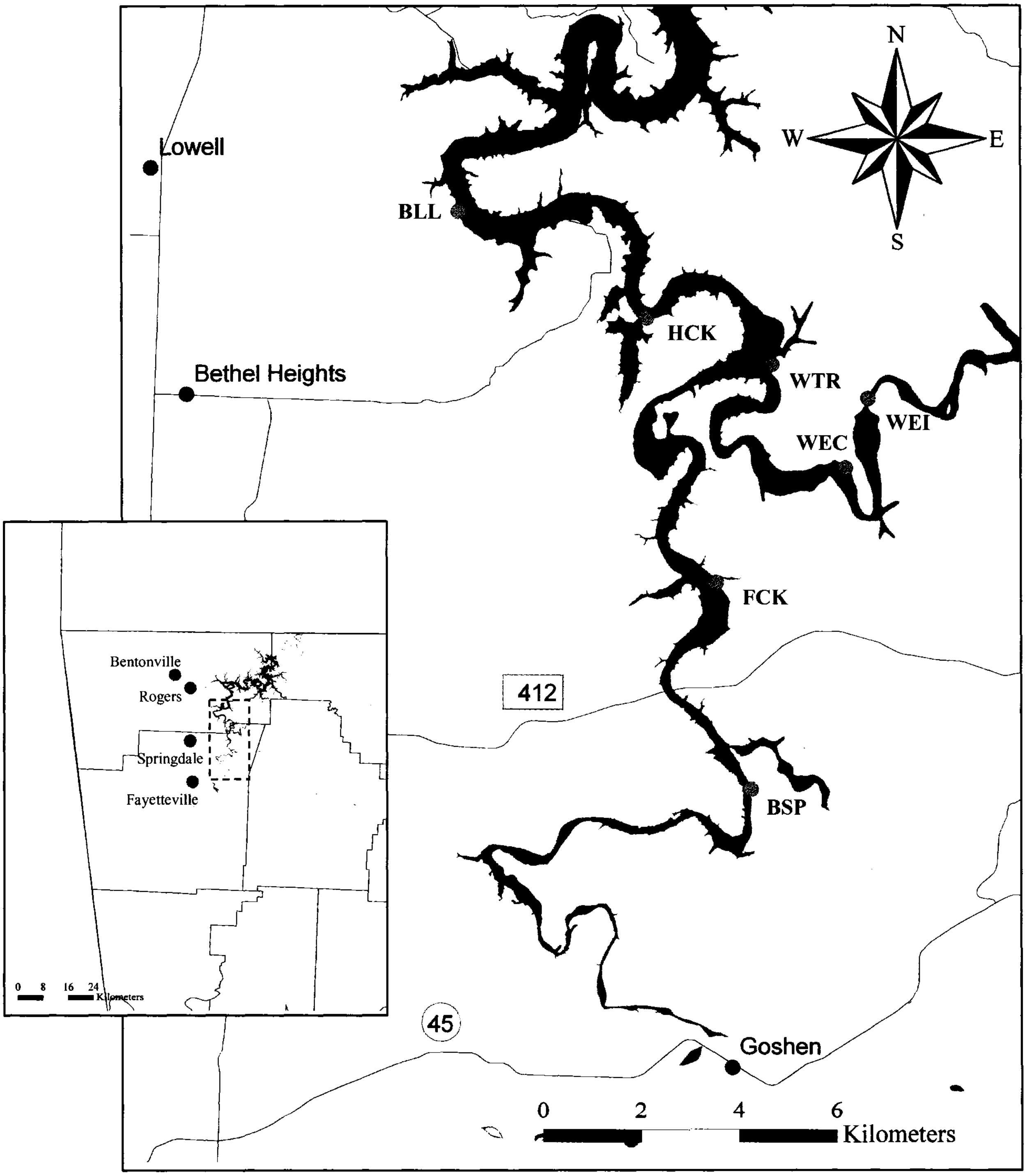


Figure 1.1. Beaver Lake reservoir and magnified view of the upper reaches of Beaver Lake and location of sampling stations.

CHAPTER 2

LITERATURE REVIEW

WATER QUALITY AND SOURCES OF CONTAMINATION

Industrialization and urban development are activities that contribute to significant environmental problems, although these activities are viewed as progress. Water quality is defined as the physical, chemical, and biological conditions of the water, where the presence and abundance of aquatic species depends upon physical and chemical characteristics (e.g., pH, temperature, nutrient concentrations and sediment concentrations) (Wang, 2001). Water quality in rivers, lakes, and other natural aquatic systems is impacted by the addition of pollutants, including nutrients. Nutrient enrichment (mainly N and P) of surface waters causes eutrophication, which is the most common impairment of surface water in the United States (U.S.) (Carpenter et al., 1998; Dodds and Oakes, 2006; Smith, 1999). Approximately half of the lakes and rivers in the U.S. are impaired because of accelerated anthropogenic eutrophication (USEPA, 2006). Typical natural factors that affect water quality are climate, watershed topography and geology, nutrients, and lake or reservoir stratification. Human pollution sources are classified as point and non-point sources (Atasoy et al., 2001; Hroncich, 1999).

Point sources (PS) by definition are discrete and stationary which makes this source easy to be identified, monitored, controlled, and regulated by government agencies. For example, the National Pollutant Discharge Elimination System (NPDES) is a program that manages the nation's water resources by regulating point source effluent discharges in aquatic ecosystems (USEPA, 2006). Point sources are the result of discharge from defined sources, such as municipal wastewater treatment plants and spills.

Non-point sources (NPS) are related to diffuse, landscape activities and episodic precipitation events. Non-point sources are difficult to measure and regulate, because these sources are often from extensive areas of land, such as runoff from agricultural lands and urban areas. Non-point sources are considered to be the major source of water pollution in the United States.

Several studies have provided evidence of the effects of urban residential construction and land use on water quality because these sources will determine the concentration of the contaminant transported to the water body (Atasoy et al., 2001; Fisher et al., 2000). For example, developed areas can also contribute to pollution with the generation of sediments (nutrients) from construction (Atasoy et al., 2001). Runoff from urban areas may be enriched with heavy metals and other pollutants (Tong and Chen, 2001). Moreover, runoff coming from agricultural land is enriched with nutrients and sediments (Fisher et al., 2000).

Young et al. (1995) reported the annual average nutrient export from different land uses in North America. Table 2.1 illustrates how different land uses effect water quality by exporting different amounts of pollutants into the water. According to the information show in Table 2.1, row crops and urban areas are the main sources of P and mixed agriculture is the main source of N.

Table 2.1. Summary of North America's annual average nutrient export (Kg ha^{-1} yr^{-1}) (Beaulac and Reckhow, 1982)

	P export (Kg ha^{-1} yr^{-1})		**N export (Kg ha^{-1} yr^{-1})**	
Land use type	**Range**	**Median**	**Range**	**Median**
Urban	0.7-2.8	1.2	4-12	5.5
Pasture	0.3-2.8	0.9	2-11	5.0
Mixed agriculture	0.5-1.5	1.0	9-26	14.0
Row crops	1.0-5.3	2.3	4-23	8.5
Non-row crops	0.7-1.6	0.8	4-7	6.0
Forest	0.1-0.4	0.3	2-4	2.5

Investigation and the knowledge acquired about the land-water relation is a useful tool for planners and decision makers to prevent and correct water quality issues. The objective of land-use planning is to maximize the uses of land while minimizing any negative environmental impacts (Wang, 2001).

RESERVOIR LIMNOLOGY

Reservoirs are surface water impoundments designed by man that have some characteristics that are different from those of natural lakes. In the U.S., the majority of reservoirs are in the southern, central, and western parts of the country. The primary uses of surface water impoundments are generally: flood control, recreation, water supply, and fisheries. Predominantly, reservoirs are located in non-glaciated areas having a greater water temperature and higher primary productivity than lakes, which are often in glaciated regions. Reservoirs tend to have a narrow and elongated drainage basin, while natural lakes often have a circular basin (Thornton, 1990; Wetzel, 1990).

Reservoirs are generally located near the outlet or the base of the drainage basin. The majority of the inflow entering the reservoirs comes from large tributaries and results in high energy flow for erosion processes, large sediment-carrying capacity, and extensive mixing of dissolved and particulate loads in the water. For this reason, reservoirs have more total sediment delivery when compared to natural lakes. This increased sediment loading influences light penetration, water clarity, nutrient and contaminant transport, and primary productivity (Thornton, 1990). Sediments are considered an important additional factor influencing the amount of nutrients in reservoirs and the degree of primary productivity. Since the headwaters present high concentration of suspended particles phytoplankton productivity may be low because of high turbidity (Kennedy and Walker, 1990; Wetzel, 2001).

James et al. (1987) proved that turbidity measurements were high in the headwaters of DeGray Lake, Arkansas but it decreased towards the dam. Also, during the high inflow rates the headwater stations present more suspended solids than the stations closer to the dam, because most of the material settles upstream.

Reservoirs develop longitudinal patterns identifying three zones: riverine, transitional, and lacustrine (Figure 2.1). The *riverine zone* is relatively shallow, narrow and well mixed. The suspended particles (e.g., silts, clay, and organic matter) are transported and enter the reservoir limiting light penetration and primary production. This zone presents the higher sedimentation rates decreasing towards the dam. The *transitional zone* is marked by an increased reservoir width and depth (decreasing flow velocity). Light penetration increases because of sedimentation processes. This zone can be productive, because of increased light and nutrient availability from internal and external sources. Finally, the *lacustrine zone* is closer to the dam and is characterized by clear and less productive water. Although light penetration increases, the availability of dissolved nutrients is generally reduced, and primary production is slow (Thornton, 1990). These zones are very dynamic and the boundaries between the zones are not static; thus, the size and location of these zones fluctuate, particularly between the riverine and transitional zones. The characteristics of the zones are a response to watershed runoff events, density flow characteristics, and reservoir operating procedures (Kimmel et al., 1990).

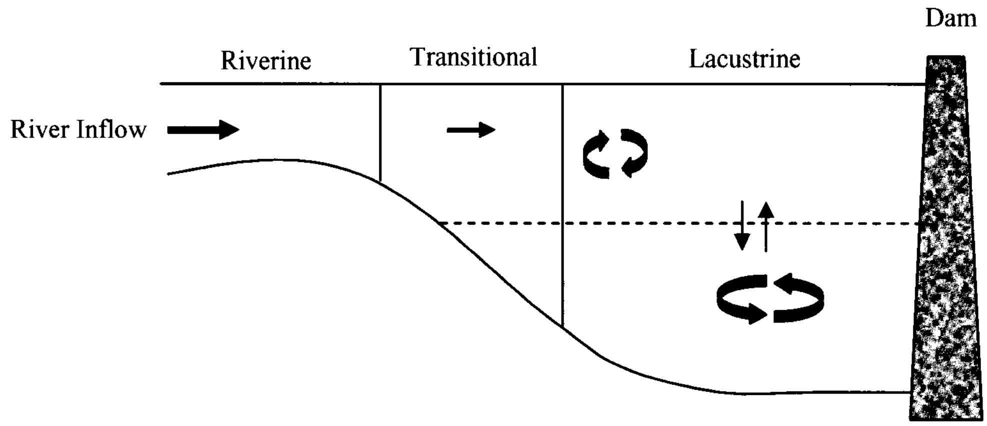


Figure 2.1. General schema of longitudinal gradients in reservoirs where the symbols represent the natural flow of the water and the recirculation and mixing processes (Modified from Wetzel, 2001).

In addition to longitudinal gradients, reservoirs are vertically stratified. This pattern is irregular and the riverine and transitional zones are often too shallow to demonstrate stratification. Vertical stratification, also called thermal stratification, is caused by changes in water temperature and density with depth. In late spring, the water heats from the surface down and thermal stratification occurs. The warm surface water floats on top of the layer of colder water below. Respectively, these two layers are called *epilimnion* and *hypolimnion* and these water layers are separated by the *thermocline*. The thermal stratification lasts during the whole summer. When the fall approaches, the *epilimnion* begins to decrease in depth and eventually the lake loses its stratification, because surface water gets cooler and denser. Therefore, the water near the surface sinks and helps to mix the lake, producing fall mixes or overturn. From late fall to early spring the water has generally uniform temperatures and the wind mixes the reservoir water (Wetzel, 2001).

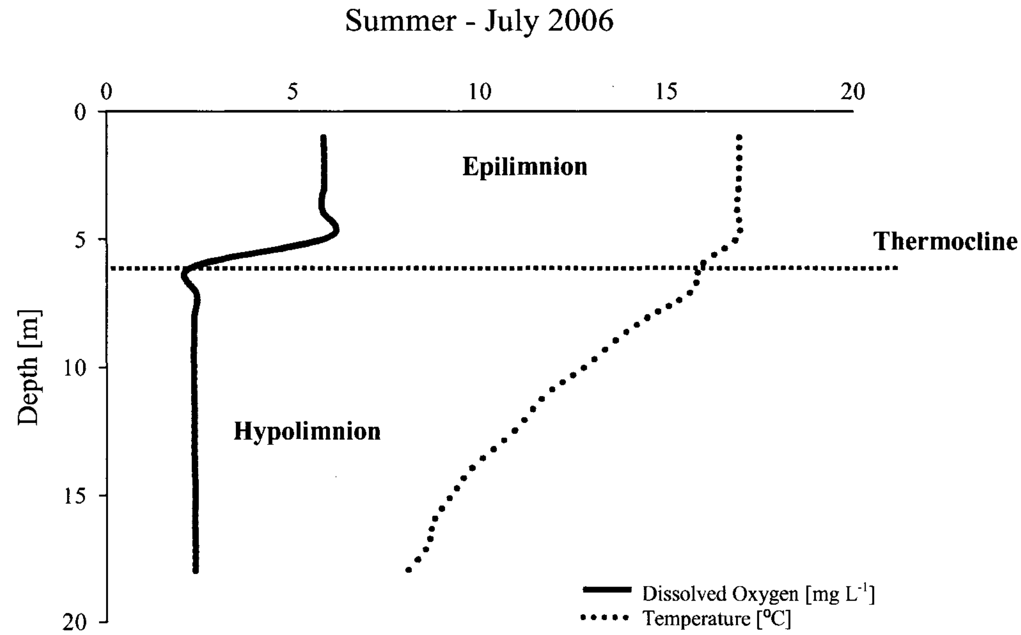


Figure 2.2. Example of a general summer stratification in Beaver Lake, AR July 2006.

RESERVOIR NUTRIENT DYNAMICS

Water quality and productivity of reservoirs are controlled by the quantity and quality of external and internal loading of sediments and nutrients. Nutrients and sediment are mainly transported with surface runoff, and the amount is dependent upon the presence of point sources, rainfall, and catchment characteristics. Other transport mechanisms for nutrients are subsurface flows and wind. For example, the amount of nutrients exported from sandy soils is less than the amount of nutrients transported from organic soils, which are commonly found in agricultural areas (Young et al., 1995; Tong and Chen, 2001). Internal loading is the process responsible for the cycling of nutrients and depends upon reservoir morphometry, inflow-outflow regimes, depth of discharge, and stratification characteristics (Gloss, 1980). In addition, external and internal loadings will depend on climatic regime, watershed characteristics, morphology, soil type, and land use (Kennedy and Walker, 1990; Sodergaard, 2003).

Nitrogen (N)

Nitrogen is essential for aquatic life; however, the excess of N can cause the degradation of water quality. Nitrogen exists in multiple forms in water: organic nitrogen (ON), ammonia (NH_3) or ammonium (NH_4^+), nitrite (NO_2^-), and nitrate (NO_3^-). The sources of N are wastewater treatment plants, septic systems, runoff from fertilized land, runoff from animal manure storage areas, industrial discharges, and atmospheric depositions (USEPA, 2006).

Nitrogen chemistry can be complex because N can adopt different oxidation states, depending on anaerobic or aerobic conditions. The water soluble species that are considered as environmental concerns are ammonium (NH_4^+), NO_2^-, and NO_3^- (Sawyer et al., 2003). Ammonium concentrations can be very low in unpolluted waters (0 to 1 mg/L), reaching over 10 mg/L in the anaerobic hypolimnion in eutrophic lakes. In the same way, NO_2^- concentrations are generally very low (0 to 0.01

mg/L), increasing under anaerobic and reducing conditions. Nitrate concentrations vary seasonally and spatially and the range can go from undetectable values to greater than 10 mg/L in unpolluted waters (Wetzel, 2001).

The organic N in aquatic ecosystems is gradually mineralized to NH_3, and the NH_3 is converted to NO_2^- and NO_3^-, under aerobic conditions. This process is called nitrification, and it is defined as the process by which nitrogen compound goes from a reduced state to a more oxidized one (Sawyer et al., 2003; Wetzel, 2001). Nitrate compounds can also reach the aquatic ecosystems by runoff or direct effluent discharges (Wetzel, 2001).

Nitrogen concentrations vary from season to season from lake to lake. During late summer and fall NH_4^+ concentrations are low in the epilimnion increasing under anoxic conditions in the hypolimnion. After the fall turnover NH_4^+ concentrations are uniform throughout the water column, but NH_4^+ will be rapidly oxidized to NO_2^- and then to NO_3^- (nitrification). During the summer, NO_3^- concentrations decrease drastically due to algal uptake (Wetzel, 2001). The internal loading of N modifies water nutrient concentrations and phytoplankton productivity. In the hypolimnion (sediment-water interface), NO_3^- is consumed by facultative bacteria producing NO_2^-, N_2O, N_2, or NH_4^+ (nitrate reduction), under anaerobic conditions. The NH_4^+ produced can be accumulated in the interstitial water or be adsorbed to the sediments (Nowlin et al., 2005; Medina et al., 2002).

The relationship between TN and Chl-*a* can be vary greatly depending upon chemical, physical, and biological characteristics of the aquatic ecosystem. The dependence of phytoplankton production on TN concentrations will depend upon optimal conditions and availability. Studies have shown (Jones and Knowlton, 1993; Guildford and Hecky, 2000; Smith, 1982) positive relation between these variables but phytoplankton production has also shown a co-dependence on other parameters such as P concentrations and availability.

Phosphorus (P)

Phosphorus is considered to be in short supply in natural impoundments. The increment of even small concentrations (under the right conditions) can cause eutrophication problems (algal blooms, low dissolved oxygen, and change in algal species composition). Therefore, P is considered the most important factor in determining water quality. In aquatic ecosystems P can only be in the pentavalent form for example orthophosphate, pyrophosphate, polyphosphate, and organic phosphate (phosphate molecule PO_4^-). Phosphorus is transported to the aquatic ecosystems in two forms: a mixture of dissolved forms or soluble reactive phosphorus (SRP) and particulate P. The main sources of P are soils, weathered rocks, wastewater treatment plant effluent, runoff from fertilized croplands, septic systems, and land areas disturbed by construction or other human activities (USEPA, 2006; Sondergaard, 2003; Correll, 1998).

Internal loading is a very important process for recycling P in natural waters. Phosphorus release can be so intense and persistent that it prevents any improvement in water quality even if the external loading sources have been reduced. Once delivered to the lake or reservoir, a certain amount of P is retained in sediment bounded to calcium, manganese, aluminum, and iron minerals. The proportion of P retained in sediments and the amount released to the water depends upon the retention time, physical, chemical, and biological characteristics. The anoxic conditions in the hypolimnion provide optimal conditions for P release from the sediment. After being released, P can reach the epilimnion as a result of the turnover particularly during fall mixes (Tsujimura, 2004). The main release mechanisms are:

- *Resuspension*. This process is very important in shallow lakes and is caused by wind that produces the resuspention of solid particles and total phosphorus. A study in a shallow Danish lake showed an increment of a factor of 5 – 10 in the concentration of suspended solids and total phosphorus after wind events (Sondergaard et al., 2003).
- *Temperature (T)*. Temperature controls biological processes and P release. Mineralization of organic matter releases inorganic P as T increases. When the proportion of iron-bound phosphorus is large

the effect of T is more significant. Changes in T will modify the biological and chemical reactions. Therefore, when T increases, the diffusion rate increases promoting to P release. However, when T increases biological activities and mineralization rates are stimulated (Wassmann and Olli, 2004; Sondergaard et al., 2003).

- *Redox conditions.* Oxidation and reduction are the classical explanation for P release from sediments. When the hypolimion presents aerobic conditions, iron (Fe) is found in its oxidized form Fe (III). Under these conditions Fe (III) can bond to P forming very stable Fe-P complexes. When oxygen concentrations are depleted and anoxic conditions occur, in the hypolimnion, iron is reduced to Fe (II) and P returns to solution. In addition, the conversion of sulfate to sulfide affects the availability of Fe to bond P (Carpenter et al., 1999; Sondergaard et al., 2003).
- *pH.* It can be particularly important when the retention of phosphorus depends on iron. When high pH is present the hydroxyl ion increases and it competes with P ion to bind with iron. Therefore, P release rates increased at higher pH (Sondergaard et al., 2003).
- *Mineralization and microbial processes* have an important role in uptake, storage, and release of P. Organic mineralization takes place at high rates when oxygen or NO_3^- are available. If NO_3^- concentrations are low and sulphate levels are sufficient, sulphur

cycling may become important in the sediment processes (Sondergaard et al., 2003).

Sondergaard et al. (2003) reported that shallow eutrophic lakes present high concentrations of P during the summer. Tsujimura (2005) has reported large quantities of PO_4^- in the hypolimnion of Lake Yogo that seemed to disappear after the fall turnover and also high concentrations of PO_4^- are observed in the epilimnion in the early fall. Haggard et al. (2005) concluded that P released from the bottom sediments in Lake Eucha, Oklahoma is four times greater in anaerobic conditions than in aerobic conditions. This internal P load was approximately 25% of the annual P load estimated from external sources.

RESERVOIR PHYTOPLANKTON PRODUCTION

Primary productivity is one of the most used biological indicators to evaluate water quality. Also, it has been demonstrated to be a useful and complementary tool to the physical and chemical analysis (Wang, 2001). Primary productivity in reservoirs is basically dominated by phytoplankton, particularly in relatively clear water and stable water levels. Small algae (<8μm) dominates phytoplankton productivity. Primary productivity is controlled by physical (light and temperature), chemical (nutrient availability), and biological (competition and predation) factors which also depend on watershed characteristics, reservoir morphology, inflow, and the food-web structure of the reservoir (Kimmel et al., 1990).

Phytoplankton is the group of algae or photosynthetic bacteria with limited power of locomotion; therefore, their distribution depends on water movement. In contrast, periphyton is the term applied to microbiota living on any substratum (Wetzel, 2001). Phytoplankton abundance presents a seasonal succession caused by changes in physical and chemical characteristics and trophic conditions, in which not only the total biomass changes, but also the species in each community (Wetzel, 2001).

Winter: During the winter the algal growth rate is reduced, because light and temperature are low and the dominant species move deeper into the water column. The population is dominated by small motile species including *Crypyophyceans* such as *Rhodomonas* and *Cryptomonas*, dinoflagellates (*Gymnodinium*), small green algae (*Chlamydomas*), some diatoms, and euglenophyceans (Holmes, 2000).

Spring: Maximum algal growth occurs after stratification has begun. Early spring is dominated by *Rhodomonas* and spring blooms are dominated by diatoms and its growth eventually will decrease, because nutrients in the epilimnion are depleted. *Astorionella* often dominates other diatoms and its growth rate depends on P concentrations (Holmes, 2000).

Summer: This period present the best light conditions; therefore, the diversity of species increases. Summer population will vary, depending upon the trophic conditions of the lake. In eutrophic lakes the algae population is dominated by blue-green algae (*Oscillatoria*), because the algae are able to fix N and survive during N depletions. Blue-green algae are also able to move in order to find more optimal conditions. They also float when water is turbid (Holmes 2000; SWCSMH, 2006).

Seasonal changes in phytoplankton production could be constant or similar from year to year only if the watershed system is not disturbed by any human activities. If the watershed characteristics, weather conditions, and loadings to the reservoir are modified from year to year algae responses could be different and sometimes unexpected for the season (Holmes, 2000). Studies in lakes in Madison, Wisconsin have shown that during the winter time, when algal growth decreases and the lake is not stratified, soluble N and P concentrations increased significantly (Gerloff and Skoog, 1957).

Dillon and Rigler (1974) analyzed the spring mean P concentrations versus summer mean Chlorophyll concentrations for 19 lakes in Southern Ontario combined with data collected from 49 lakes in Connecticut. The study found a strong relationship between these two variables ($r = 0.93$) which was compared to results reported in Sakamoto (1966) who found the same relationship between Chl-*a* and TP in Japanese Lakes.

Brylinsky and Mann (1973) found that energy-related variables had a higher correlation than the chemical factors like TP with primary production. However, the authors considered TN as an exception because this variable is highly correlated to phytoplankton production and energy-related variables. Moreover, when these two variables (energy and chemical factors) were analyzed together a better regression was produced. This study also found a very strong direct correlation between phytoplankton production and Chl-*a* concentrations, considering Chl-*a* as good representation of primary production and nutrient conditions in lakes. Finally, Chl-*a* concentrations presented positive correlations with mean nutrient concentrations, for example: Chl-*a* - TP r = 0.54, Chl-*a* - PO_4 r = 0.78, Chl-*a* - TN r = 0.49, and Chl-*a* - NO_3 r = 0.59.

Dzialowski et al. (2005) reported that no correlation was found between Chl-*a* and nutrient concentrations (total or dissolved). However, when light was a controlled variable these correlations were significant. The correlations presented a positive slope between the Log_{10} transformed Chl-*a* concentrations and the Log_{10} transformed TP and TN concentrations. This conclusion shows that primary production which is directly related with Chl-*a* concentrations does not only depend on nutrient availability, but also on light penetration and energy-related variables. Jones and Knowlton (1993) in their limnological study of Missouri reservoirs had found that the Log_{10} transformed Chl-*a* and TP concentrations present a positive linear correlation; however, when the effect of non-algal material is significant the relationship between Log_{10} Chl-*a* and Log_{10} TP becomes curvilinear.

An evaluation of a reservoir in the Korean peninsula (An and Kim, 2003) showed that the Log_{10} transformed of Chl-*a* concentrations versus

Log_{10} transformed TP concentrations had a very strong positive linear regression (R^2 = 0.915). On the other hand, TN results showed that phytoplankton production has a weak correlation to TN concentrations. Therefore, P was considered as a limiting factor in algal production.

NUTRIENT LIMITATION AND PHYTOPLANKTON PRODUCTION

Phytoplankton production in reservoirs is primarily limited by temperature, light, and the availability of macro and micronutrients. Because of the complex and dynamic environment, algal growth limitation can not be attributed only to one limiting factor. In addition, algal growth rate is determined by the relative availability of one or more of these factors. Algal growth increases when the availability of the limiting factors increases or until other factors become more limiting. Specifically in reservoirs, the two main limiting factors (light and nutrient availability) are influenced by inflow characteristics and vertical mixing regime, because phytoplankton production occurs only in the photic zone (Kimmel et al., 1990).

Kimmel et al. 1990 classified limiting factors into two categories: physical and chemical factors such as light, turbidity, temperature, pH, and external and internal loading. Biotic factors include: influences of algal photosynthesis efficiency and phytoplankton losses from predation and parasitism (food-web structure).

Nutrient enrichment is the most widely used experiment to determine nutrient limitation in reservoirs and is a useful tool to understand the nutrient and phytoplankton dynamics. These experiments consist in the addition of nutrients to the water and the measurement of primary production as a response (Horvatic et al., 2006). These experiments can be developed in situ or in experimental stations and for short-term (hours-days) or longer-term bioassays (weeks-months). Fang et al. (1993) reported that short-term bioassays are widely used to determine phytoplankton nutrient limitations and microcosm experiments have been used to manipulate nutrient additions and determine estuarine algae nutrient limitations.

Some of the advantages and disadvantages of long-term and short term bioassays were reported in Dodds and Priscu (1990). Long-term experiments may be affected by the containers which sometimes limit light penetration. Also, when the containers are sealed gas exchange is affected. However, the authors reported that in experiments developed for over 4.5 d the effect of the containers was negligible. Short-term experiments may be required smaller scale systems and long-term experiments may be more adequate for large and in situ experiments. Although short-term small-scale bioassays do not represent the entire system, they can provide insight into system conditions. In addition, short-term experiments that only focused in the effect of one nutrient may not reflect the combined effect with other limiting factors.

Gerhart and Likens (1974) classified the enrichment methods in four groups: ^{14}C bioassays, enrichment of continuous or batch cultures, enrichment *in situ* polyethylene enclosures, and experimental enrichment of whole lake ecosystems. Experiments *in situ* permit the manipulation and addition of nutrients in bigger volumes of water under natural conditions during longer periods of time; however, none of the parameters affecting algal growth can be controlled. Bioassays develop in experimental stations, need the transportation and transference of water to artificial environments where the parameters of interest can be controlled by maintaining the rest of the parameters constant. This type of experiment uses smaller volumes of water for shorter periods of incubation.

A review of North America freshwater bioassay experiments reported that the most common response of phytoplankton communities to nutrient addition is the co-limitation of N and P rather than just a limitation by either N or P (Elser et al., 1990). However, other studies like Smith et al. (2002) and Havens and Walker (2002) concluded that P is the primary nutrient limiting phytoplankton growth and, for this reason, most of the management efforts are oriented to reduce and control P loadings.

The results of bioassay experiments in 19 eastern Kansas reservoirs are reported in Dzialowski et al. (2005). The results proved that the majority of the reservoirs present a co-limitation (68%) response during the spring but 11% of the reservoirs present a P limitation, the percentage of N-limited reservoirs increased during late summer/fall experiments and the P-limited reservoirs decreased during the same period. Even though, seasonal changes have been observed during the experiments there was not enough evidence to determine the seasonal patterns and they are assumed to be caused by the difference in nutrient requirements of phytoplankton taxa and composition of the phytoplankton communities shifts every season leading to a different nutrient limitation. The authors used Tukey's Honestly Significant Difference ($P = 0.05$) to determine the significance in difference between treatments and the control, in order to understand nutrient limitation. The sequence of analysis included: if N > Control and/or P > Control then system was N and/or P limited, if NP > N or NP > P the system presents a co-limitation, and if the Control > or = P, N, or NP then the system did not present any nutrient limitation. For this study light was a controlled parameter; therefore, only 16% of the bioassay experiments were affected by light limitation.

Piehler et al. (2004) bioassay experiments in Palmico Sound, NC have found that during the fall phytoplankton growth presented a N-P co-limitation. In addition, during the winter N limitation was observed; during the spring the NP treatment and the N treatment did not present a response significantly different from each other. Finally, in the summer there were significant statistical differences between treatments. Overall, the authors concluded that Palmico Sound was mainly N limited.

Missouri reservoirs (Jones and Kwolton, 1993) have reported low TN:TP ratios. For this reason, N limitation is suspected in these reservoirs. Comparing the results with other studies, the authors have found that other reservoirs in the southern U.S. present similar situation with N limitation and temporal N-P co-limitation for algal growth.

Phytoplankton responses to nutrient addition can vary from reservoir to reservoir. Also, seasonal changes have been observed within reservoirs and these seasonal variations can be different among water bodies depending on their physical, chemical, and biological characteristics and interactions between them. Bioassay experiments are controversial because they are not considered an adequate response of the ecosystem to nutrient enrichment since they are developed on a small scale. In order to reduce this uncertainty, Piehler et al. (2004) evaluated and analyzed the effects of addition of nutrients (that are likely to limit productivity) to phytoplankton productivity with the purpose of finding consistent responses and logical patterns. Piehler et al. (2004) found that the Pamlico Sound lagoon is N limited and occasionally presents a N-P co-limitation.

EUTROPHICATION AND DRINKING WATER PROBLEMS

Carlson (1977) defined trophic state as "the total weight of living biological material in a water body at a specific location and time. The trophic state is understood to be the biological response to forcing factors such as nutrients addition, but the effect of nutrients can be modified by factors such as season, grazing, mixing depth, and others".

The most common way to classify lakes is based on their productivity. This is determined by the nutrients from internal and external loadings, atmosphere deposition, depth, shape, and geological nature of the drain basin, and climate. Based on "trophic" characteristics lakes are divided into three characteristics: oligotrophic, mesotrophic, and eutrophic (Wetzel, 2001).

Oligotrophic are those lakes in which primary productivity is limited by a low chemical concentration (N and P). In this kind of lake the variety of species is high but the number of each species is low. These lakes are usually large, deep, and clear with high levels of dissolved oxygen (Wetzel, 2001; Kevern et al., 1996).

Mesotrophic lakes are at an in-between nutritional stage. They support a diverse community of aquatic plant and animal life. Many U.S. lakes are at this point of evolution (Wetzel, 2001; Kevern et al., 1996).

Eutrophic lakes have large supplies of nutrients and heavy layers of organic sediments on their bottoms. The inflow of sediments makes the lakes shallow and turbid, because the high nutrient levels cause the excessive growth of algae. As a lake becomes increasingly eutrophic, the number and type of species on the bottom will change. The high rate of decomposition produces excessive demand of oxygen, so these lakes have low dissolved oxygen levels (Wetzel, 2001; Kevern et al., 1996).

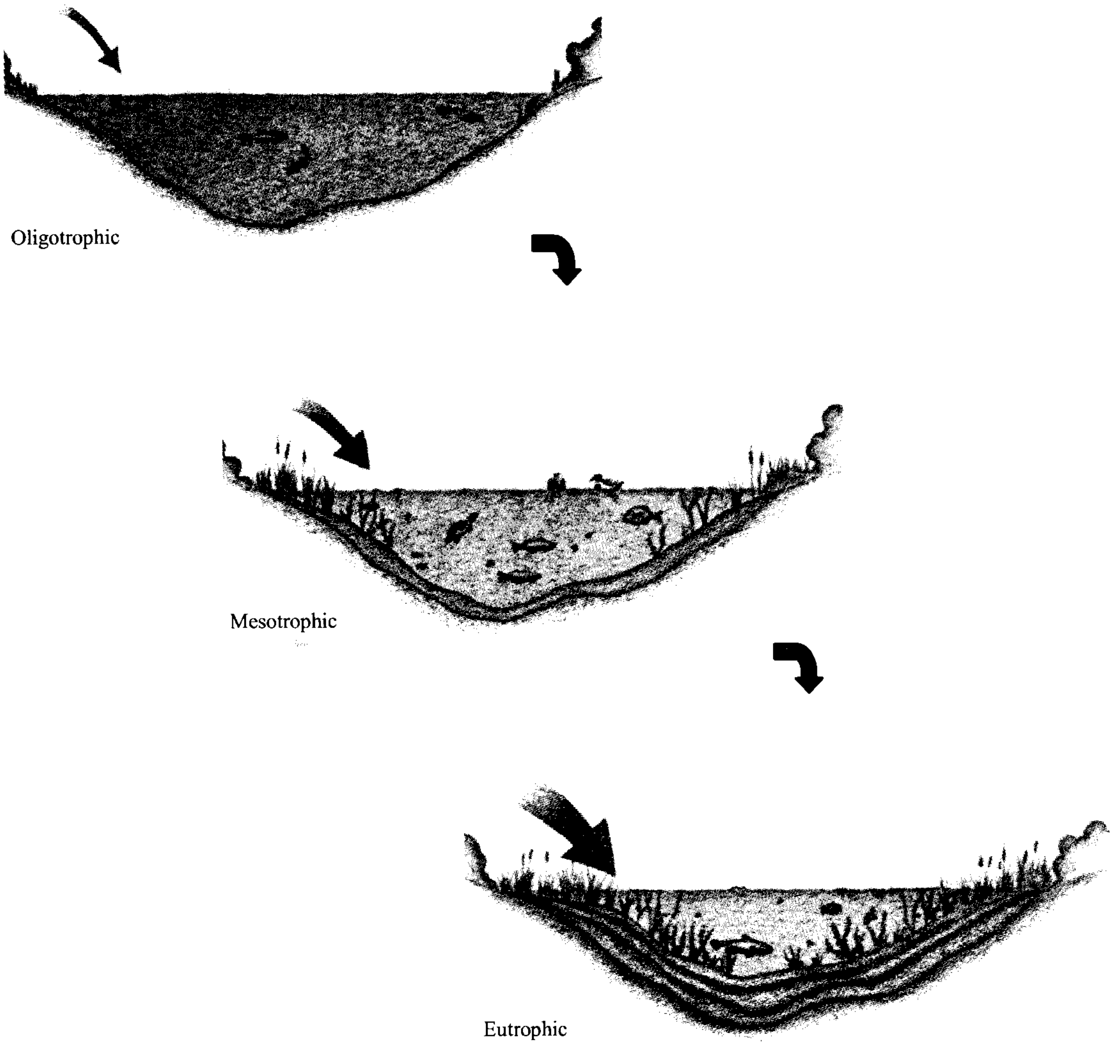


Figure 2.3. Graphical and sequential representation of lake classification based on nutrient concentration and primary productivity.
(Modified from Lander, 1980)

Carlson trophic state index (TSI) is one of the most commonly methods used to determine trophic state of water bodies. This method uses three variables: chlorophyll (Chl), Secchi depth (SD), and total phosphorus (TP). These three variables are interrelated by linear regression models. In theory, any of these three variables can be used to classify a water body. However, chlorophyll is considered the most accurate and is a better predictor (Carlson, 1977).

Carlson (1977) defined TSI based on the criteria that a new trophic condition is reached every time the base chlorophyll value is doubled. In addition, every time algae double in biomass the Secchi transparency decreases in half. The equation to calculate the TSI (SD) values is:

$$TSI\,(SD) \;=\; 10\left(6 - \frac{\ln SD}{\ln 2}\right)$$

If the water body has an influence of non-algal particulate matter the Secchi disk transparency readings could not be representative and erroneous values may be recorded. The advantages of this method are the simplicity and low cost and usually generates values similar to those obtained for chlorophyll. According to Carlson (1977) Chl values are free from interferences. The equation for to calculate TSI (Chl) is:

$$TSI\,(Chl) \;=\; 10\left(6 - \frac{2.04 - 0.68 \ln Chl}{\ln 2}\right)$$

The TSI (TP) value is calculated based on the criteria that TP is the major limiting factor for algal growth and the value can be calculated using the equation below:

$$TSI\ (TP) \ = \ 10\left(6 - \frac{\ln \frac{48}{TP}}{\ln 2}\right)$$

The TSI values can not be averaged because the three variables (chlorophyll pigments, Secchi depth, and total phosphorus) estimate algal biomass independently (Carlson, 1977). Once the TSI values are calculated, they can be analyzed using Figure 2.4.

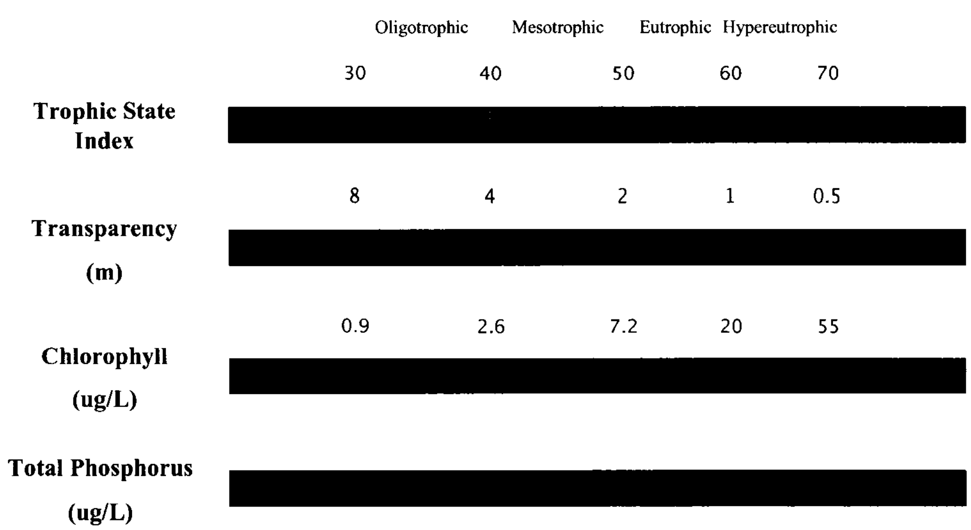


Figure 2.4. TSI values for each variable and their relationship with lake classification. (Adopted from Carlson, 1977)

Table 2.2 is a summary of the relationship between TSI values for the three variables of interest and the probable conditions of the lake depending on their classification (Carlson, 1977).

Table 2.2. TSI values relationship and possible conditions of water bodies. (Adopted from Carlson, 1977)

Relationship between TSI variables	**Conditions**
TSI (Chl) = TSI (TP) = TSI (SD)	Algae dominate light attenuation; TN/TP ~ 33:1
TSI (Chl) > TSI (SD)	Large particles, such as Aphnizomenon flakes, dominate.
TSI (TP) = TSI (SD) > TSI (Chl)	Non-algal particulates or solor dominate light attenuation.
TSI (SD) = TSI (Chl) > TSI (TP)	Phosphorus limits algal biomass (TN/TP > 33:1)
TSI (TP) > TSI (Chl) = TSI (SD)	Algae dominate light attenuation but some factor such as nitrogen limitation, zooplnakton grazing or toxics limit algal biomass.

Table 2.3 shows a more complete information for TSI values and the concentrations for each variable (SD, Chl, and TP) related to the trophic state and their most important characteristics. Some of these characteristics may vary depending specially with the altitude and latitude of the water body (Carlson, 1977).

Table 2.3. TSI values and the possible characteristics of north temperate lakes depending on algal biomass changes. (Adopted from Carlson, 1977)

TSI	Chl (ug/L)	SD (m)	TP (ug/L)	Attributes	Water supply	Fisheries & recreation
< 30	< 0.95	> 8	< 6	**Oligotrophy**: Clear water, oxygen throughout the year in the hypolimnion.	Water may be suitable for an unfiltered water supply.	Salmoid fisheries dominate.
30 - 40	0.95 - 2.6	8 - 4	6 - 12	Hipolimnia of shallower lakes may become anoxic.		Salmoid fisheries in deep lakes only.
40 - 50	2.6 - 7.3	4 - 2	12 - 24	**Mesotrophy**: water moderately clear; increasing probability of hypolimnetic anoxia during summer.	Iron, manganese, taste, and odor problems worsen. Raw water turbidity requires filtration.	Hypolimnetic anoxia results in loss of salmoids. Walleye may predominate.
50 - 60	7.3 - 20	2 - 1	24 - 48	**Eutrophy**: Anoxic hypolimnia, macrophyte problems possible.		Warm - water fisheries only. Bass may dominate.
60 - 70	20 - 56	0.5 - 1	48 - 96	Blue - green algae dominate, algal scums and macrophyte problems.	Episodes of severe taste and odor possible.	Nuisance macrophytes, algal scums, and low transparency may discourage swimming and boating.
70 - 80	56 - 155	0.25 - 0.5	96 - 192	**Hypereutrophy**: (light limited productivity). Dense algae and macrophytes.		
> 80	> 155	< 0.25	192 - 384	Algal scums, few macrophytes.		Rough fish dominate.

The TSI values to determine trophic conditions of a water body are a relatively easy and very useful method, which can be used as a guide for water management to improve water quality. Eutrophication is one of the biggest causes of reservoirs and lakes impairment around the world. Eutrophication is caused by the excessive addition of nutrients and organic inputs from human activities. Eutrophication is characterized by increment in primary production (phytoplankton biomass), loss in water clarity (decrease in light penetration), and depletion in hypolimnetic oxygen. From a human perspective eutrophication problems can cause aesthetic appeal, interfere with recreational activities, loss in accessibility, the overproduction of bacteria, algae and fungi may be part of the cause of odor and taste problems, and increased costs of treating drinking water (Dzialowski et al., 2005; USEPA, 2006, Smith, 1999).

In eutrophic conditions, phytoplankton communities are basically dominated by filamentous and/or colonial cyanobacteria, which are considered as a nuisance. These phytoplankton species are responsible for taste and odor problems, can cause anoxia when decomposing and may be harmful if endotoxins are produced (Havens and East, 1997). In addition, taste and odor problems are related to volatile organic compounds, such as geosmin (trans-1, 10 dimethyl-trans-9-decalol) and MIB (2-methyl isoborneol), which are produced by many aquatic microorganisms. Blooms of blue-green algae are associated with this problem too (Smith et. al., 2002). Studies in Clinton Reservoir, Kansas, concluded that the control of cyanobacteria would alleviate taste and odor. Smith et al. (2002) have recommended P control, to reduce total algal biomass, as an effective management tool to prevent and avoid taste and odor events.

Eutrophication of reservoirs and lakes in southern states is enhanced due to warm, fertile climates, and human activities. Most of the lakes in Texas were classified based on Carlson's Trophic State Index (TSI). The evaluation of 94 reservoirs in Texas, with TSI, has shown different tendencies. The estimation of TSI Chl*a* shows an improvement in 11 reservoirs but an increment in biomass in 78 out of the 94 reservoirs evaluated (Texas Commission on Environmental Quality, 2005)

Carlson's TSI values have a certain limitations in its application in natural lakes and reservoirs. Randolph and Wilhm (1984) studied Lake Carl Blackwell in northcentral Oklahoma and the TSI values reported for this natural impoundment were significantly different. The reason for this variation was attributed to the interference of non-algal particles that limit the transparency depth.

REPORTED CONDITIONS OF BEAVER LAKE

Beaver Lake is an important area of study because of the continuous concerns about the water quality in this impoundment. Beaver Lake is impacted by point and non-point pollution sources (Haggard and Green, 2002). Galloway and Green (2006) have reported that the White River branch receives effluent discharges from Fayetteville's wastewater treatment plant (WWTP) and the city of West Fork. Also, War Eagle Creek receives effluent discharge from the city of Huntsville WWTP. Water quality in Beaver Lake is a topic of concern because the addition of nutrient can cause eutrophic conditions which could cause an increment in costs for drinking water treatment. The data collected and analyzed by Galloway and Green (2006) during 2003-2004 for Beaver Lake's inflow showed that White River had greater concentrations of N than War Eagle Creek. The concentrations of N in War Eagle Creek were attributed to agricultural activities. The concentrations of P were often low (below detection limit 0.01 mg/L for orthophosphate and 0.02 mg/L for dissolved

P). However, the P concentrations in War Eagle Creek were greater than P concentrations in the White River arm.

Haggard (1999) reported that Beaver Lake has experienced an increment in the level of eutrophy in the last 10 yr. The data collected showed that the riverine zone was eutrophic and the transitional zone was mesotrophic. However, the increment of external and internal loadings can change the algae biomass and trophic conditions. The differences in trophic conditions show the formation of longitudinal patterns where Chl-*a* concentrations decrease from upstream to the dam. Beaver Lake in Arkansas was also classified based on its production and the results were reported in Haggard et al. (1999). The riverine zone presented the highest concentrations of Chl-*a* and it was classified as eutrophic; however, the transitional zone was classified as mesotrophic. In addition, the authors reported seasonal gradients with peaks of TN and NO_3^- during the summer and winter. Total P concentrations displayed a peak during the spring and fall and low concentrations during the summer. In addition, TP concentrations show decreasing longitudinal differences from the headwaters to the lacustrine zone (Haggard et al., 1999). Galloway and Green (2006) reported similar seasonal, vertical, and longitudinal patterns.

Haggard et al. (1999) reported a log-linear relationship between Chl-a and TP or PP for Year 1 (1993-1994). Also, it was found that SRP and NO_3-N presented a decreasing log-linear trend. In addition, ammonia presented a positive relationship. According to Haggard et al. (1999) Total Kjeldhal Nitrogen (TKN) was the nutrient that explain the greatest portion of the variance in Chl-*a* concentrations. Finally, a multilinear analysis between Chl-*a* and TN and TP showed a correlation between primary productivity and nutrient limitation.

WATER QUALITY ASSESMENT

Different methods have been developed in order to assess water quality in water bodies. The specific methods depend upon the type of evaluation. Methods used in order to determine water trophic status and nutrient limitation in reservoirs include measuring physical and chemical water quality parameters, and using bioassays to measure trophic conditions and primary productivity response to nutrient addition.

Water Quality Sampling

Randolph and Wilhm (1984) collected water samples from the surface (0.5 m) to determine Chl-*a* and SRP concentrations and seasonal variations during one year of sampling (September 1980 to November 1981). In addition, physical characteristics of the reservoir were measured including pH, dissolved oxygen (DO), Secchi Depth, and turbidity. All of these parameters were used to determine seasonal variations, algal growth relationships, and trophic conditions of the reservoir using TSI values.

Several other studies (Jones and Knowlton, 1993) have used Chl-*a* as a measurement for primary productivity for general reservoir limnology and water quality evaluation. Nutrient concentrations were also measured as they related to algae blooms, trophic state, and water contamination. Smith (1979) limited the sampling dates to the growing season (May-September) in order to determine the dependency of algal growth to nutrient concentrations.

The long and narrow morphology often presented in reservoirs leads to the formation of longitudinal gradients in water quality characteristics (Thornton, 1990). Kennedy et al. (1982) found a marked longitudinal gradient of TP, Chl-*a*, and suspended solids (turbidity) from the headwaters to the dam in West Point Lake, GA. Wang et al. (2005) in order to study the longitudinal gradients of Clinton Lake, Kansas monitored different sites based on the impoundment morphology. The sites were classified in three zones: riverine, transitional, and lacustrine. Water samples were collected from each site monthly and at different depths starting from the surface to the bottom.

In conclusion, to study seasonal changes or seasonal patterns samples must be collected for a whole year. The surface water samples will give an idea of the longitudinal variations. However, if vertical patterns need to be known samples from surface to bottom need to be considered. The lacustrine zone can give a very good idea of the stratified or non-stratified behavior of the lake, but for a more complex study, samples throughout the whole reservoir must be taken, analyzed, and evaluated.

Nutrient Enrichment Bioassays

In order to determine the main limiting factors for primary productivity in reservoirs, diverse methodologies have been developed. Nutrient enrichment bioassays can be developed in situ or ex situ. An (2003) used an in situ test to determine whether primary production varied with nutrient addition spatially and temporally. To accomplish this goal the experiment was performed in 4 sites (headwaters, midlake, downlake, and point-source) for a period of 5 months (May-September). Water samples were placed in 10L cubitainers and nutrients were added in different combinations during an incubation period of 5-6 days. The experimental design included a Control, individual N and P additions, and a combination of both (N and P).

Havens and East (1997) studied nutrient limitation in Lake Okeechobe, FL using in situ bioassays for algae. The in situ bioassay experiments used 20L carboys and four different treatments: Control (with no nutrients added), N and P in different concentrations, and a combination of both. The carboys were placed at half of the Secchi Depth reading and incubated for three days. Chlorophyll and nutrient readings were taken at the initial day and day three.

Dzialowsk et al. (2005) have used the ex situ bioassay experiments, in order to determine nutrient limitation and algal growth in Kansas Reservoirs. They collected 20-L of surface water from the main basin. The samples were transferred to the Environmental Bioassay Research Facility

(EBRF) and placed individually in 1L bottles where different treatments were applied which included the addition of TN and TP. The experimental design is similar to the one previously described. For this experiment light conditions were carefully controlled. The incubation period was between seven and nine days.

Maberly et al. (2002) presented a modified version of the ex situ nutrient enrichment bioassay. They used boiling tubes for incubating algae. Samples were kept at 20°C and continuously illuminated at a constant radiation. The treatments used were: Control, N, P, NH_4, N+P, and P+NH_4. Measurements of Chl-*a* concentration were taken at initial day (day 0), four readings were taken between days three and seven, and the final reading was at day fourteen. Ex situ bioassays are developed in artificial environment were certain parameters can be controlled depending on the objectives of the study.

CHAPTER 3

TROPHIC CONDITIONS AND NUTRIENT LIMITATIONS IN THE HEADWATERS OF BEAVER LAKE, ARKANSAS, DURING A DRY HYDROLOGIC YEAR, 2005-2006

ABSTRACT

Algal nutrient limitation and trophic state were evaluated in the headwaters of Beaver Lake, AR. Water samples were collected approximately monthly from July 2005 to July 2006. Longitudinal gradients were observed in nitrogen (N) and phosphorus (P), where generally concentrations were greatest in the riverine zone and decreased in the transitional zone. In addition, vertical patterns were observed in N and P concentrations, particularly with soluble reactive phosphorus (SRP) which was greater in the hypolimnion and near the sediment-water interface. Chlorophyll-*a* concentrations showed a positive and significant linear relationship with TP and TN. Water quality results were compared with historic data, suggesting that in Beaver Lake Chl-*a* and TP concentrations have increased in the past 25 years. Conversely, TN concentrations have been reduced. The trophic status of the reservoir was eutrophic for the riverine and transitional zone, and this may be attributed to a very dry season and low water levels in the reservoir, resulting in higher nutrient residence times in the riverine and transitional zones. Nutrient enrichment bioassays were used to determine nutrient limitation status and results showed that Beaver Lake nutrient limitation can shift from either N or P limitation to a co-limitation. This behavior can be related to the availability of nutrients (N and P) for algal uptake in the epilimnion.

INTRODUCTION

Reservoirs are artificial impoundments, with different characteristics than lakes. Reservoirs develop longitudinal patterns generally characterized as three distinct zones: riverine, transitional, and lacustrine. The riverine zone is in the headwaters of the reservoirs, and this zone is generally most influenced by external nutrient sources. Thus, the headwater of the reservoir might be an appropriate management target where water quality should be monitored to evaluate long term changes.

The formation of longitudinal and vertical patterns in reservoirs has a significant influence in nutrient availability and primary productivity (White et al., 1991; Tsujimura, 2004). The development of longitudinal patterns can be responsible for the availability of nutrients and primary productivity in the three different zones of the reservoir. Vertical patterns depend on climate (temperature, rainfall) and the season. Therefore, seasonal conditions may determine the nutrient availability in epilimnion and hypolimnion (Kuang et al. (2004). Trophic conditions in the reservoir are determined by the availability of nutrient from external and internal sources. Several studies have shown that nitrogen (N) and/or phosphorus (P) were responsible for limiting algal growth in reservoirs (Dzialowski et al. 2005; Dillon and Rigler, 1974; Maberly et al. 2003). The limiting nutrient of algal growth in reservoirs can shift seasonally within individual reservoirs (Piehler et al., 2003). The amount of algal growth should be related to the magnitude of nutrients available within the reservoir.

Phosphorus is a nutrient that is generally present in a short supply in reservoirs and even a small increase can cause eutrophic problems. For this reason, P was widely studied in many reservoirs to determine the relationship with primary productivity. Several reservoirs have reported that TP and Chl-*a* concentrations had a positive and strong relation and algal growth was limited mainly by the availability of P (Jones and Knowlton, 2005; Guildford and Hecky, 2000; Smith, 1982). However, other studies have shown a positive and significant relationship between Chl-*a* and TN finding also a N limitation for algal growth (Dillon and Rigler, 1974; Jones and Knowlton, 1993; An and Kim, 2003). Sakamoto (1966) and Forsberg et al. (1978) found that Chl-*a* was a logarithmic function of TN and TP. However, this relationship can be highly affected by TN:TP ratios because this promotes to the change in algal species (Smith, 1982; Sakamoto, 1966).

In general, nutrient loadings in the water are responsible increasing trophic levels (eutrophication) and deterioration of water quality in reservoirs. The main consequences are algal blooms, decreased light penetration (loss in water clarity), low dissolved oxygen concentrations in the hypolimnion, shifts of algal species, decrease in reservoir aesthetics, loss in accessibility, interferences in recreational activities, and odor and taste problems that increase costs of drinking water treatment (Dzialowski et al., 2005; USEPA, 2006, Smith, 1999). Thus, the investigation and evaluation of primary production in reservoir is important for developing water quality policies and preventive and corrective actions (Kuang et al., 2004).

The goal of this project was to evaluate the changes in trophic status of the upper reaches of Beaver Lake, AR. The following objectives were formulated in order to accomplish this goal:

- Determine the trophic conditions in the headwaters of Beaver Lake in terms of chlorophyll *a* (Chl-*a*) and nutrient concentrations. The hypothesis to be tested (H_{o1}) was that a marked longitudinal gradient does not exist in trophic conditions at Beaver Lake.
- Evaluate the variability in nutrient (particularly P) concentrations with depth in the headwaters of Beaver Lake. The hypothesis (H_{o2}) was testing if nutrient (particularly P) concentrations do not vary with depth in the transitional zone of Beaver Lake.
- Determine the relationship between Chl-*a* and nutrients with the hypothesis (H_{o3}) that algal (phytoplankton) growth measured as Chl-*a* concentrations is not correlated to the limiting nutrient.
- Evaluate and compare the current conditions versus conditions found in previous studies conducted at Beaver Lake.

- Determine the limiting factor of algal (phytoplankton) growth at Beaver Lake. The hypothesis to be tested (H_{o4}) was that algal (phytoplankton) growth is not limited by nutrients.

METHODS

Water Quality Sampling

The area of study is Beaver Lake located in Northwest Arkansas at the headwaters of the White River (Figure 1.1). Beaver Lake is the youngest reservoir in a chain of four on the White River constructed in the early 1960s with the purpose of flood control, hydropower generation, and water supply. Beaver Lake is also use for fish and wildlife habitat, recreation, and waste assimilation.

Sampling sites and dates. Site selection on Beaver Lake was based on previous studies (Haggard et al., 1999) that evaluated trophic conditions and water quality in the reservoir headwaters. Samples were collected at seven sites once per month for one year, from June 2005 to July 2006; except in August 2005 and January 2006. Three of the sampling sites were in the transitional and the remaining four sites were in the riverine zone. Two of the riverine sites were on White River and two were in the War Eagle Creek (Table 3.1).

Table 3.1. Sampling sites location and description for limnology and trophic state study at Beaver Lake, AR 2005-2006

Site Name	Location	Latitude	Longitude	Zone	Inflow characteristics
BLL	Beaver Lake at Lowell	36°15'33N	94°04'08W	Transitional	None
HCK	Hickory Creek	36°14'23N	94°01'33W	Transitional	None
WTR	White River	36°13'53N	93°59'50W	Transitional	None
WEC	War Eagle Creek	36°11'29N	94°00'04W	Riverine[a]	Inflow affected by agricultural activities and suburban development.
WEI	War Eagle Inlet	36°09'12N	94°00'04W	Riverine[a]	
FCK	Friendship Creek	36°12'45N	93°58'49W	Riverine[b]	Inflow affected by agricultural land, suburban development, and effluent from Fayetteville's wastewater treatment plant.
BSP	Blue Spings	36°13'31N	93°58'31W	Riverine[b]	

[a] War Eagle Creek arm on Beaver Lake

[b] White River arm on Beaver Lake

Sampling procedure. Water samples were taken from each site using an Alpha Sampler horizontal style at the following depths: 1.0, 3.0, and 5.0 m below the surface, 1.0 m below the thermoclime, the middle point of the hypolimniom, and 1.0 m above the reservoir bottom (Figure 3.1). This sampling procedure was not applicable for every site because depths varied and were often too shallow and without a defined thermocline. Therefore, the number of samples collected per site and depths varied from one sampling event to another and from site to site.

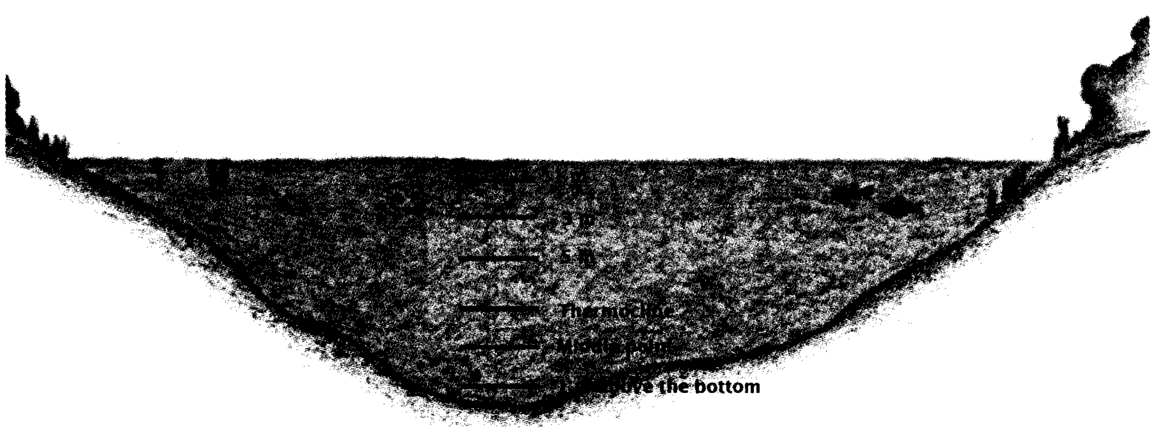


Figure 3.1. Schematic view of depth at which water samples were taken at Beaver Lake during the period of 2005-2006 (not at scale).

At each site sampling and depth, 1L water was collected for chlorophyll-*a* (Chl-*a*) analysis, 250mL unfiltered water was collected and then acidified (pH<2) in order measure total phosphorus (TP), total nitrogen (TN) and total organic carbon (TOC). Approximately 120-mL of filtered (0.45 μm membrane) water was collected and from this volume; 60-mL were acidified with 6N HCl to measure soluble reactive phosphorus (SRP) and ammonia (NH_4-N) and the rest was stored un-acidified to analyzed nitrate (NO_3-N).

Field measures. During sampling events, physico-chemical parameters were measured every meter using a YSI Environmental Multiprobe Model 556. The parameters measured included: dissolved oxygen (DO), conductivity, temperature, and pH. In addition, light was measured using an LI-COR SA type sensor (spherical and quantum sensor and underwater quantum sensor) connected to a LI Multipurpose Data logger, during the sampling events of June 2005 and March, April, May, and July 2006. Turbidity was measured in laboratory after sample collection using an Orbeco-Hellige Model 966 Portable Turbidimeter that measures the clarity or cloudiness of water using Nephelometric Turbidity Units (NTUs). Secchi transparency depth was also measure at each site during each sampling event.

Water Quality Analysis. The 1L water sample collected for Chl-*a* analysis was filtered through a Whatman GFIF (glass fiber filter), and the filter was placed in a glass test tube with 5 mL of 90% aqueous acetone saturated with $MgCO_3$. The filters were ground using a glass stirring rod, and then the glass test tube was centrifuged for 20 minutes. The supernatant was analyzed for Chl-*a*, b, and c using the trichromatic method with a UV/Visible wavelength Spectronic Genesys spectrophotometer (APHA, 1998). Nitrate was measured hydrazine reduction to NO_2-N and colorimetric analysis (APHA, 1998) on a Skalar San Plus wet chemistry autoanalyzer (Skalar, The Netherlands). Soluble Reactive Phosphorus (SRP) was measured using the automated ascorbic acid method (APHA, 1998) on the Skalar San Plus wet chemistry autoanalyzer. Ammonia was measured using the Salicylate Method and colorimetric method (660 nm) on a spectronic Genesys spectrophotometer (APHA, 1998). Total Phosphorus was measured by digesting 25-mL aliquot with the Persulfate Method 4500-N C (Autoclave Digestion Method, APHA, 1998). After the digestion, TP was determined using the ascorbic acid method for SRP, this is a colorimetric method which measures absorbance at 880 nm using a spectronic Genesys spectrophotometer. Total Nitrogen and TOC were measured by combustion using the Shimadzu TOC-VCSH with a TNM-1 TN analyzer and an ASI-V auto sampler.

Table 3.2. Summary of analytical methods use to determine nutrient concentrations on Beaver Lake during 2005-2006.

Parameter	Sample specification	Analytical Method
Ammonia (NH_4-N)	Filtered and acidified	Salicylate Method (APHA, 1998) Spectronic Genesys Spectrophotometer (Rochester, NY)
Nitrate-N (NO_3^--N)	Filtered and un-acidified	Hydrazine reduction and colorimetric analysis on a Skalar Sanh Wet Chemistry Autoanalyzer.
Soluble Reactive Phosphorus (SRP)	Filtered and acidified	Ascorbic acid method on a Skalar Sanh Wet Chemistry Autoanalyzer.
Total Phosphorus (TP)	Unfiltered and acidified	(1) Autoclave Digestion Method: Persulfate Method 4500-N C (APHA, 1998) (2) Ascorbic Acid Method 4500-P E. Soluble Reactive Phosphorus (APHA, 1998) Spectronic Genesys Spectrophotometer (Rochester, NY)
Total Nitrogen (TN)	Unfiltered and acidified	Combustion Shimadzu TOC-VCSH with a TNM-1 TN analyzer ASI-V auto sampler
Total Organic Carbon (TOC)	Unfiltered and acidified	Combustion Shimadzu TOC-VCSH with a TNM-1 TN analyzer ASI-V auto sampler
Chl-*a*	Unfiltered and un-acidified	Trichromatic method (APHA, 1998) Spectronic Genesys Spectrophotometer (Rochester, NY)

Nutrient Limitation Studies

Only one of the seven sites was selected to evaluate nutrient limitation of phytoplankton. The site selected for this study was near Hickory Creek (HCK), because this site is close to the end of the transitional zone and the beginning of the riverine zone of Beaver Lake. Nutrient limitation of phytoplankton was evaluated five times: July, September, and November 2005 and January and May 2006.

Sampling procedure. Approximately 303L (80gal) of lake water were collected at the HCK station from just under the surface. Reservoir water was collected in 190L capacity dark containers and transported to the laboratory facilities. Then, approximately three liters of water were transferred from the containers to 3.7L (1 gallon) cubitainers (as show in Figure 3.2) using a submersible pump.

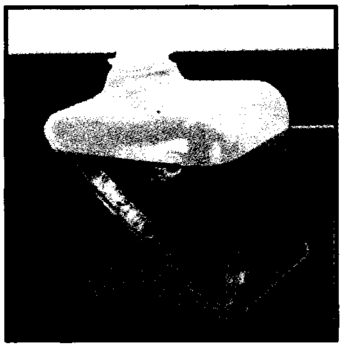


Figure 3.2. 3.71L (1 gallon) cubitainer fill out with approximately 3L of water from Beaver Lake for nutrient limitation analysis.

Water samples were collected from the larger containers in order to determine Chl-*a* concentration before nutrient addition and incubation. The method used to analyze Chl-*a* was the same described previously.

Incubation procedure. After the 3L of reservoir water was transferred into the cubitainers, the cubitainers were spiked with different concentrations of PO_4-P and NO_3-N in order to create an array of treatments of PO_4-P and NO_3-N, including a control at ambient conditions (Table 3.3 and 3.4). Each treatment was replicated 5 times for a total of 80 cubitainers. Once each cubitainer was spiked and placed in the greenhouse in water baths for temperature and irradiance control.

Table 3.3. Summary of nitrate and phosphate concentrations applied in the algal potential growth test.

Treatment	NO_3-N [ug/L]	PO_4-P [ug/L]
Ambient	Variable	Variable
Low	100	30
Medium	500	50
High	1000	100

Table 3.4. Four by four matrix for all possible nutrient combinations for bioassay experiment for nutrient limitation analysis in Beaver Lake.

Treatments		PO_4-P Ambient	Low	Medium	High
NO_3-N	**Ambient**				
	Low				
	Medium				
	High				

The first incubation was for seven days in order to determine the period in which the maximum algal response to nutrient enrichment occurred. Samples were analyzed daily for seven days to determine the critical incubation period. Based upon the results, a period of four days was selected for other cubitainer incubations. During the incubation period, each cubitainer was inverted twice every day to minimize settling of biomass and was opened to promote gas exchange.

At the end of incubations water samples were collected from each cubitainer. Approximately 500mL of water was filtered (Whatman GF/F glass fiber filter) for Chl-*a* analysis. Also, 250mL of water was collected and stored for alkalinity; 30mL was filtered (0.45μm membrane) for PO_4-

P and NO_3-N and 60mL was filtered (0.45µm membrane) and acidified for later analysis of NH_4-N. During the incubation period, water temperature in the ponds and light intensity in the green house were measured using HOBO micro-stations (Onset Computer Corporation) equipped with Photosynthetically Active Radiation (PAR) and temperature sensors.

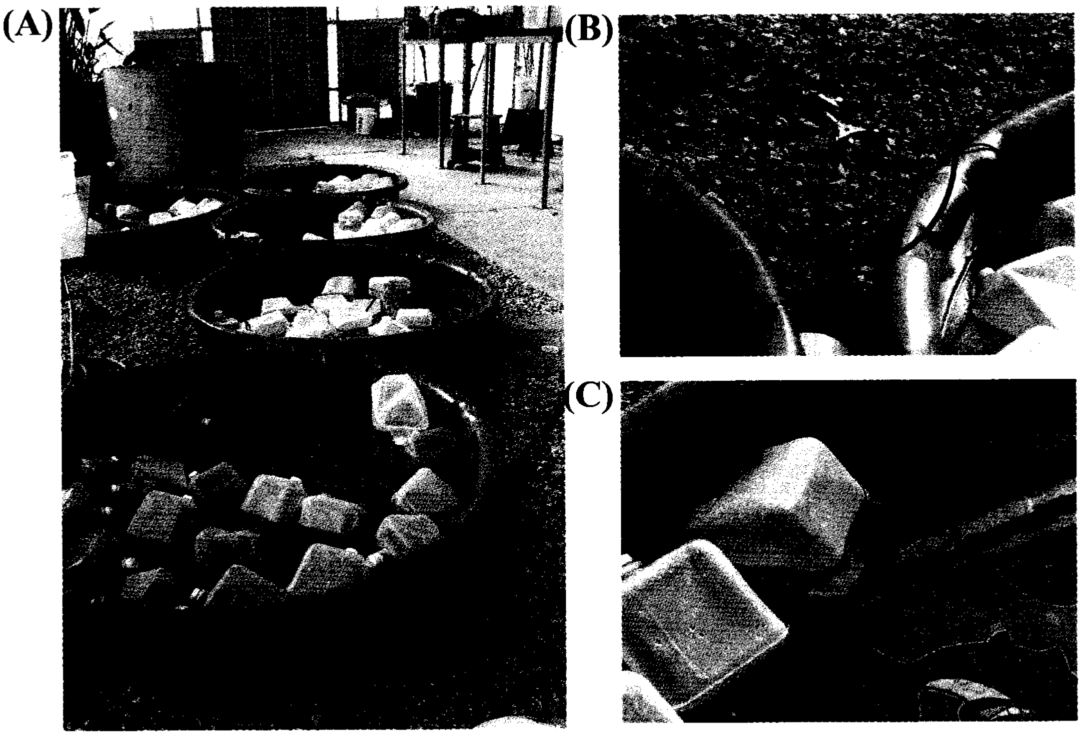


Figure 3.3. (A) Cubitainers used for bioassay, placed in pools for temperature regulation in a greenhouse. (B) and (C) HOBO temperature and light micro-loggers used to measure PAR and temperature during the nutrient enrichment experiment for Beaver Lake (2005-2006).

Analytical analysis. For Chl-*a* analysis, 500mL of samples from each cubitainer was filtered, stored, ground, and analyzed as previously described. NO_3-N and PO_4-P were analyzed using ion chromatography

(761 Compact IC, Metrohm Peak), and NH_4-N was measured as previously described. Alkalinity was measured using the Titration Method 2320 B (APHA, 1998).

Table 3.5. Summary of analytical methods for nutrient analysis for Beaver Lake trophic status evaluation for 2005-2006.

Parameter	Sample specification	Analytical Method
Ammonia (NH_4-N)	Filtered and acidified	Automated Phenate Method 4500-NH_3 (APHA, 1998) Spectronic Genesys Spectrophotometer (Rochester, NY)
Nitrate-N (NO_3^--N) and Phosphate (PO_4-P)	Filtered and un-acidified	Ion chromatography 761 Compact IC from Metrohm Peak
Chl-*a*	Unfiltered and un-acidified	Trichromatic method (APHA, 1998) Spectronic Genesys Spectrophotometer (Rochester, NY)
Alkalinity	Unfiltered and un-acidified	Titration Method: 2320 B (APHA, 1998)

Statistical Analysis

Seasonal changes. Chl-*a* and nutrient mean concentrations were determine in the photic zone (1 -3 m) per sampling event at each sampling station. The data was analyzed using analysis of variance (ANOVA). When significance difference was identified, means were compared using Each Pair Student's *t*–test Least Significant Difference (LSD) with a confidence level of 0.1 using JMP 6.0 (2006 SAS Institute, Cary, NC, USA).

Chl-*a* and nutrient concentration relationship. Chl-*a* and nutrient concentrations were calculated at the photic zone (1-3 m). Then annual means were calculated and Log_{10} transformed per each sampling site. Linear regression between Chl-*a* concentrations and nutrients were determine and the significance was find with ANOVA analysis of variance ($\alpha = 0.1$) using JMP 6.0.

Longitudinal gradient. Mean values were calculated for Chl-*a* and nutrients for each site during the stratified and non-stratified seasons. The depth considered to calculate the mean depended upon the deepest sampling point in the shallower site at the riverine zone. The shallower site considered was War Eagle Creek (WEC) which for most of the sampling events the deepest sampling point was 3-m; except for July, September 2005, and June 2006 where the deepest point was at 5-m. The stratified period include the months of: July, September, October 2005 and April, May, June, July 2006. The non-stratified period included the months of: November and December 2005 and February and March 2006. The means per each site and parameter were analyzed using ANOVA analysis of variance and Each Pair Student's *t*–test Least Significant Difference (LSD) using JMP 6.0 with $\alpha = 0.1$.

Vertical gradient. The vertical gradient of nutrients were analyzed by calculating the means of each parameter for the epilimnion (sampling points above the thermocline) and hypolimnion (sampling points bellow the thermocline) for the stratified and non-stratified season. The analysis was for overall results and individual results per sampling site. Means were evaluated using ANOVA analysis of variance and when significant differences were found the means were separated using Each Pair

Student's *t*–test Least Significant Difference (LSD) using JMP 6.0 with $\alpha = 0.1$.

Trophic state. Using the means obtained for Chl-*a*, TP, and SD at the photic zone Carlson's TSI values were calculated for each sampling station. The TSI values and mean concentrations were evaluated against the criteria specified by Carlson according to productivity, P concentration, and transparency. The equations used to determine the trophic conditions were (Carlson, 1977):

$$TSI\,(SD) = 10\left(6 - \frac{\ln SD}{\ln 2}\right)$$

$$TSI\,(Chl) = 10\left(6 - \frac{2.04 - 0.68 \ln Chl}{\ln 2}\right)$$

$$TSI\,(TP) = 10\left(6 - \frac{\ln \frac{48}{TP}}{\ln 2}\right)$$

Where:	
SD,	Secchi disc transparency [m]
TP,	Total Phosphorus concentrations in the photic zone (1-3 m) [ug L^{-1}]
Chl,	Chlorophyll-a concentrations in the photic zone (1-3 m) [ug L^{-1}]

Nutrient limitation evaluation. Mean concentrations for Chl-*a* were determine by using all the replicates in each experiment. The means for all the treatments were compared using using Each Pair Student's *t*–test Least Significant Difference (LSD) using JMP 6.0 with $\alpha = 0.1$. Finally, the Control, high N, high P and high NP treatments were compared and the analysis of variance was evaluated using ANOVA in JMP 6.0.

RESULTS AND DISCUSSION

Water Quality

Temperature and dissolved oxygen

Thermal stratification is one of the main characteristics of reservoirs in the southern U.S. Dissolved oxygen and temperatures measurements taken at every meter at each sampling station indicated the periods and extent of stratification. The dissolved oxygen and temperature profiles for four seasons starting from fall 2005 throughout summer of 2006 at Beaver Lake at Lowell (transitional zone) illustrate the profile of the reservoir (Figure 3.4). Stratification patterns in Beaver Lake are similar to those described in James et al. (1987) and a traditional reservoir limnology. Stratification begins in early spring (April) and mixes occurs in late fall (November). The thermocline is typically at about 6m depth below the water surface during the summer.

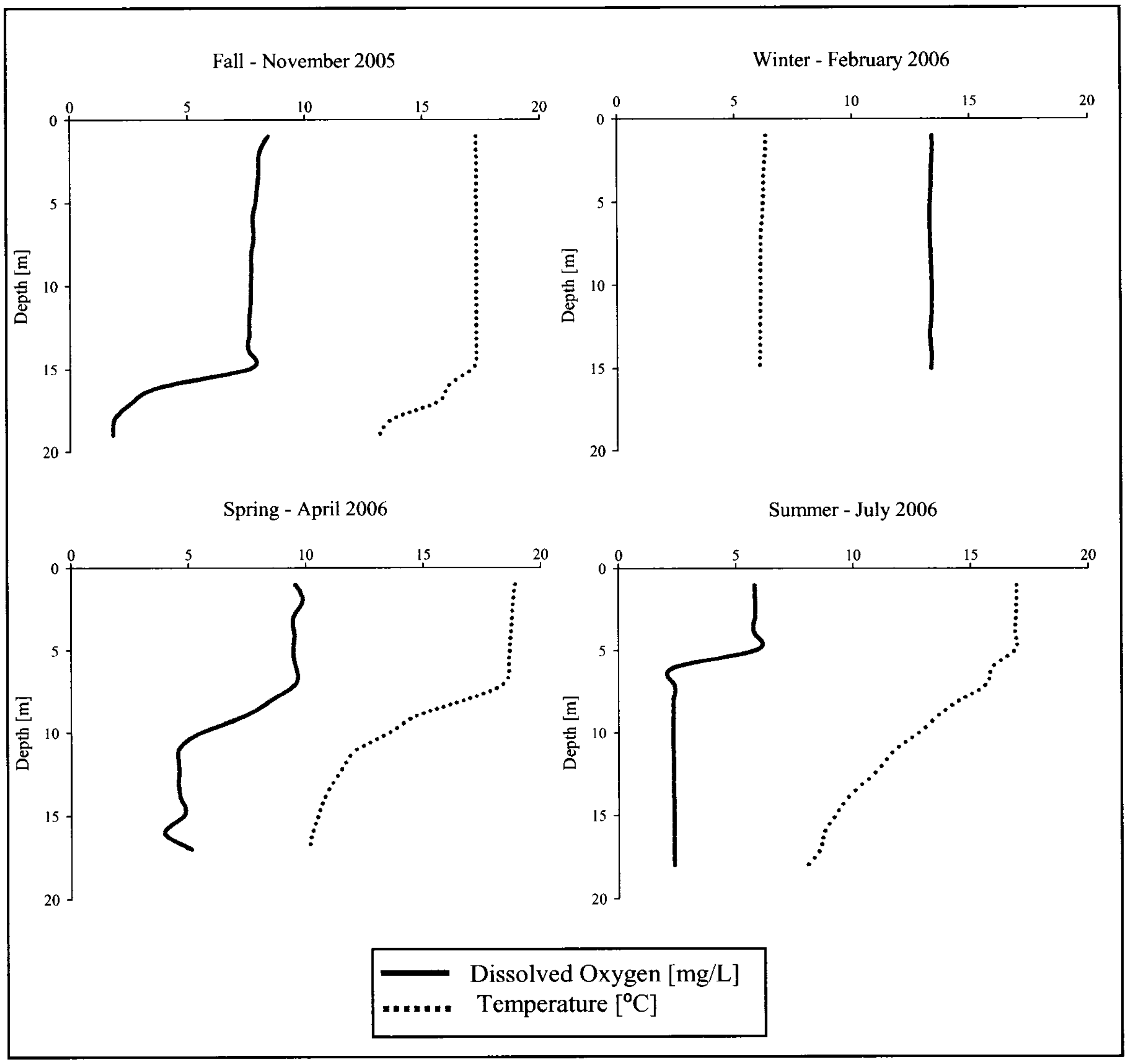


Figure 3.4. Temperature and Dissolved Oxygen concentrations recorded at every meter at Beaver Lake at Lowell (transitional zone) during November 2005, February, April, and July 2006.

Nitrogen

Nitrogen concentrations were measured as TN, NO_3-N and NH_4-N in water samples collected at all sites at Beaver Lake during 2005-2006. The concentrations for the different forms of N were analyzed per sampling date and statistical differences were found for all N forms. In general, TN concentrations in the photic zone varied from 181ug/L in July 2006 to 658ug/L in March 2006. Nitrate-N concentrations varied from 2ug/L in July 2006 to 95ug/L in March 2006. Finally, NH_4-N varied from 24ug/L in July 2005 to 65ug/L in December 2005.

All N forms analyzed in Beaver Lake presented some seasonal changes. Total N and NO_3-N concentrations were greatest during late winter and spring 2006 with ranges of 658ug/L – 596ug/L for TN and 96ug/L - 85ug/L for NO_3-N (Table 3.6). Elevated N concentrations, especially NO_3-N, might be attributed to contributions from episodic storm events that delivered N to the headwaters of the reservoir and possible completely mixed conditions after the fall overturn. In addition, high N concentrations may be attributed to a reduction in the N uptake rates because of the low temperature during the winter time. Generally, during the summer, the N concentrations decrease and it might be attributed to an increment in primary productivity uptake. Ammonia presented the greatest concentrations during the winter time probably because of reduced nitrification rates at lower temperatures. No seasonal gradients were found in N concentrations at Beaver Lake as observed in previous studies (Haggard et al., 1999). However, other studies have reported similar seasonal trends in other reservoirs (White et al., 1991).

Table 3.6. Nitrate-N (NO_3-N), Total Nitrogen (TN), and Ammonia-N (NH_4-N) mean concentrations and standard deviation in the photic zone per sampling date for Beaver Lake, 2005-2006.

Sampling Date	Mean NO_3-N [ug/L]	Standard Deviation NO_3-N [ug/L]	Mean TN [ug/L]	Standard Deviation TN [ug/L]	Mean NH_4-N [ug/L]	Standard Deviation NH_4-N [ug/L]
July, 2005	3	4	292	80	24	15
September, 2005	2	1	306	78	31	5
October, 2005	3	3	196	87	23	14
November, 2005	14	14	366	184	48	46
December, 2005	42	28	440	302	65	25
February, 2006	87	53	596	341	30	20
March, 2006	95	43	658	260	33	1
April, 2006	85	37	598	252	35	23
May, 2006	85	33	604	155	35	22
June, 2006	28	7	318	39	48	23
July, 2006	2	0	181	30	33	11

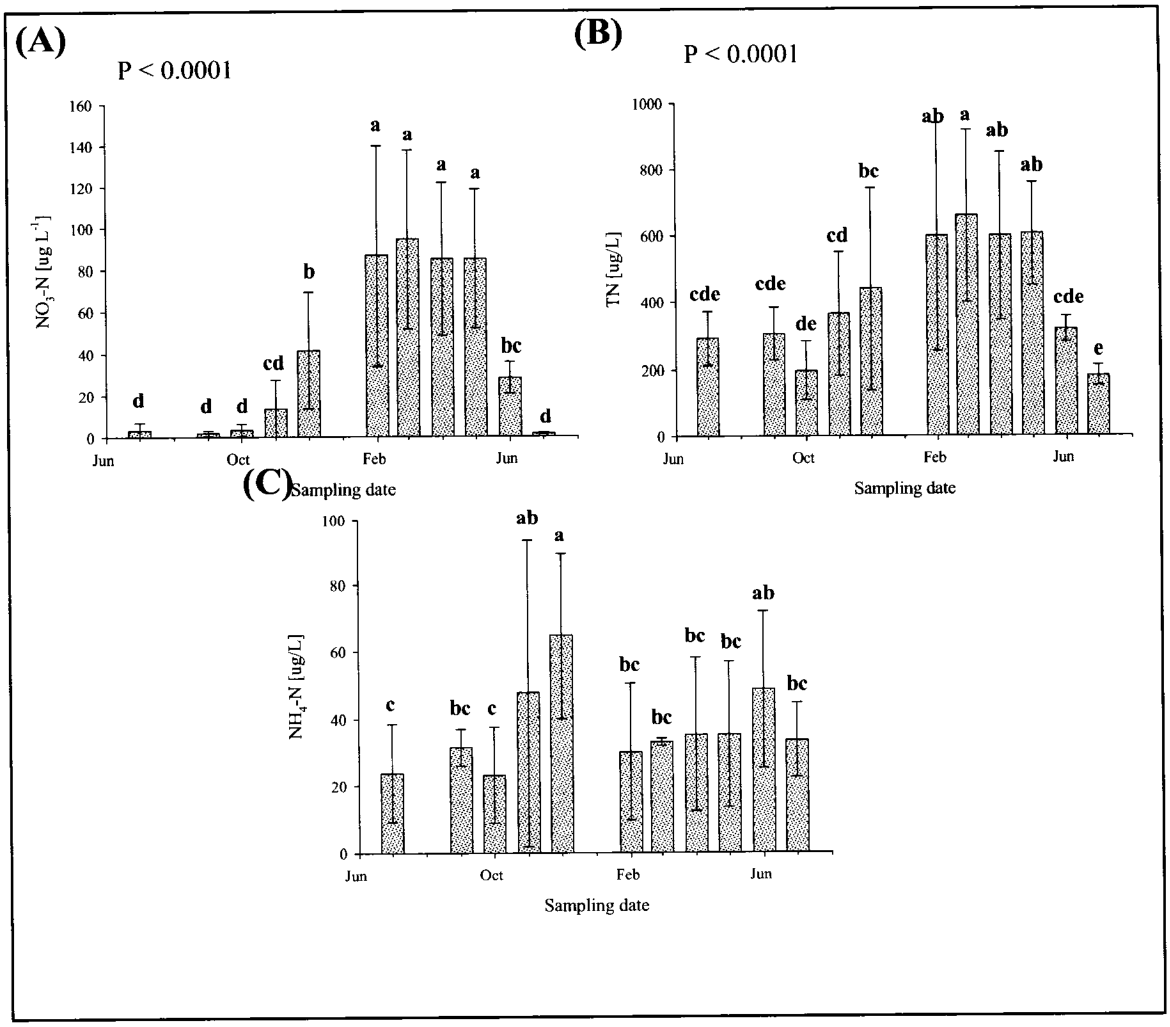


Figure 3.5. (A) Nitrate (NO_3-N), (B) Total Nitrogen (TN), and (C) Ammonia (NH_4-N) mean concentrations (plus or minus standard deviation) per sampling date at the photic zone (1-3 m) in Beaver Lake. Significant differences denoted by letters on top of each bar (a, b, c, d, e) at $\alpha = 0.10$.

Based on sampling dates it was evident that N concentrations were relatively constant during the stratified period. Ammonia concentrations showed a peak after the fall turnover and decreased during the stratified period (Figure 3.5a, b, and c). During the non-stratified period TN concentrations were not significantly different by date (P>0.1). During the non-stratified season, the presumption is that N concentrations increased after the fall turnover gradually throughout the winter time (mixed conditions). Matthews et al. (2001) in Lake Whatcom, Washington considered that TN concentrations presented consistent seasonal trends where the lower concentrations were observed from late summer throughout fall.

Beaver Lake did not show a marked longitudinal pattern for N forms during the stratified season (Table 3.7), since the differences in concentrations for this period were not significant ($P>0.1$). However, during the non-stratified season Beaver Lake presented a longitudinal gradient for TN, where concentrations were greater in the riverine zone (FCK and BSP) (Figure 3.6). The greater concentrations present in the riverine zone may be attributed to external loadings that have a direct effect over the headwaters of the reservoir. In addition, since the riverine zone is shallow the water column tented to be well mixed and N present in the sediment can be released. In this case, the sites located on the White River arm showed the greatest concentrations of N. This might be attributed to the wastewater treatment plant discharges. Nitrogen concentrations appear to become diluted with increasing distance from the headwaters in the White River arm and the effluent discharge.

Table 3.7. Nitrate-N (NO_3-N) and Total Nitrogen mean concentrations and standard deviation in the photic zone per sampling station for Beaver Lake, 2005-2006.

		Stratified Period				Non-stratified Period			
Sampling site	**Distance [km]**	**Mean NO_3-N [ug/L]**	**Standard Deviation NO_3-N [ug/L]**	**Mean TN [ug/L]**	**Standard Deviation TN [ug/L]**	**Mean NO_3-N [ug/L]**	**Standard Deviation NO_3-N [ug/L]**	**Mean TN [ug/L]**	**Standard Deviation TN [ug/L]**
BLL	22	25	27	310	150	36	17	331	61
HCK	30	27	35	302	160	35	25	355	100
WTR	37	25	32	306	158	42	33	381	146
FCK	45	19	23	331	113	97	87	767	440
BSP	50	24	30	402	175	87	37	814	188
WEC	40	31	47	409	216	56	48	419	206

Nitrogen concentrations were also analyzed by depth during the stratified and non-stratified periods (Table 3.8). During the stratified period statistical analysis showed that NO_3-N concentrations in the epilimnion were not significantly different from the hypolimnion. This may be attributed to NO_3-N uptake for primary productivity in the epilimnion and denitrification in the hypolimnion under anaerobic conditions. De Medina et al. (2002) reported similar conditions in the analysis of N concentration in sediments and overlying water during anaerobic conditions in Lake Maracaibo, Venezuela. The difference in TN concentrations in the epilimnion and hypolimnion was significantly different ($P<0.001$), having the greatest concentrations in the hypolimnion (Figure 3.7). This may be attributed to the internal loading and production of N at the bottom of the reservoir where N may be released from the sediment under anaerobic conditions as NH_4-N. During the non-stratified period NH_4-N concentrations were greater in the hypolimnion but TN and NO_3-N did not present any significant difference.

Table 3.8. Nitrate-N (NO_3-N), Total Nitrogen (TN), and Ammonia (NH_4-N) mean concentrations and standard deviation in the epilimnion and hypolimnion per sampling station for Beaver Lake during the stratified period.

Site name	Depth [m]	Mean NH_4-N [ug/L]	Standard Deviation NH_4-N [ug/L]	Mean NO_3-N [ug/L]	Standard Deviation NO_3-N [ug/L]	Mean TN [ug/L]	Standard Deviation TN [ug/L]
BLL	Epilimnion	76	126	28	32	313	165
	Hypolimnion	43	23	54	48	613	113
HCK	Epilimnion	34	18	32	42	318	185
	Hypolimnion	42	32	44	54	672	243
WTR	Epilimnion	36	25	30	39	324	193
	Hypolimnion	52	28	48	62	607	315
FCK	Epilimnion	32	11	20	25	322	107
	Hypolimnion	39	19	32	44	430	161
BSP	Epilimnion	24	10	27	31	382	131
	Hypolimnion	41	35	48	73	627	381
WEC	Epilimnion	34	27	32	49	401	211
	Hypolimnion	42	24	25	36	441	257

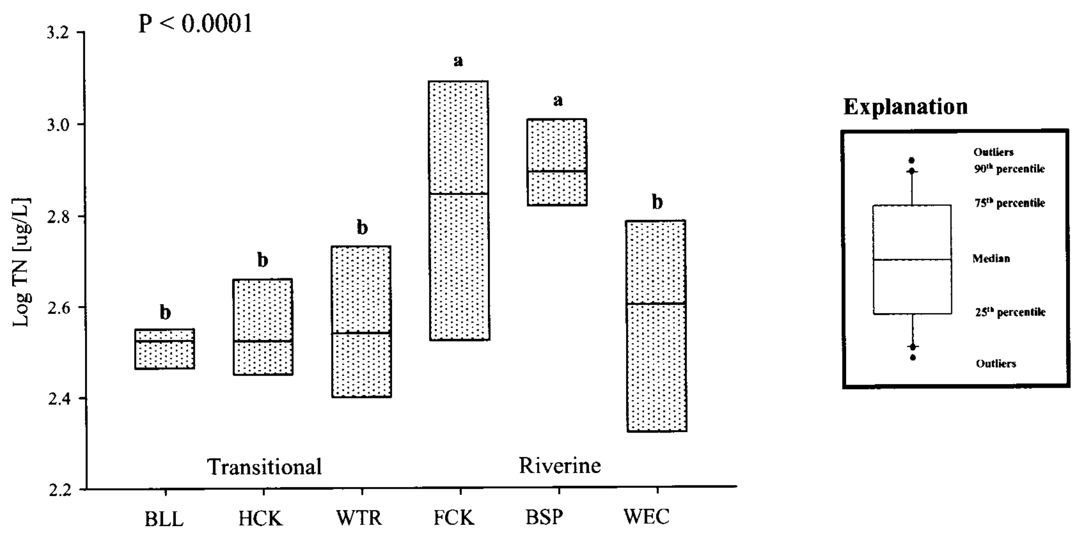


Figure 3.6. Box plots of Total Nitrogen (TN) concentrations at the photic zone (1 -3 m) by sampling site during the non-stratified season in Beaver Lake and P-value calculated using ANOVA analysis with $\alpha = 0.10$. Letters denote the significant differences (P<0.0001) for Log_{10} transformed data.

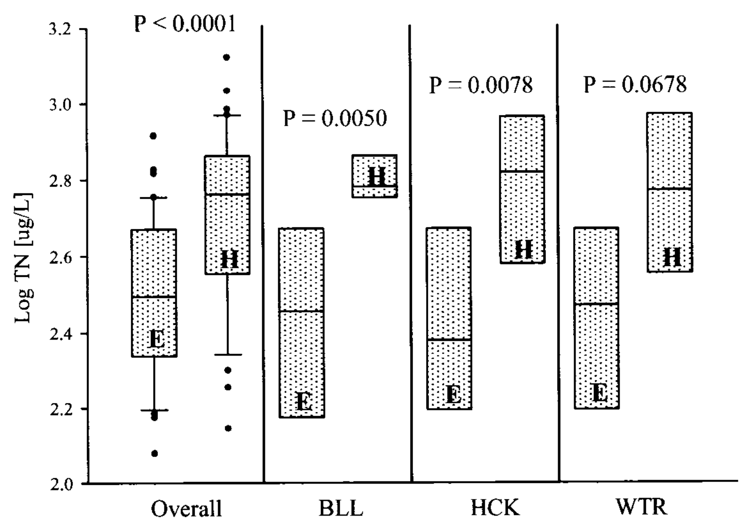


Figure 3.7. Box plot of pair wise comparison of Total Nitrogen (TN) concentrations calculated for the epilimnion and hypolimnion for individual sites and the whole reservoir during the stratified period. The letters E and H represent the epilimnion and hypolimnion, respectively. P values were calculated using ANOVA analysis with $\alpha = 0.10$ for Log_{10} transformed data.

In addition to the overall analysis for Beaver Lake, individual sampling sites were evaluated in order to determine changes per zone. During the stratified season sites located at the transitional zone Beaver Lake at Lowell (BLL), Hickory Creek (HCK), and White River (WTR) had significantly greater TN concentrations in the hypolimnion than in the epilimnion ($P<0.1$) (Table 3.8). None of the sites located in the riverine zone showed significant differences in concentrations with depth (epilimnion and hypolimnion). During the stratified season, dissolved N concentrations in the epilimnion get depleted faster because primary productivity takes place in this layer under favorable conditions. During the non-stratified season none of the sampling stations presented significant differences of N concentrations in the epilimnion and hypolimnion because the reservoir is well mixed.

Phosphorus

During this study P concentrations were analyzed as SRP, TP, and Particulate Phosphorus (PP). Phosphorus concentrations can change from one sampling date to another depending on the season and stratification conditions in reservoirs (Table 3.9). Total phosphorus concentrations in the photic zone varied from 34ug/L in July 2005 to 80ug/L in September 2005. Haggard et al. (1999) reported that TP displayed two peaks during the spring and the fall. In the present study, SRP values varied from 1ug/L in July 2006 to 12ug/L in July 2005 and April 2006. Annual means were determined for Beaver Lake in order to determine seasonal patterns.

Between all P forms measured, only SRP values presented a significant difference per sampling date (Figure 3.8).

Table 3.9. Soluble Reactive Phosphorus (SRP), Total Phosphorus (TP), and Particulate Phosphorus (PP) mean concentrations and standard deviation in the photic zone per sampling date for Beaver Lake, 2005-2006.

Sampling Date	Mean SRP [ug/L]	Standard Deviation SRP [ug/L]	Mean TP [ug/L]	Standard Deviation TP [ug/L]	Mean PP [ug/L]	Standard Deviation PP [ug/L]
July, 2005	12	4	34	10	25	13
September, 2005	6	2	80	70	73	72
October, 2005	7	6	38	12	36	17
November, 2005	5	3	52	22	47	22
December, 2005	5	1	37	17	32	18
February, 2006	7	7	46	11	40	13
March, 2006	5	1	43	20	38	20
April, 2006	12	10	71	52	59	47
May, 2006	6	2	60	19	55	19
June, 2006	3	3	49	5	45	4
July, 2006	1	1	56	26	55	25

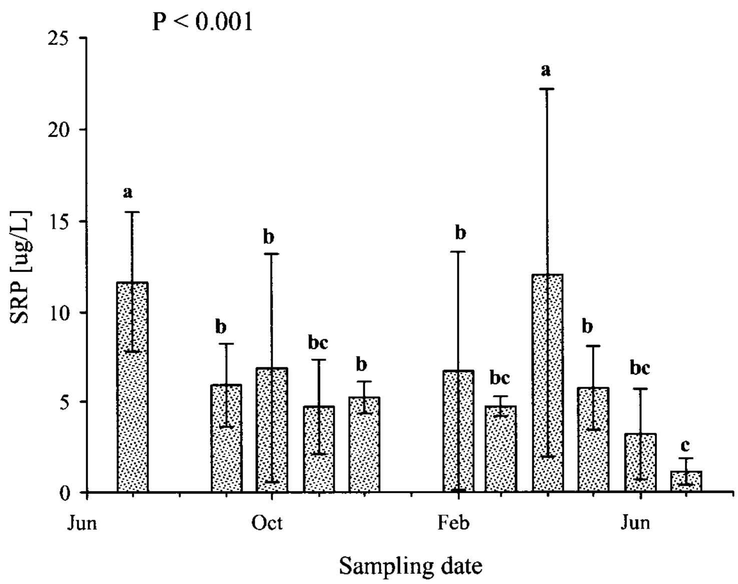


Figure 3.8. Soluble Reactive Phosphorus (SRP) mean concentrations (plus and minus standard deviation) per sampling date at the photic zone (1-3 m) in Beaver Lake. Significant differences denoted by letters on top of each bar (a, b, c) at $\alpha = 0.10$ ($P<0.001$).

Soluble reactive phosphorus was the only form of P that presented a significant different in concentrations over sampling date (P=0.0008) (Figure 3.7). However, in general, P concentrations in the reservoir were low and the spikes presented in July 2005 and April 2006 were likely from storm events. Randolph and Wilhm (1984) and Matthews et al. (2001) found and reported similar patterns for P seasonality.

Longitudinal patterns were also evaluated for P concentrations (Table 3.10). During the stratified period none of the P forms presented a significant longitudinal difference. However, during the non-stratified period SRP presented a longitudinal difference where the greater concentrations were found in riverine zone (Figure 3.9). The increment of P concentrations in this zone may be attributed to adsorption and desorption of P by bottom sediments and the tendency to reach equilibrium P concentrations in the water column. Since sampling sites located in the riverine zone were shallower, P released from bottom sediments might result from resuspension induced by wind. Haggard et al. (1999) reported longitudinal changes for SRP and TP decreasing the concentrations from the headwaters to the lacustrine zone. In addition, the authors reported that TP did not present a longitudinal gradient during the winter time of the second year of study. The present study did not find a longitudinal gradient for this parameter during any season.

Table 3.10. Soluble Reactive Phosphorus (SRP), Total Phosphorus (TP), and Particulate Phosphorus (PP) mean concentrations and standard deviation in the photic zone per sampling station for Beaver Lake, 2005-2006.

		Stratified Period						Non-stratified Period					
Sampling site	**Distance [km]**	**Mean SRP [ug/L]**	**Standard Deviation SRP [ug/L]**	**Mean TP [ug/L]**	**Standard Deviation TP [ug/L]**	**Mean PP [ug/L]**	**Standard Deviation PP [ug/L]**	**Mean SRP [ug/L]**	**Standard Deviation SRP [ug/L]**	**Mean TP [ug/L]**	**Standard Deviation TP [ug/L]**	**Mean PP [ug/L]**	**Standard Deviation PP [ug/L]**
BLL	22	7	6	37	17	30	17	4	2	45	26	40	25
HCK	30	8	10	41	19	33	23	6	3	44	16	37	16
WTR	37	6	4	42	21	36	21	5	1	49	12	45	12
FCK	45	6	4	64	93	58	93	3	2	50	21	47	21
BSP	50	4	4	53	30	48	31	10	11	54	32	40	38
WEC	40	9	11	51	16	39	22	4	1	38	21	34	21

Stratified and non-stratified conditions are, also, responsible for differences in nutrient concentrations in the epilimnion and hypolimnion (Table 3.11). The results for Beaver Lake showed that SRP concentrations are significantly greater in the hypolimnion (P=0.077) during the stratified period (Figure 3.10a). The individual analysis of every site showed that in the transitional zone TP and SRP presented significantly greater concentrations in the hypolimnion at Beaver Lake at Lowell (P=0.0475 and P=0.0611, respectively) and Hickory Creek (P=0.0486 and P=0.0814, respectively) (Figure 3.10). The non-stratified season did not present any significant differences between concentrations in the epilimnion and hypolimnion. The greater concentrations of P in the hypolimnion may be attributed to P internal loading produced under anoxic conditions in the hypolimnion during the stratified period. Randolph and Wilhm (1984) and Gloss et al. (1980) found similar patterns in the evaluation of Lake Powell during summer and spring (stratified season) finding that P gets depleted in the epilimnion due to phytoplankton uptake and the lack of P regeneration.

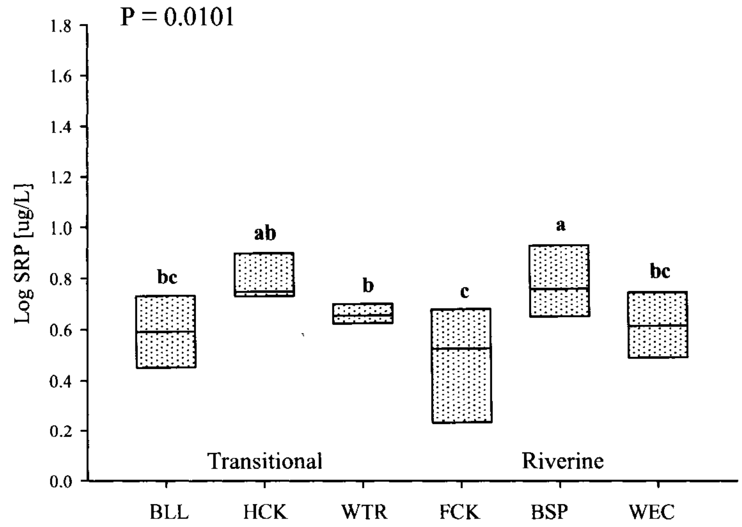


Figure 3.9. Box Plots of Soluble Reactive Phosphorus (SRP) concentrations at the photic zone (1 -3 m) by sampling site during the non-stratified season in Beaver Lake and P-value calculated using ANOVA analysis with $\alpha = 0.1$ for Log_{10} transformed data.

Table 3.11. Soluble Reactive Phosphorus (SRP), Total Phosphorus (TP), and Particulate Phosphorus (PP) mean concentrations and standard deviation in the epilimnion and hypolimnion per sampling station for Beaver Lake during the stratified period.

Site name	Depth [m]	Mean SRP [ug/L]	Standard Deviation SRP [ug/L]	Mean TP [ug/L]	Standard Deviation TP [ug/L]	Mean PP [ug/L]	Standard Deviation PP [ug/L]
BLL	Epilimnion	7	5	38	12	32	14
	Hypolimnion	31	31	56	19	34	17
HCK	Epilimnion	8	7	37	9	30	14
	Hypolimnion	34	39	76	57	49	30
WTR	Epilimnion	6	3	48	24	42	24
	Hypolimnion	15	14	43	13	33	7
FCK	Epilimnion	6	3	74	72	68	73
	Hypolimnion	8	4	47	21	39	20
BSP	Epilimnion	4	3	54	20	50	22
	Hypolimnion	6	7	47	27	41	23
WEC	Epilimnion	7	5	50	23	44	22
	Hypolimnion	13	17	49	13	36	21

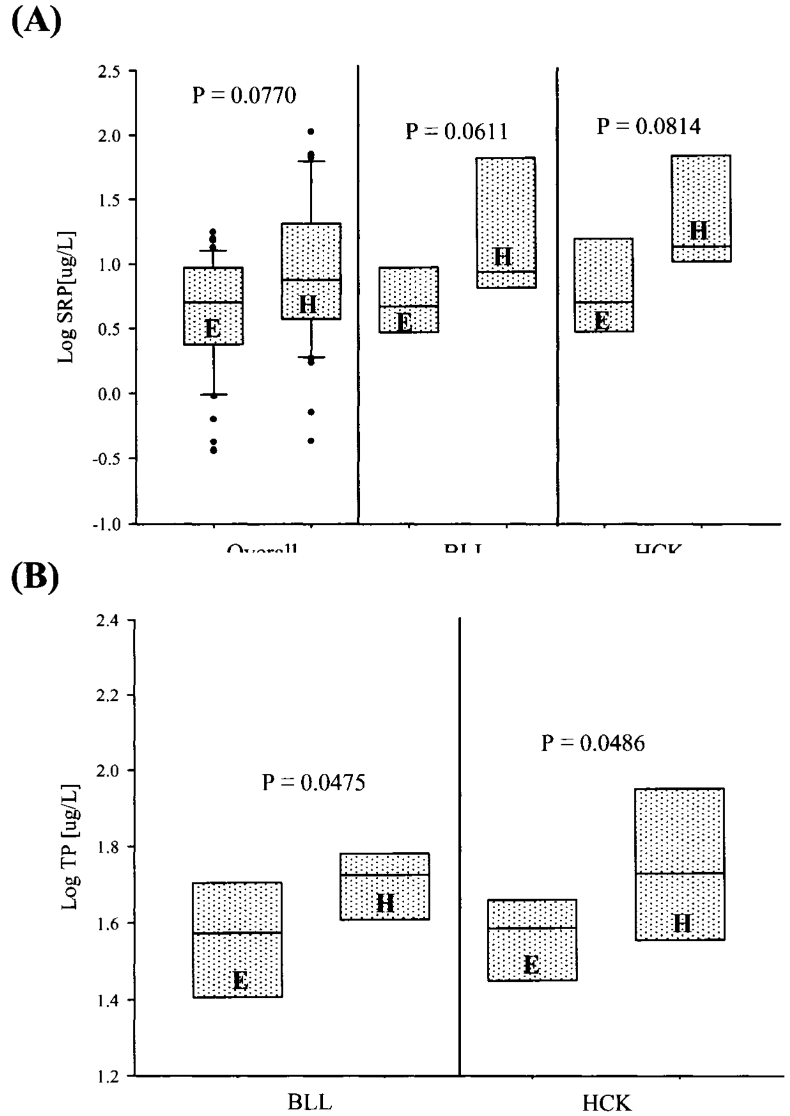
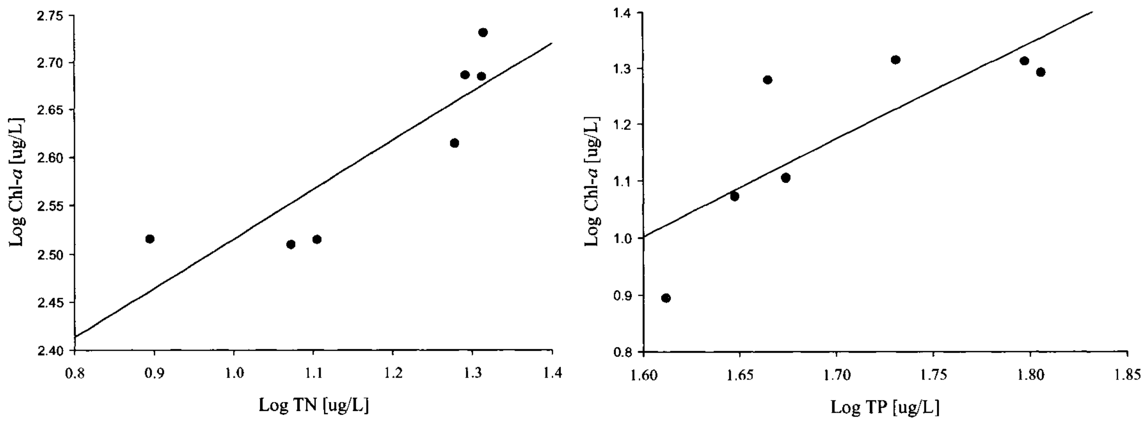


Figure 3.10. (A) Box Plot of pair wise comparison for Soluble Reactive Phosphorus (SRP) and (B) Box Plot of pair wise comparison for TP mean concentrations calculated for the epilimnion and hypolimnion for individual sites and the whole reservoir during the stratified period and P-values calculated using ANOVA analysis with $\alpha = 0.1$ for Log_{10} transformed data.

Phytoplankton production

Chlorophyll concentration is widely use as a measurement of primary productivity, being, Chl-*a* concentrations equivalent to primary productivity. Algae growth and nutrient relationships has been extensively studied in different natural and artificial impoundments with diverse characteristics. The present study found a relationship between Chl-*a* concentrations and TN and TP concentrations respectively, on an annual basis across the sampling stations (Figure 3.11). Both Log-linear regressions show a positive strong and significant relationship between Chl-*a* and TN and TP, respectively. The study of Beaver Lake in 1994-1995 found a Chl-*a* and TP and PP log-linear relationship for Year 1 (1993-1994) (Haggard et al. 1999). In addition, the soluble nutrients (SRP and NO_3-N) presented a decreasing log-linear trend and NH_4-N presented a positive relationship. According to Haggard et al. (1999) study Total Kjeldhal Nitrogen (TKN) was the nutrient that explain the greatest portion of the variance in Chl-*a* concentrations. Dillon and Rigler (1974) and Randolph and Wilhm (1984) have found relationships similar at the one presented in Beaver Lake during 2005-2006. According to the data, TN and TP present significant correlation with Chl-*a* ($P<0.1$) (Figure 3.11). When both TN and TP were included in the multi-linear regression, the results showed that these two variables explain approximately 79% of the Chl-*a* variability ($P=0.0431$). However, the slope p-values were not significant ($P>0.1$).



Log Chl-*a* = -2.86 + 1.57 Log TN

$R^2 = 0.69$

P = 0.02

Log Chl-*a* = -4.94 + 3.66 Log TP

$R^2 = 0.71$

P = 0.02

Figure 3.11. Log linear relationship between Chlorophyll-*a* (Chl-*a*), Total Phosphorus (TP), and Total Nitrogen (TN) means by sampling station in the photic zone (1-3 m) of Beaver Lake. P-values were calculated using ANOVA analysis with $\alpha = 0.10$ for Log_{10} transformed data.

Chlorophyll-*a* concentrations during the study showed a significant difference between sampling dates (P=0.0243) (Figure 3.12). The greatest concentration was found during summer (June 2006) and the least concentration was observed during early spring (April 2006) (Table 3.12). The least concentrations observed in April may be attributed to the sampling after a storm event which may cause high turbidity. Beaver Lake presented the same Chl-*a* seasonal patterns in the study reported by Haggard et al. (1999). According to Haggard et al. (1999), the growing season for algae on Beaver Lake started mid-May and continued through summer.

Table 3.12. Chlorophyll-*a* (Chl-*a*) mean concentrations and standard deviation in the photic zone per sampling date for Beaver Lake, 2005-2006.

Sampling Date	**Mean Chl-*a* [ug/L]**	**Standard Deviation Chl-*a* [ug/L]**
July, 2005	15	10
September, 2005	20	8
October, 2005	19	10
November, 2005	15	10
December, 2005	17	7
February, 2006	15	8
March, 2006	16	8
April, 2006	6	3
May, 2006	15	2
June, 2006	24	9
July, 2006	14	7

Beaver Lake presented patterns similar to those described by James et al. (1987) in DeGray Lake, AR and Matthews et al. (2001) in Lake Whatcom. However, some variations might be attributed to different climate conditions and watershed characteristics.

In the same way as nutrient concentrations presented longitudinal patterns, Chl-*a* concentrations showed longitudinal patterns during stratified and non-stratified periods (Table 3.13). The statistical analysis showed that this difference in concentrations were significant within sites providing a $P < 0.0001$ for both cases. Higher Chl-*a* concentrations were measured in the sampling sited in the riverine zone (Figure 3.13 (A) and (B)).

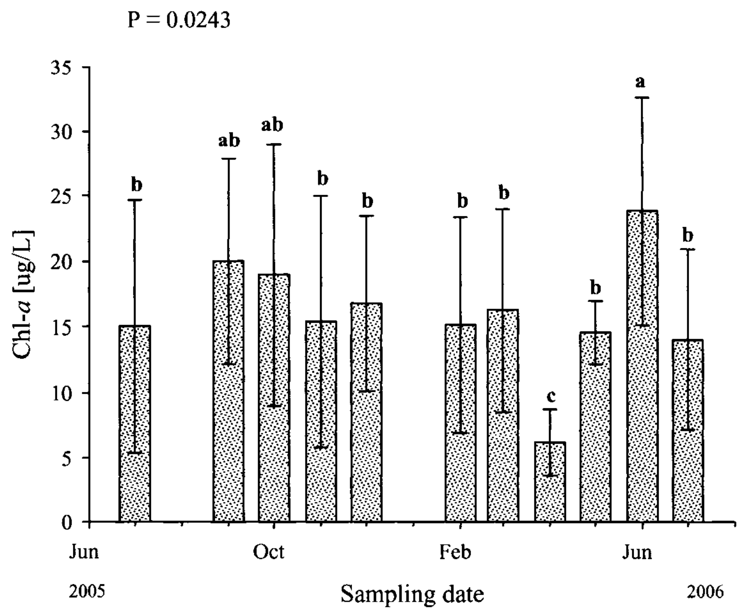


Figure 3.12. Chlorophyll *a* (Chl-*a*) mean concentrations (plus or minun standard deviation) per sampling date at the photic zone (1-3 m) in Beaver Lake. Significant differences denoted by letters (a, b, c) at $\alpha = 0.10$.

Table 3.13. Chlorophyll *a* (Chl-*a*) mean concentrations and standard deviation in the photic zone per sampling station for Beaver Lake during the stratified and non-stratified periods.

		Stratified Period		Non-stratified Period	
Sampling site	**Distance [km]**	**Mean Chl-*a* [ug/L]**	**Standard Deviation Chl-a [ug/L]**	**Mean Chl-*a* [ug/L]**	**Standard Deviation Chl-a [ug/L]**
BLL	22	8.27	4.16	7.96	1.49
HCK	30	11.93	5.23	11.32	2.15
WTR	37	12.63	5.74	12.90	4.65
FCK	45	18.43	8.41	23.51	4.27
BSP	50	16.29	6.88	26.03	11.42
WEC	40	18.79	7.22	16.81	8.20

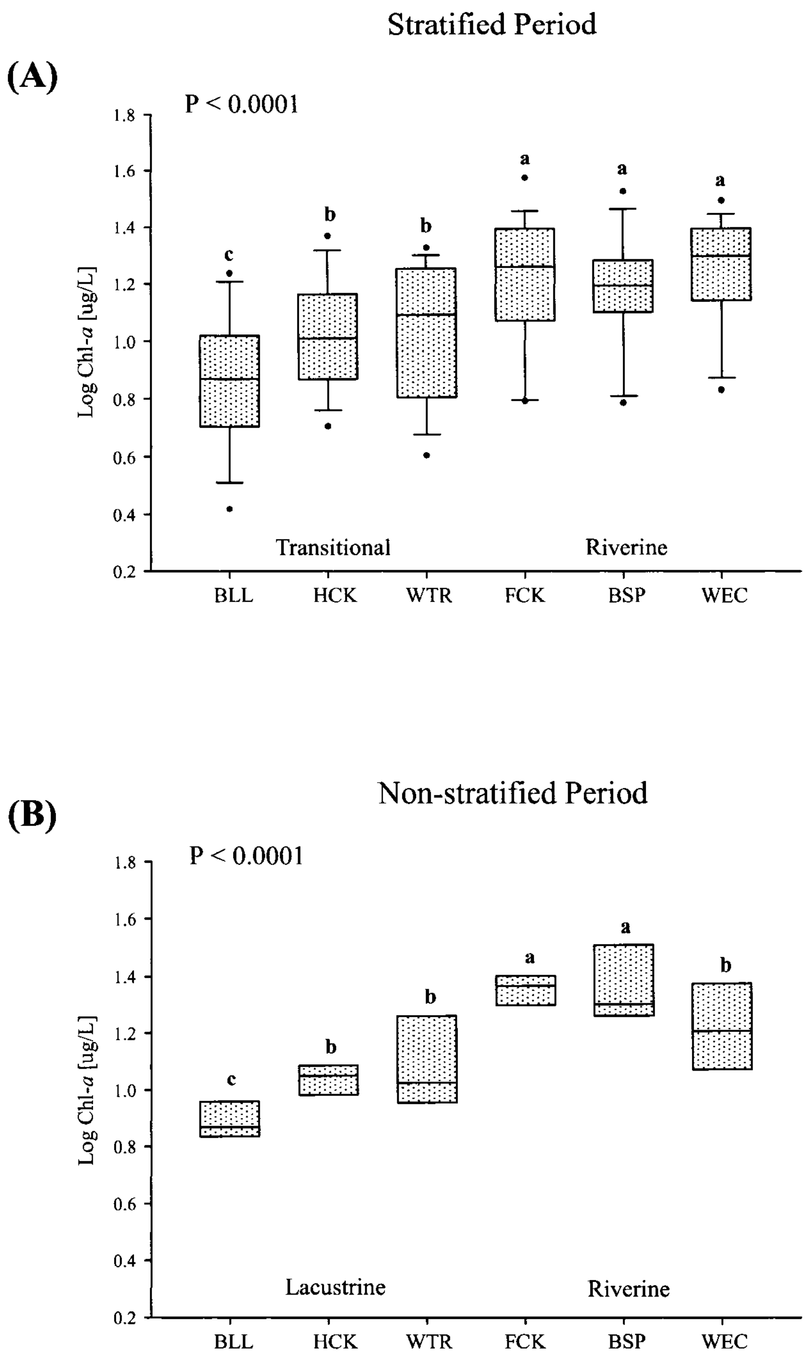


Figure 3.13. Box Plots of Chlorophyll *a* (Chl-*a*) concentrations at the photic zone (1 - 3 m) by sampling site in Beaver Lake during the stratified (A) and non stratified (B) periods. P-value calculated using ANOVA analysis with $\alpha = 0.10$ for Log_{10} transformed data.

The formation of longitudinal patterns is related to the development of trophic conditions in a reservoir. For Beaver Lake TSI values were calculated based on SD, TP, and Chl-*a*. The TSI values were analyzed based on ranges established by Carlson and presented in Table 2.3 and Figure 2.4. Table 3.14 shows a summary of Beaver Lake parameters used to determine its trophic conditions. No significant difference was found when comparing TSI values by site. However, some significant differences were found when comparing the three TSI values calculated for each sampling station.

Table 3.14. Chlorophyll *a* (Chl-*a*) and Total Phosphorus (TP) annual average (from July 2005 to July 2006) concentrations from the photic zone (1-3 m), SD transparency, and TSI values calculated by sampling stations used to determine the trophic conditions of Beaver Lake.

Samling Station	Zone	Chl-a [ug/L]	TP [ug/L]	SD [m]	TSI (Chl)	TSI (TP)	TSI (SD)
BLL	Transitional	7.8	40.9	1.8	50.0[b]	57.3[a]	52.8[b]
HCK	Transitional	11.8	44.4	1.2	54.2[b]	58.3[a]	57.8[ab]
WTR	Transitional	12.7	47.2	1.2	54.6[b]	59.4[a]	58.0[ab]
FCK	Riverine	20.6	62.7	0.8	59.4[a]	60.6[a]	63.2[a]
BSP	Riverine	20.6	53.8	0.7	59.7[b]	60.3[b]	66.5[a]
WEC	Riverine	19.0	46.2	0.9	58.8[a]	58.8[a]	61.7[a]
WEI	Riverine	19.6	64.0	0.6	57.7[b]	60.4[b]	69.5[a]
Overall	Transitional	10.8	44.2	1.4	53.0	58.3	56.2
Overall	Riverine	19.9	56.7	0.8	58.9	60.0	65.2

Letters represent significant differences between TSI values calculated per sampling site with $\alpha = 0.1$

The TSI values calculated based on Carlson's equations presented some differences in the prediction of trophic conditions, especially in the riverine zone. Basically, TSI(TP) and TSI(Chl) showed that the transitional and riverine zone present eutrophic conditions. However, TSI(SD) values showed that the riverine zone presented more hypereutrophic conditions. This difference can be attributed to the influence of suspended sediments (non-algal particles) that affect the transparency. In addition, according to the literature, TSI values for any of the three parameters used should be close to each other and predict the same conditions. However, the TSI values calculated for Beaver Lake showed that the TSI values calculated for TP are higher than the ones calculated for Chl-*a*. This difference can be attributed to the fact that TSI values were based on the assumption that reservoirs are P limited. Because TSI(Chl) is the only one that represents the actual biomass, Carlson recommends using this value to estimate lake trophic condition (Carlson, 1977; Matthews et al., 2001). Finally, it must be considered that unusual dry conditions were produced in Beaver Lake during the sampling period (July 2005 to July 2006) and the reservoir presented low water levels.

Comparing the results from Table 3.14 to the ranges defined by Carlson, we can conclude that Beaver Lake is eutrophic in the transitional zone. Even though, the riverine zone presents some eutrophic characteristics some of the values show a shift to hypereutrophic conditions. The headwaters of a reservoir are directly affected by the inflow nutrient and sediment loadings. For this reason, the nutrient concentration in sediments is higher here than in sites closer to the dam

(James et al., 1987). Therefore, riverine zones (headwaters) can present favorable conditions for primary productivity when light penetration is not limited. Towards the dam, even though light penetration increases, nutrient concentrations decrease and so as primary productivity (James et al., 1987).

Haggard et al. (1999) reported that the riverine zone presented eutrophic conditions in 1993-1995. This current study shows the same characteristics but this may have been increased in the last 10 years since some of the TSI values and concentrations of TP were in the high end of the eutrophic range.

Comparison of Beaver Lake water quality data

The annual mean for Beaver Lake found in the present study is about double of the greatest annual average reported by Haggard et al. (1999). The reservoir have increased in Chl-*a* concentrations about 10 years later (Table 3.15). Total Phosphorus concentrations are also greater in the current study with an annual concentration of 51-ug/L. Even though, TP and Chl-*a* concentrations have increased in the last 10 years, TN concentrations were three fold less than the concentrations reported by Haggard et al. (1999). The main reason for this difference in concentrations may be attributed to differences in climate conditions (particularly precipitation), inflow characteristics, and reservoir water levels. In addition, the water quality data presented by Haggard et al.

(1999) includes the lacustrine zone and the current study does not include this zone.

Haggard et al. (1999) reported that during one year the reservoir experienced atypical high water levels with high discharge episodic events. These episodic events increased the amount of total suspended solids decreasing light penetrations and algal production. On the other hand, during the current evaluation the reservoir presented an extremely dry season with very low water levels. The average for a normal total annual precipitation for Beaver Watershed is approximately 1,200mm/yr (based on 10 yr average) and the total annual precipitation for 2005-2006 falls below this average with will give us enough evidence to assure that this period was a dry. In general, it can imply that water quality in the last 10 years has been affected by climatic conditions, especially precipitation (Table 3.5).

Galloway and Green (2006) in their study in Beaver Lake during 2001 to 2003 showed that nitrogen concentrations varied seasonally, longitudinally, and vertically. The site located in War Eagle Creek presented the greater concentrations in TN, attributing this to agricultural land use. In general, phosphorus concentrations were low with some concentrations below method detection limit. During the stratified period, phosphorus concentrations were higher in the hypolimnion in the riverine and transitional zones.

Table 3.15. Comparison of annual mean Chlorophyll-*a* (Chl-*a*) concentrations for the photic zone in the upper reaches of Beaver Lake.

	Chl-a [ug/L]		TP [ug/L]		TN [mg/L]		Total Annual *
	Mean	Median	Mean	Median	Mean	Median	Precipitation [mm]
Meyer (1974) [a]	4.98	3.36					
Haggard et al. (1999)[b]	6.99	5.11	34	28	1.29	0.93	1098
Haggard et al. (1999)[c]	3.29	1.73	42	30	1.21	0.91	1370
Current (2005-2006)	16.02	18.99	51	47	0.41	0.41	769

[a] Values from 1974 data of Meyer RL.
[b] Values from August 1993 - July 1994 data of Haggard et al. (1999)
[c] Values from August 1994 - July 1995 Haggard et al. (1999)
* NCDC - Eureka Springs 3 WNW

Nutrient Limitation Bioassay

Nutrient enrichment experiments resulted in the addition of nutrients in the water and the measure of response as an increment in biomass, productivity, or changes in cellular characteristics (Gerhart and Likens, 1975). The general purpose of these experiments is to predict the response of algae to nutrient inputs in aquatic ecosystems. In this specific case, the nutrient enrichment experiment was developed during the months of September and late November 2005 and January and late May 2006, in Beaver Lake, AR (Table 3.16). The purpose was to determine the response of algae to the addition of nutrients at different seasons.

July 2005 bioassay

The initial Chl-*a* concentration at HCK during September 2005 near the surface (approximately 0.5m) was 2.2ug/L. In this bioassay, primary productivity was stimulated by the presence of both N and P. If we compared the four main treatments: Control (no nutrients added), NA:PH (only P added), NH:PA (only N added), and NH:PH (N and P added) we can see that the response when N is added is greater than when only P is added. Even though, the greatest response is when both nutrients are added (Figure 3.14a). From this experiment, it can be concluded that during this period the reservoir presents N limitation but the addition of N and P showed the greatest response in Chl-*a* concentrations. Similar conditions were observed for different authors such as Havens and East (1997), Randolph and Whilm (1984), and Piehler et al. (2003) were the reservoirs presented N limitation.

September 2005 bioassay

The initial Chl-*a* concentration was 7.9ug/L. Great algal response was observed when N and P were added at intermediate concentrations and not at the high concentration range. When comparing the control and the treatments with the additions of either N or P, no significant differences in the algae response were found. In conclusion, the addition of just one the nutrients (N or P) does result in a significant increase in algal production; however, when both nutrients are added a significant response is observed (Figure 3.14b). This response indicates co-limitation (N and P) of algal growth. Responses similar to the one observed in this nutrient enrichment experiment were found in other studies like Gerhart and Likens (1975).

November 2005 bioassay

The initial Chl-*a* concentration at the surface was greater than in September, with a concentration of 11.2ug/L. The higher responses were observed during treatments where N was added at low concentration and P varied from low to high concentrations (look the four by four matrix for nutrient concentration combinations). By looking at the four treatments (Control, N, P and NP) it can observed that the growth of algae in this period is stimulated by the treatments were P and NP are added. The difference in Chl-*a* concentrations for both treatments (P and NP) was not significant (Table 3.14c). During this period, we can conclude that the reservoir presented a P limitation. Piehler et al. (2003) in his experiment for the fall season reported the same co-limitation (N-P).

January 2006 bioassay

The initial Chl-*a* concentration at the surface was 14.2ug/L, the greatest observed. In this treatment greater Chl-*a* concentrations were observed when high concentrations of P were added. The same situation can be observed if we compared the four treatments. When comparing the Control (no nutrients added) and the treatment with just N addition we can see that no significant difference in mean concentrations was found (Figure 3.14d). In conclusion during the winter time Beaver Lake was classified as P-limited. Piehler et al. (2003) for the winter season reported a N limitation.

May 2006 bioassay

In the month of May the Chl-*a* concentration measured at the surface during the initial day was 11.8ug/L. The treatments in this period showed algae responses basically in all treatments except the ones with no addition of phosphorus. This behavior can be observed one more time comparing the control, N, P and N-P treatments. The greater response was when N and P were added in high concentrations. Also, a response significantly different from the control occurred with the addition of P. In conclusion, during this experiment Beaver Lake presented a P limitation with a tendency to co-limitation (NP) (Figure 3.14e). Gloss et al. (1980) in his study of Lake Powell in Utah reported similar conditions. The P limitations in the epilimnion for the growth of algae may be attributed a lack of regeneration of this nutrient.

Table 3.16. Least Significant Difference (LSD) test comparison of Chl-*a* concentrations for Control, Phosphorus (NA:PH), Nitrogen (NH:PA), and Nitrogen plus Phosphorus (NH:PH) treatments used in the nutrient enrichment bioassay for different seasons in Beaver Lake, AR 2005-2006.

Period	**Treatment**	**Number of Replicates**	**Mean Chl-a [ug/L]**	**Standard Deviation [ug/L]**	**LSD level (a = 0.10)**
July, 2005	C	4	2.99	0.67	C
	NA:PH	4	3.09	0.23	C
	NH:PA	4	13.41	2.60	B
	NH:PH	4	48.11	5.36	A
September, 2005	C	5	4.19	0.74	C
	NA:PH	5	8.39	0.97	B
	NH:PA	5	8.59	1.39	B
	NH:PH	5	30.45	12.57	A
November, 2005	C	5	21.01	2.86	B
	NA:PH	5	40.96	2.99	A
	NH:PA	5	22.82	1.29	B
	NH:PH	5	28.66	2.43	A
January, 2006	C	5	17.04	1.39	B
	NA:PH	5	35.99	2.45	A
	NH:PA	5	16.81	0.79	B
	NH:PH	5	32.07	1.26	A
May, 2006	C	5	10.84	2.25	C
	NA:PH	5	45.39	10.30	B
	NH:PA	5	12.32	2.55	C
	NH:PH	5	69.35	7.31	A

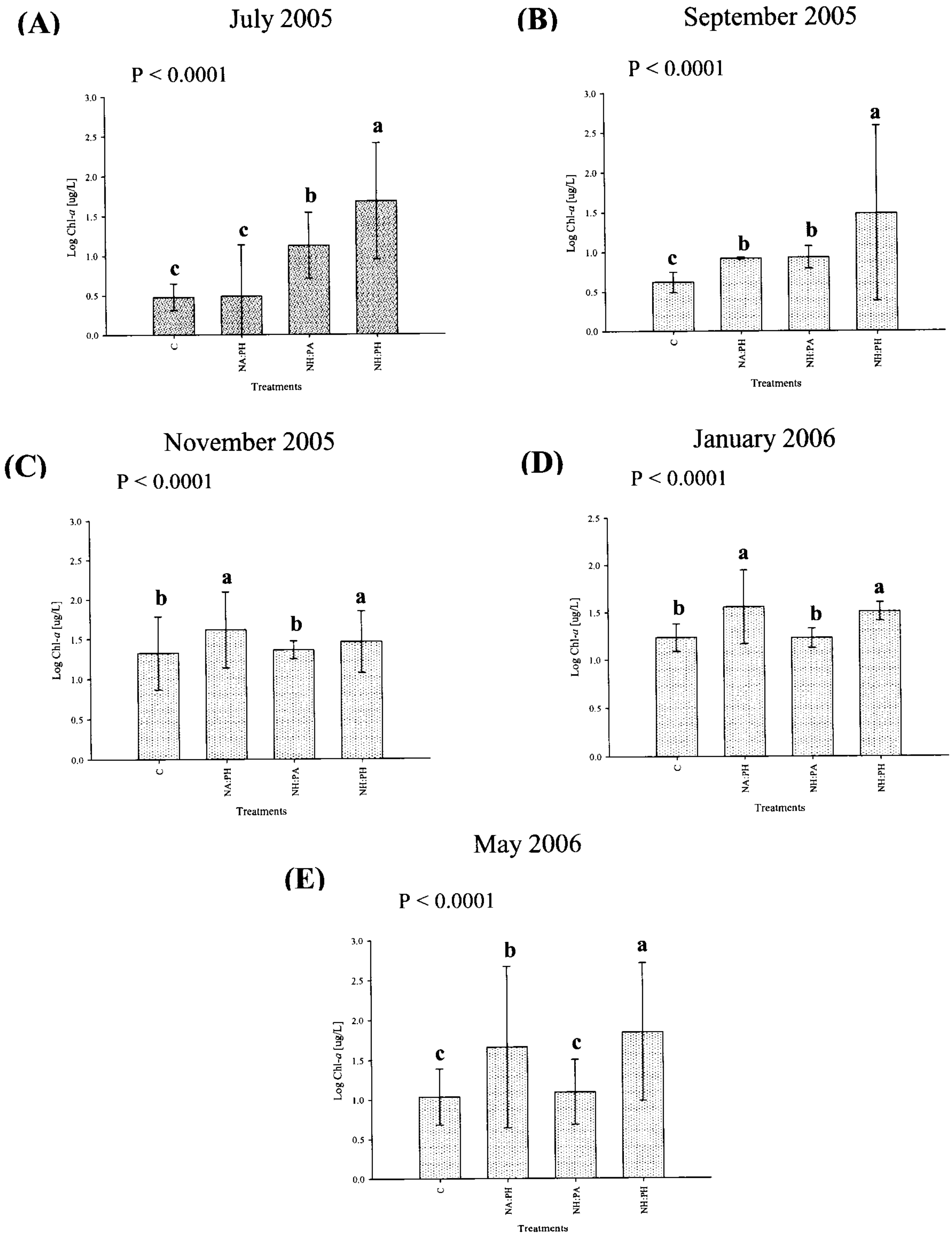


Figure 3.14. Comparison of algae responses (means plus or minus standard deviation) to nutrient enrichment treatments: C (no nutrient addition), NA:PH (P added), NH:PA (N added) and NH:PH (NP added) in Beaver Lake for all seasons. Letters represent the significant difference determined using $\alpha = 0.10$ for Log_{10} transformed data: (A) July 2005, (B) September 2005, (C) November 2005, (D) January 2006, and (E) May 2006.

In conclusion, the results showed that Beaver Lake was N-limited during July 2005, then NP co-limited in September 2005, and finally P-limited during November 2005, January and May 2006. Water quality results for Beaver Lake showed that July and September 2005 presented the least NO_3-N concentrations (below detection limit 5ug/L), respectively) and SRP concentrations were greater in July 2005 (12 ug/L) and SRP concentrations were depleted by September 2005 (6 ug/L) which may the reason of why the reservoir shifts from N limitation to a co-limitation. During the periods of November 2005, January, and May 2006 when the reservoir showed P-limitation NO_3-N concentrations increased (14, 65, and 85 ug/L) and P concentrations were reduced (5, 6, and 7 ug/L). Since water quality samples were not collected for the month of January 2006, NO_3-N and SRP concentrations were estimated as an average of the values measured during December 2005 and February 2006. Finally, Beaver Lake nutrient limitation conditions can shift according to the availability of nutrients in the water column.

CONCLUSIONS

The goal of this project was to evaluate changes in trophic status in the upper reaches of Beaver Lake. The objectives were to: 1) determine the trophic conditions in the headwaters of Beaver Lake in terms of chlorophyll *a* (Chl-*a*) and nutrient concentrations, 2) evaluate the variability in nutrient (particularly P) concentrations with depth, 3) determine the relationship between Chl-*a* and nutrient concentrations, 4) evaluate and compare the current conditions versus conditions found in previous studies conducted at Beaver Lake, and 5) determine the limiting factor of algal (phytoplankton) growth at Beaver Lake. These objectives were met by testing the following hypothesis:

H_{o1}: A marked longitudinal gradient does not exist in trophic conditions at Beaver Lake. Beaver Lake during 2005-2006 presented a longitudinal gradient were TN, SRP, and Chl-*a* concentrations presented higher concentrations in the riverine zone during the non-stratified season. In addition, longitudinal gradients were observed during the stratified season for Chl-*a* concentrations. Therefore, we could reject the null hypothesis and accept the null hypothesis were longitudinal gradients are found in Beaver Lake, AR.

H_{o2}: Nutrient (particularly P) concentrations do not vary with depth in the transitional zone of Beaver Lake. Phosphorus forms also presented a vertical gradient were higher concentrations were observed in the hypolimnion during the stratified period with significant statistical differences ($P<0.1$).

H_{o3}: Algal (phytoplankton) growth measured via Chl-*a* concentrations is not correlated to the limiting nutrient. Algal (phytoplankton) growth presented a positive and significant Log-linear relationship with TP and TN ($P=0.02$).

H_{o4}: Algal (phytoplankton) growth is not limited by nutrients. Bioassay experiments confirm that algal production is stimulated by the addition of both nutrients, N and P. Therefore during the study of Beaver Lake 2005-2006, the reservoir presented a nutrient limitation shift from N limitation to co-limitation (N-P) and P limitation. This shift may be related to whether or not one of these nutrients is available for algal uptake.

The comparison between historic and current water quality data from Beaver Lake, showed that Chl-*a* and TP concentrations have increased and TN concentrations have decreased. The high concentrations of Chl-*a* and TP during the current study may be attributed to dry conditions during 2005-2006 and very low water levels, creating optimal conditions for algae blooms and P release from sediment. Seasonal, vertical and longitudinal patterns analyzed in previous studies (Haggard et al., 1999; Galloway and Green, 2006) showed similar tendencies during 2005-2006.

CHAPTER 4

CONCLUSIONS AND RECOMMENDATIONS

CONCLUSIONS

The goal of the present study was to evaluate changes in trophic status in the upper reaches of Beaver Lake. The goal was accomplished by testing the objectives proposed in Chapter 1.

Objective 1:

Determine the trophic conditions in the headwaters of Beaver Lake in terms of chlorophyll-*a* (Chl-*a*) and nutrient concentrations.

Longitudinal patterns are related to trophic conditions. In this case the null hypothesis to be tested was, H_o: A marked longitudinal gradient does not exist in trophic conditions at Beaver Lake. The results showed that TN and SRP presented longitudinal gradients with higher concentrations in the riverine zone. In addition, Chl-*a* showed the highest concentrations in the riverine zone. The statistical test showed that the longitudinal difference in concentration of nutrients and Chl-*a* are significant with p-values less than 0.1. Therefore, the null hypothesis can be rejected.

In general, Beaver Lake during the study (2005-2006) presented eutrophic conditions in the riverine and transitional zone. However, the riverine zone presented a potential shift to hypereutrophic conditions due to high concentrations of TP. This can be attributed to the presence high concentrations of total suspended solids. Also, this potential shift to hypereutrophic conditions may be attributed to dry conditions during the sampling period (2005-2006).

Objective 2:

Evaluate the variability in nutrient (particularly P) concentrations with depth in the headwaters of Beaver Lake.

The null hypothesis to be tested to accomplish this objective was: H_o: Nutrient (particularly P) concentrations do not vary with depth in the transitional zone of Beaver Lake. The data show that SRP and TP concentrations are higher in the hypolimnion of the transitional zone. The statistical test showed that this difference is significant ($P<0.1$). Therefore, the null hypothesis is rejected, and the alternative hypothesis that P vary with depth in the transitional zone in Beaver Lake is accepted.

Objective 3:

Determine the relationship between Chl-*a* and nutrient concentrations.

This objective was tested with the following null hypothesis: H_o: algal (phytoplankton) growth measured via Chl-*a* concentrations is not correlated to the limiting nutrient. For the period 2005-2006 the data showed that Chl-*a* has a positive and significant Log-linear regression ($P<0.1$) with TP, TN, and the combination of both. Therefore, changes in concentrations of these nutrients will produce changes in concentrations of Chl-*a*. Total P explains approximately 71% of the total variation of Chl-*a* about the mean. Total Nitrogen explains 69% of the variance of Chl-*a* concentrations. Therefore, the null hypothesis was rejected with a $P < 0.1$.

Objective 4:

Evaluate and compare the current conditions versus conditions found in previous studies conducted at Beaver Lake.

The comparison between the present study and Haggard et al. (1999) showed that Chl-*a* and TP concentrations are higher in the current conditions. However, TN concentrations 10 years ago had about 3 times the concentration present in current conditions. This may be attributed at the very dry season during 2005-2006 and the scattered precipitation events which may transport N by runoff from agricultural lands in the watershed. In addition, this may be attributed to the reduction in N concentration from the wastewater treatment plant effluent.

Objective 5:

Determine the limiting factor of algal (phytoplankton) growth at Beaver Lake.

This objective was met by testing the following null hypothesis. H_0: algal (phytoplankton) growth is not limited by nutrients. The nutrient enrichment experiments developed in Beaver Lake during 2005-2006 showed that nutrient limitations for algae growth shift from N, P, and to co-limitation N-P. This may be related to the presence and availability of N and/or P during a certain period of time. Therefore, enough evidence was found to reject the null hypothesis and accept the alternative hypothesis that primary productivity depends upon nutrient availability.

RECOMMENDATIONS

The present study showed that nutrient concentrations vary with longitude and depth. Also, nutrient variation was related to Chl-*a* concentrations and their availability will limit primary productivity. For future research studies it may be useful to consider the following recommendations.

- Analyze and evaluate land use/cover and their nutrient contribution to main tributaries to Beaver Lake. Compare this information with water quality data on Beaver Lake, AR during the same period.
- Determine the importance of internal loading (especially for P) in algal blooms.
- Evaluate light and turbidity effects on primary productivity, simultaneously with physical and chemical variables.
- Evaluate the influence of carbon (C) to primary production.
- Determine how the algae species will vary depending on the availability of nutrients and optimal physical conditions.

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