USE OF WATER QUALITY MEASUREMENTS TO DETECT POTENTIAL SEPTIC SYSTEM INPUT AT GRAND LAKE, OKLAHOMA

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USE OF WATER QUALITY MEASUREMENTS TO DETECT POTENTIAL

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CHAPTER I

Prologue

INTRODUCTION

The Pensacola Hydroelectric Project impounds the Neosho and Spring Rivers in northeastern Oklahoma to form Grand Lake O' the Cherokees (Grand Lake). This lake covers 18,818 hectares, holds $2.06X10^9$ M^3 of water, and is the third largest reservoir in Oklahoma with 2092 km of shoreline (OSE 2004). Lake depth averages 11m with a greatest depth of approximately 50m. Pensacola Dam, constructed by Grand River Dam Authority (GRDA) in 1940. The lake spans a total of four Oklahoma counties including Ottawa, Mayes, Delaware, and Craig, while the entire drainage area of the lake covers over 2.5 million hectares and extends across state borders into Arkansas, Kansas, and Missouri (OSU and OWRB 1995). The fact that this watershed spans across state borders results in several different government jurisdictions and United States Environmental Protection Agency (USEPA) regions coordinating issues related to environmental impacts or restoration.

Grand Lake is an extremely valuable natural resource and has fostered considerable economic growth in northeastern Oklahoma. This area is one of the most popular and fastest growing retirement locations in the United States (OSE 2004). During Memorial Day, 4^{th} of July, and Labor Day, the Grand Lake community grows to the third largest in Oklahoma followed only by Oklahoma City and Tulsa (Alberty 2005). While communities surrounding the lake have enjoyed both physical and economic growth due to the popularity of the lake, this popularity has led to extensive development of property close to the shoreline over the entire lake. Grand Lake is one of the few in Oklahoma that allow homes on the waterfront and where nearshore development is considered a factor in the lakes current state of cultural eutrophication (OSE 2004).

JUSTIFICATION

While the recreational opportunities associated with the creation of Pensacola Dam are valuable assets to communities in and around the Grand River Basin, many human activities may jeopardize ecosystem integrity and function. Detrimental activities in the Grand Lake watershed include nutrient inputs from confined animal feeding operations, golf course fertilization, wastewater treatment facilities, and septic systems (OSE 2004). According to GRDA, the increasing popularity of the lake has caused an increase in residential and commercial development with the majority utilizing onsite wastewater treatment systems i.e. septic systems. This influx of human activities has likely accelerated the delivery of pollutants capable of impacting local water quality and may be contributing to eutrophication. This is particularly important because Grand Lake not only supplies raw water to local residents and marinas, but it is also the primary drinking water supply for many local municipalities. Thus, malfunctioning, inadequate, or poorly sited septic systems are of particular concern to rural residents and local communities. Effluent from malfunctioning or improperly installed septic systems has often been identified as a major source of pollution and can pose serious threats to water quality in some reservoirs (Lipp et al. 2001, USEPA 2002). Fecal bacteria, biochemical oxygen demand (BOD), and nutrients (mainly N and P), originating from inadequate septic systems, can act together to contaminate surface and groundwater sources.

Our primary objective is to address lakeside development. The construction of dwellings along the shoreline occurs primarily in rural areas where access to municipal sewage treatment plants is unfeasible. Septic systems have been utilized to treat household wastewater and are often installed at the minimum distance from the shore (5m) permitted by law at the time of installation (Chen 1988). Each dwelling is usually accompanied by a septic system; however, in grouped housing such as trailer parks and condominiums, each septic tank may provide wastewater treatment for more than one dwelling. The Oklahoma Department of Environmental Quality (OKDEQ) has documented instances where developers on Grand Lake have overburdened individual septic systems with multiple dwellings, producing wastewater in excess of the systems ability to provide adequate treatment. This heavy utilization of septic systems in shoreline development could contribute to the degradation of water quality.

PROJECT OBJECTIVES

Our objective is to characterize local housing development and identify areas where septic systems are a possible source of surface water contamination and monitor water quality at these sites to see if any indicators of septic input can be detected. Our primary objectives include:

1. Review county and state tax record and plat maps associated with residential subdivisions and/or commercial development located on or near the shoreline of Grand Lake to determine if site-specific characteristics influence water quality.

2. Classify subdivisions located on or near the lakeshore into the following classes: 1) no septic present (control) and 2) various age classes at ten year intervals from <10 yrs->40 yrs to determine the relative influence of aging septic systems on water quality.
3. Evaluate water chemistry at selected sites to determine presence and concentration of wastewater indicators.
4. Utilize infrared technology to identify and locate discharge effluent originating from failing septic systems or pipes that discharge directly into the lake with a helicopter-mounted FLIR camera.

LITERATURE REVIEW

Septic System Overview

The large majority of homes in the United States are served by public sewer systems (U.S. Census Bureau 1990). However, onsite wastewater treatment systems serve roughly 25 percent of all homes in this country (U.S. Census Bureau 1990). Despite a large number of homes serviced by onsite wastewater treatment systems, research in this area is somewhat lacking with most work focusing on larger municipal wastewater treatment facilities. Hain and O'Brian (1979) list several reasons for this: 1) Municipal sewage systems are high-flow point sources, which must meet rigorous federal standards, but are not monitored by the federal government, 2) These facilities are also designed for ease of effluent observation and sampling unlike onsite wastewater treatment systems, 3) While a lone onsite wastewater treatment system represents a point source of pollution, clusters of tanks in a particular area represent a non-point source making source tracking more difficult, 4) Many individual systems lie under landscaping which is relatively expensive to restore after septic system sampling, with homeowner objections to that investigation of their septic systems will result in additional expense on their part by suggesting expensive repairs and maintenance. There is a need to remedy the lack of research in this area given the frequent use of septic systems in rural areas.

Although there are a number of different types of alternative wastewater treatment systems (i.e. aerobic with land application, evapotranspiration/absorption, and lagoon), only those utilizing septic tanks will be considered in this study.

Figure 1.1 illustrates the most widely used septic system, referred to as a conventional system, consisting of two parts: a septic tank to collect and hold wastewater and a soil absorption system utilizing long perforated pipes (lateral lines) to evenly distribute wastewater into the soil. Soil functions as a biological, physical, and chemical treatment medium for wastewater, as well as a porous medium to disperse the wastewater in the receiving environment as it percolates to the groundwater (USEPA 2002). In the state of Oklahoma, a septic tank used in an individual sewage disposal system for a residential unit with four or less bedrooms must have a liquid capacity of at least 3785 L with an additional 946 L for every additional bedroom >four (ODEQ 2004). These regulations also state that absorption fields must be installed > five feet from the septic tank and the trenches containing lateral lines must lie between 46-76cm deep with at least 5cm of absorption media (i.e. rock, gravel, or tire chips) above and below the lateral lines. Many different configurations of conventional septic systems exist; however, all utilize the same basic components and rely on the hydrologic properties of soil for wastewater treatment. The limitations of this design are directly related to the inherent variability and heterogeneity of soil and soil biogeochemical processes (Beal et al. 2005). According to the same study, appropriate design, construction, and maintenance, based on prior knowledge of the site and soil conditions, are crucial for the sustainable and successful operation of these systems. State codes assume the relationship among soil characteristics, size of dwelling, and size of the installed onsite system will determine how well the system will function (Hudson 1986). It has been indicated that these assumptions may be inappropriate due to the fact that a set of codes could be adequate for a three-bedroom house in sandy soil and at the same time be inadequate for a five-bedroom house in tighter soil in the same jurisdiction (Hudson 1986). The state of Oklahoma only requires that a septic system absorption field installation site meet standards based on percolation test results. This test determines how well a particular soil type passes water through it. OAC Title 252 (Individual and Small Public On-site Sewage Disposal Systems) establishes a procedure that requires digging approximately three 10-31cm diameter holes 46-76cm deep in site to be tested. A presoak period follows requiring the holes to be filled with water to a depth of 31cm and maintained for four hours prior to the test. Upon completion of the presoak, water level is adjusted to 25cm and the drop in water level in sixty minutes or the time it takes to drop 10cm is measured. The site is deemed suitable if the percolation rate is less than or equal to 24 min per centimeter (sixty minutes per inch). Anything greater than 24 minutes per centimeter or if groundwater is encountered while digging would cause the site to be classified as unsuitable.

Flow path of Household Wastewater

Household wastewater enters the septic tank where solids settle out and undergo anaerobic digestion resulting in sludge of lesser volume remaining in the bottom of the septic tank. This sludge must be pumped out periodically. While the solid portion of wastewater settles, the floatable portion of wastewater, consisting mainly of grease and oil (scum), floats to the surface above the inlet/outlet fittings (Figure 1.2). As wastewater enters the septic tank, the partially treated liquid portion is pushed out of the discharge opening of the sanitary tee, which has an inlet port that extends down into the relatively clean septage layer between the scum and sludge layers, and passes into the perforated lines to be distributed into the drainfield. The most widely used absorption field design uses lines that are placed in trenches of sand or gravel to partially filter the wastewater before it passes into the soil layer below. Onsite wastewater treatment systems have a range of cost saving benefits for different types of communities and conditions instead of utilizing an expensive wastewater treatment facility and associated collection and distribution piping. A USEPA report to a Congressional House Appropriations Committee (1997) states that onsite wastewater treatment systems benefits include: 1) more cost effective for low density communities rather than more expensive large facilities, 2) can be used over a wide range of site conditions, 3) are also suitable for ecologically sensitive areas where advanced nutrient removal and disinfection is necessary, 4) when properly installed, operated, and maintained, they can recharge local aquifers and provide water reuse opportunities close to points of wastewater generation (USEPA 1997). These benefits only apply when the right system is installed under the right circumstances. According to the Oklahoma Department of Environmental Quality (ODEQ) and a review of septic system permits filed in Delaware, Craig, and Mayes counties, OK, indicates that approximately 98% of septic systems in the Grand Lake area are of the conventional design described earlier. This includes systems installed near the water line. One major attribute contributing to this fact is that a conventional system usually costs less than five thousand dollars to install and has relatively low maintenance requirements (Swann 2001). This makes the conventional system more attractive to homeowners than an alternative system. A review of county soil survey maps indicated that the poorest site conditions for a conventional system occur at or near most of the waterfront on Grand Lake (USDA 1973, USDA 1972, and USDA 1970). This suggests the need for alternative septic system designs, such as systems that distribute septic system effluent above ground through a spray apparatus. These systems spray septic system effluent over vegetation. The vegetation removes nutrients and the aerobic environment inside the system tank itself fosters a favorable environment for organisms that remove organic constituents (Roth 2005).

Fate and Transport of Septic System Effluent

Septic system effluent passes through two zones after leaving the system drainfield. The first is an aerated/unsaturated zone (vadose zone), while the second is a saturated zone also known as the water table. Figure 1.3 illustrates routes of transport of effluent leaving the drain field. Upon reaching the saturated zone, septic system effluent moves along with groundwater, thus, taking on the same properties as groundwater and affected by the same factors that affects groundwater. It is possible that effluent never reaches groundwater. In unsuitable soils, an impermeable confining layer or extremely porous layer can lay between the absorption field and groundwater. In cases such as this, untreated or partially treated effluent reaches the confining layer that impedes movement causing surface ponding or rapidly travels along the layer receiving inadequate treatment before reaching lakes or streams (Brown 1992).

The ultimate goal of any onsite wastewater treatment system is to lower contaminant levels in the effluent before it reaches groundwater. The movement of septic system plumes is heavily dependent on soil type, soil layering, underlying geology, topography (slope), and rainfall (USEPA 2002). The proximity of a septic system drainfield to surface water used as a drinking/recreation source increases the risk that the associated wastewater plume will have some impact on human health or water quality. Unsuitable soil types, listed in Septic System Siting section, require a larger separation distance. Oklahoma state law requires a minimum of 46cm separation distance between the absorption trench and water saturated soil (ODEQ 2004). Domestic sand-point wells located within 31m of a septic system and less than 14m deep in a shallow aquifer are most vulnerable to septic waste contamination (Verstraeten et al. 2004). A suite of factors affect the fate and transport of septic system effluent and associated contaminants. The factors that will be considered in this project will be addressed and described in more detail below.

Septic System Density

A figure of 15 septic systems per square kilometer has been quoted as the density at or above which catchment scale impacts from septic effluent are likely to be observed (Whitehead et al. 2001). Borchardt et al. (2003) found that the risk of viral diarrhea in central Wisconsin children increased by 8 percent for every additional septic tank/259ha, while the risk of developing bacterial diarrhea increased by 22 percent for each additional septic tank/ 16ha. A study by Lipp et al. (2001) found that risk of wastewater contamination increased in areas of high on-site sewage disposal system densities. This study goes further to state that pollution from these systems undergoes subsurface transport to surface waters. Given the high system densities, based on the Whitehead et al. (2001) study, this subsurface transport of septic system effluent may be occurring in areas around Grand Lake. Other studies have indicated septic system density is a factor in elevated concentrations of various wastewater constituents. A study by Brendle (2004) found significant differences in concentrations of nitrate, chloride, and boron in samples taken from groundwater in areas of different septic system densities. Results indicated a mean nitrate concentration 75 percent higher for samples from high-density areas than those of low-density categories. Median chloride concentrations were 65 percent greater for high-density areas and median boron concentrations were 39 percent greater.

Grand Lake development occurred without regard to the placement of septic systems (OWRB 1995). Furthermore, 8,093 homes have been built within 150m of the lake perimeter at flood pool elevation and 1,273 between 150m and 400m from this elevation by 1991 (OWRB 1995). Based on current GIS information and recent census data, there has been a 27.6% increase in population growth since 1991 which has resulted in an additional 2,185 homes located within 150m of the lake perimeter at flood pool elevation and 1,617 at the 150m to 400m range (OSE 2004). A large portion of homes around the lake are grouped into sub-divisions. The result is large numbers of dwellings in an area of only a few hectares. For example, The Coves at Bird Island is a residential development on the eastern shore of Duck Creek has 212 houses located within 121 hectares.

Septic System Age and Failure

Septic system age plays an important role in the effectiveness of wastewater treatment. Newer, more efficient systems may trap or inactivate chemical and biological contaminants more rapidly than older or malfunctioning systems (Hain and O'Brian 1979). Even a well maintained septic system will eventually reach the end of its useful service life. Hardware components can age and mechanically fail and absorption fields can lose the ability to treat wastewater nutrients over time. Beneficial waste treating bacteria (biomat) can become too thick to allow sufficient effluent flow (Lee et al. 2005). This biomat forms at the absorption field trenches and underlying soil interface and the growth rate is related to the amount of organic waste infiltrating below the trenches. In properly operating absorption fields, the bacterial cell growth rate and death rate is equal, but if wastewater is being added faster than the biomat can degrade it, the biomat thickens and impedes flow resulting in reduced system efficiency and even hydraulic failure (Lee et al. 2005). Septic system absorption fields are calculated to have an expected life span of 15 to 30 years (Evans et al. 1999). A Baffaut (2004) study on Shoal Creek, Missouri, correlated age to failure rate. This study estimated failure rates at 40% of systems >37 years, 20% failure of systems 22-36 years, and 5% failure of systems <22 years (Baffaut 2004). Average lifespan of most septic systems is 20 years, depending on soil and climate conditions (Center for Watershed Protection 1999). Age alone is not an adequate indicator of how long a septic system will provide effective treatment. Siting and maintenance have also been documented as important factors (Center for Watershed Protection 1999). The Center for Watershed Protection (CWP) (1999) For instance, the relative age of a system does not guarantee its proper function and an improperly installed system can fail within three to five years (CWP 1999). A new system installed under poor site conditions may never provide adequate treatment. Inadequate maintenance can also greatly reduce treatment efficiency of conventional septic systems.

As sludge builds up in the septic tank, the relatively clean septage layer between the sludge and scum layer becomes smaller resulting in some discharge of sludge to the absorption field. Sludge that is allowed into the absorption field can clog lateral lines and soil pores resulting in hydraulic failures. Hudson (1986) stated several direct causes for visible septic system surface failures include: hydraulic overload, lack of maintenance resulting in an overflow of solids into absorption field lines, and reduced soil permeability caused by smearing of soil by heavy equipment during construction. The same study lists several common reasons for system failure with respect to human activity and natural conditions, which are: improper design for site conditions, poor construction practices, and insufficient maintenance. Most literature concerning maintenance of onsite wastewater disposal systems recommends yearly inspection of septic systems to minimize costs and extend system service life (CWP 1999). CWP (1999) also reports that most septic system specialists recommend that systems be pumped out at least every 3 years depending on size of septic tank and number of people per household.

Septic System Siting

Proper siting of septic systems is arguably the most important aspect of proper onsite wastewater treatment. It is especially critical to closely evaluate sites located near surface water that is used for drinking and recreation. Site evaluations usually proceed in three phases: 1) a preliminary review of documented site information, 2) a ground-truthing activity to potential sites, and 3) a detailed evaluation of the most promising location for placement (USEPA 2002).

Ground-truthing many areas around Grand Lake has indicated that land surrounding the lake is parceled out in small lots <1ha with barely enough area to meet state septic system requirements (i.e. setback distances from water sources). The ODEQ requires septic tanks and drain fields to be at least 15m from private water wells or surface water requiring lot sizes to be large enough to accommodate this regulation. The same regulations state that these systems must be at least 91m from public supply water wells. Furthermore, minimum lot size for a subsurface absorption field with a percolation rate of 30 minutes or less containing a public water supply is 0.2 hectares and for percolation rates greater than 30 minutes minimum lot size is 0.4 hectares (ODEQ 2004). When an onsite wastewater treatment system is needed, waterfront property owners should consider more specialized systems such as aerobic units or public wastewater treatment systems (Scheinkman et al. 2001).

Soil suitability is also an important consideration in site selection. On-site wastewater treatment systems require appropriate soil characteristics to provide effective wastewater treatment. Soil conditions must allow for absorption and filtration of wastewater. Most homes along the shoreline of Grand Lake utilize septic tank-lateral line systems for disposal of domestic waste; however, the geology of this lake consists of very shallow soils overlying highly fractured rock formations such as chert beds (Figure 1.4). These formations allow septic effluent to move through soil too rapidly which severely restricts the efficiency of soil microbial degradation of domestic waste (OWRB 1995). In this process, wastewater contaminants adsorb onto soil molecules. Soil also contains bacteria that digest a portion of the contaminants. Each soil type may inactivate biological contaminants differently (Hain and O'Brian 1979). Wastewater moves too rapidly through soil containing too much sand. Rapid infiltration rates result in inadequate treatment time. Also, any soil containing too much clay will result in extremely slow infiltration rate or a substrate that will not pass water at all.

Table 1.1 contains information obtained from individual county soil survey maps, no soil existing near the shore line in the proposed sample areas is suitable for the proper operation of an onsite soil absorption system. The Sallisaw soil series is described as having slight limitations for absorption systems, however, this soil does not exist near the water line in any of the proposed sample coves.

Detecting Septic Input in Aquatic Systems

Literature review points to a variety of indicators for septic system effluent input and methods to evaluate them. Nutrient content, bacterial content, and certain chemicals will be evaluated during this study. It is hypothesized that samples taken at heavily developed sites will contain higher levels of these indicators than samples taken at reference sites.

Nutrient Inputs

Nutrient input by onsite wastewater disposal systems will only be addressed in this project as a relevant indicator of septic system infiltration into the lake. A study done in the Florida Keys noted up to a 5000-fold increase in groundwater nutrients due to onsite wastewater disposal systems (Lapointe et al. 1990). Nutrients being evaluated are nitrogen and phosphorous. Current samples being taken from a series of developed sites and reference sites contain Total-P levels of 0.05 mg/L for developed sites and 0.02 mg/L for reference sites, possibly indicating slightly elevated P-levels in developed areas utilizing septic systems. Phosphorous limited eutrophication is presently occurring in

Grand lake as a whole (OWRB 1995), which may result in elevated phosphorous levels that make it difficult to detect elevations specifically due to septic input. However, significant differences of P concentrations among sample sites within a cove may indicate a localized wastewater source that reaches surface water. The OWRB (1995) study included a model that estimated P-loading in Grand Lake at 72% loading that was attributed to nonpoint source runoff, while 28% of the loading was attributed to point source input. Of this total, estimates of P-loading from septic systems were only 0.2% of lake totals (OWRB 1995). Since this model was developed for the Grand Lake as a whole, it may not accurately estimate P-loading for septic systems that occur on a much smaller scale, for example, in coves with significant housing development along the shoreline. Brown (1992) found that phosphorous is more likely to be detected where onsite wastewater systems are sited close to areas of surface water. The majority of phosphorous from wastewater is in the form of soluble orthophosphate (PO_4^{3-}) (McCray et. al 2005) and was used as an indicator of wastewater contamination in Grand Lake. Anaerobic conditions prevail in septic tanks resulting in nitrogen in the form of soluble ammonium and organic nitrogen (Brown 1992). Septic system effluent generally occurs in an aerobic environment resulting in the nearly complete oxidation of ammonium to nitrate (NO_3^-) within one-half meter below the absorption field trenches (Whelan and Barrow 1984). Effluent nitrogen, apart from that removed by vegetation, remains relatively unchanged in concentration in its natural form as it percolates into the groundwater (Whelan and Barrow 1984).

A study conducted in Missouri found elevated nitrogen (NO_2+NO_3) levels in a creek that was attributed to close proximity of homes with septic tanks (Schumacher 2001). Another study for the USEPA indicates that septic tank effluent will likely lead to increased concentrations of nitrate in groundwaters in areas of high densities or failed systems (Jones and Lee 1977). Given the close proximity of onsite wastewater disposal systems to the waterline on Grand Lake, it is reasonable to predict that nitrates could be detected in localized areas of high housing density and age. However, in areas where an absorption field is in poorly drained soil or below floodplain level, nitrification may be limited which may result in ammonium being the more reliable wastewater indicator due to anaerobic conditions in soil.

Bacterial Inputs

Private onsite wastewater treatment systems can be a major source of human enteric pathogens into the environment (Borchardt et al. 2003). Also, while enteric viruses may be present naturally in aquatic environments; these organisms are more commonly introduced through human sources such as leaking sewage and septic systems (Fong and Lipp 2005). Failed septic systems can also release pathogens on top of the land surface due to age or system neglect that results in system failure (Borchardt et al. 2003). Effluent released from septic systems directly to the subsurface contains microorganisms that are removed by soil filtration and adsorption (Brown 1992). Factors that affect the fate of organisms that reach surface water are water temperature and sunlight inactivation with greater survival rates in darker, cooler conditions (Fong and Lipp 2005). Table 1.2 lists organisms Macler and Merkle (1999) determined to be of concern in groundwater. A Lipp et al. (2001) study also stated that local surface water contamination, via groundwater flow, might have resulted from shallow septic systems.

Microbiological impairment of water may be assessed by monitoring, usually for the presence of indicator bacteria such as fecal coliform and *E. coli* (Simpson et al. 2002). These bacteria are present in all warm-blooded organisms. A study of the Savannah River Basin in Georgia reported that fecal coliform load might be attributed to failure of septic systems (Georgia DNR 2004).

Pang et al. (2001) conducted a modeling study that estimated minimum distances septic systems should be from the water body (setback distance) in order to minimize bacterial exposure risk of recreational users and those utilizing the lake as a drinking water source. In order to meet the New Zealand drinking water standard of <1 plaque forming unit (pfu)/100mL for viral concentrations, the minimum setback distance for a septic system is 51m; however, if the recreational water quality guideline of 126 pfu/100mL for *E. coli* is used, minimum setback distance is 16m (Pang et al. 2001). Personal observations in the field have revealed failed systems on Grand Lake at much shorter distances than the estimated setback distances listed in the New Zealand study.

Chemical Inputs

Anionic Surfactants (Detergents)

Cleaning products containing detergents are used on a daily basis in most households. These chemicals are then discharged with wastewater and introduced into the septic system. Detergents are one of a variety of chemicals which may be used as leak indicators and which may be used to delineate septic plumes (Geary 2003). Shimp et al. (1993) determined linear alkylate sulfonate (LAS) to be effectively removed in properly functioning septic tanks by biodegradation. However, failed or improperly sited septic systems discharging effluent may allow for detergent components to reach Grand Lake with little or no degradation. A study by Nielsen et al. (2002) detected LAS residues in groundwater 11.7m from a septic tank drainfield. Actual testing of a failed system on the north end of Grand Lake revealed detergent in overland flow of effluent and in the lake down gradient of the system.

Boron

Boron levels will also be evaluated as an indicator of wastewater infiltration. There are two main routes boron can enter the aquatic environment: weathering of borate-containing rock and release of borates in cleaning products through disposal to wastewater treatment systems (Dyer and Caprara 1997). One of the principal industrial uses of boron compounds is in the production of detergents (Parks and Edwards 2005). A study of river water quality in Israel's Coastal Plain used boron as a sensitive indicator of detergents; therefore, an indicator of domestic wastewater (Bar-Or 2000). Detected boron levels above those in main lake reference sites may be attributed to septic system input.

Pharmaceuticals and Caffeine

Pharmaceuticals are used in large quantities in human health care. These compounds are designed to persist in the body and may persist in the environment also (Seiler et al. 1999). These compounds can then pass through waste water treatment plants and septic systems without being removed from the effluent (Kolpin et al. 2002). Once discharged by the septic system, this contaminated effluent can reach surface water by the processes previously described. Upon reaching surface water, the use of pharmaceuticals as wastewater indicators becomes relevant.

A literature review has indicated that most work done on using pharmaceuticals as wastewater indicator has been from sampling surface water with inputs from waste water treatment plants. However, pharmaceuticals have been listed as indicators of wastewater infiltration from septic systems (Verstraeten et al. 2005; Kolpin et al. 2002). Both steroids and nonprescription drugs were detected in over 80% of samples collected downstream of intense urbanization during the Kolpin et al. study with detergent metabolites among the highest concentrations of analytes detected. Alvarez et al. (2004a) detected both hormones and antibiotics in surface water of Chesapeake Bay Tributaries. However, not enough sampling site information was available to determine a source.

Caffeine is found in a large number of products used daily, which include: medicine, tea, coffee, and soft drinks. Caffeine is also a potential indicator of domestic wastewater because it is clearly of anthropogenic origin and often has been detected in wastewater and surface water (Seiler et al. 1999). Because caffeine is present in large amounts in coffee (346 mg/L avg), a household that consumes coffee can generate hundreds to thousands of mg of coffee daily and dispose of a large portion of this caffeine un-metabolized by pouring it down the sink (Seiler et al. 1999). As a result, caffeine has been found in several Swiss lakes at concentrations of 6-250 ng/L and the Mediterranean Sea at 4-5 ng/L (Buerge et al. 2003). The presence of even low levels of caffeine and human pharmaceuticals in groundwater with elevated nitrate, another recognized wastewater indicator, concentrations is clear evidence that domestic waste water is a source of contamination (Seiler et al. 1999). Levels of caffeine have been shown to increase in surface water with increasing population. Buerge et al. (2003) found an increase in caffeine levels from 6 ng/L in a sparsely populated area to164 ng/L in more densely populated areas. Considering the similar population densities between developed and reference sites selected for this study and the waste water treatment plant on Monkey Island, a caffeine source near the reference sites, the suitability of the current reference site may need to be re-evaluated with respect to caffeine.

Thermal Imaging to Detect Wastewater Effluent

On average, sewage effluent is much warmer than ambient ground temperature and will exhibit a different thermal signature than surrounding ground or water (USEPA 2000). To locate failed systems and pipes discharging wastewater into the lake, GRDA purchased and installed a Forward Looking Infrared (FLIR) imaging system to detect these temperature differences. A previous study by the Arkansas Department of Health (ADH) used this method to identify nineteen sites with one or more pipes that probably discharge sewage into Lake Conway (Eddy 2000). This study was conducted in November when ambient ground temperature was cold enough to allow warm wastewater effluent to show up well on the FLIR system. The USEPA (1999) also used thermal imaging to identify secondary indications of septic system failures such as: small ditches or trenches constructed by homeowners to remove the effluent from failing systems, small hoses or pipes to reroute wash water from an overloaded system, and attempts to hide a failed system with an impervious material. Scientists at Macomb County Health Department in Michigan have used infrared technology to show warmer areas of Lake St. Claire due to the possible input from failed septic systems or illegal sewage discharge (USEPA 1999). In the Arkansas study, the helicopter flew at elevations between 61 and 152 m, which allowed a view of approximately 61 m inland and 30 m of water body (Eddy 2000). For optimal results, flights need to be conducted during leaf-off conditions and during the seasonally high water table, which coincides with when most failures occur (USEPA 1999). Using this method, ADH not only identified which septic tanks were malfunctioning, it also saved money by reducing the amount of ground-truthing required (USEPA 2000).

Potential Effects of Septic Leachate on Surface Water

Septic tank leachate and runoff from failed septic systems has been listed as a source of receiving water quality degradation (Carpenter et al. 1998). In cases where failed systems result in surface pooling, overland flow can transport untreated effluent directly to sensitive zones such as reservoirs and other aquatic systems (Sherlock et. al 2002). Nutrient, bacterial, and chemical inputs that infiltrate ground and surface water through normal effluent percolation are the source of this impairment.

Based on scientific literature reviewed, nutrient input into reservoirs leads to accelerated eutrophication (LaPointe and Matzie 1996; Whelan and Barrow 1984). A study by Driscoll et al. (2003) found that septic systems are an important means of transferring nutrient-rich effluent from watersheds to surface waters of Atlantic white cedar wetlands. Groundwater enriched with septic system effluent has also been stated as a major source of nutrients in nearshore surface waters of the Florida Keys (Lapointe et.al. 1990). Currently, excessive nutrient input is causing eutrophication at much faster rate than normal in Grand Lake (Tolbert 2004). Nutrient driven eutrophication affects lake systems in many different ways. Eutrophication increases total dissolved solids, epilimnetic temperature, and organic matter settling into hypolimnion and decreases in secchi depth, dissolved oxygen (hypolimnion) during summer stratification (Mackie 2004). Mackie (2004) goes on to state that eutrophication decreases species diversity at all trophic levels leaving only those organisms tolerant of the nutrient enriched conditions.

Septic system effluents are considered a risk to human health from bacterial and pathogen contamination (Ritter et al. 2002). Human wastewater may contain more than 100 viral and several bacterial and protozoan pathogens that can cause disease in humans (Pang et al. 2001). A study in Missouri observed high *E. coli* levels in a creek that was attributed to upstream septic systems (Schumacher 2001). Grand Lake is used extensively for recreational activities such as boating and swimming and has been found contaminated with pathogens posing a risk to recreational users (Tolbert 2004). Microbiological impairment of water can be determined by the presence of *Escherichia coli* and fecal coliforms, which may also signal the presence of enteric pathogens, putting human health at risk (Simpson et al. 2002).

Chemicals such as pharmaceuticals have recently been a source of concern for surface water contamination. Toxicological concerns have increased since these compounds may act as endocrine disruptors and cause developmental effects in wildlife (Ritter et al. 2002). Verstraeten et al. (2004) detected chemicals such as caffeine, antibiotics, and other pharmaceuticals in drinking water which likely originated in septic system effluent. Restrictions for pesticides and industrial chemicals are commonplace; however, few such restrictions address commonly used household chemicals, pharmaceuticals, and personal care products (Alvarez et al. 2004b). These chemicals are not known to be effectively removed by wastewater treatment processes (Desbrow et al. 1998). Estrogenic hormones from birth-control pills has been detected in surface water and been shown to contribute to vitellogenin (egg yolk protein) in male fish (Huang and

Sedlak 2001; Desbrow et al. 1998). Alvarez et al. (2004a) found a suite of endocrine disrupting hormones and antibiotics present in surface waters of Chesapeake Bay Tributaries. Little is known about the risks these chemicals pose to human health. This emphasizes the need for more study in this area for characterization of this risk. For this study a range of parameters relating to general water quality, nutrient, bacterial, and chemical input will be evaluated to determine if impairment of water quality is occurring.

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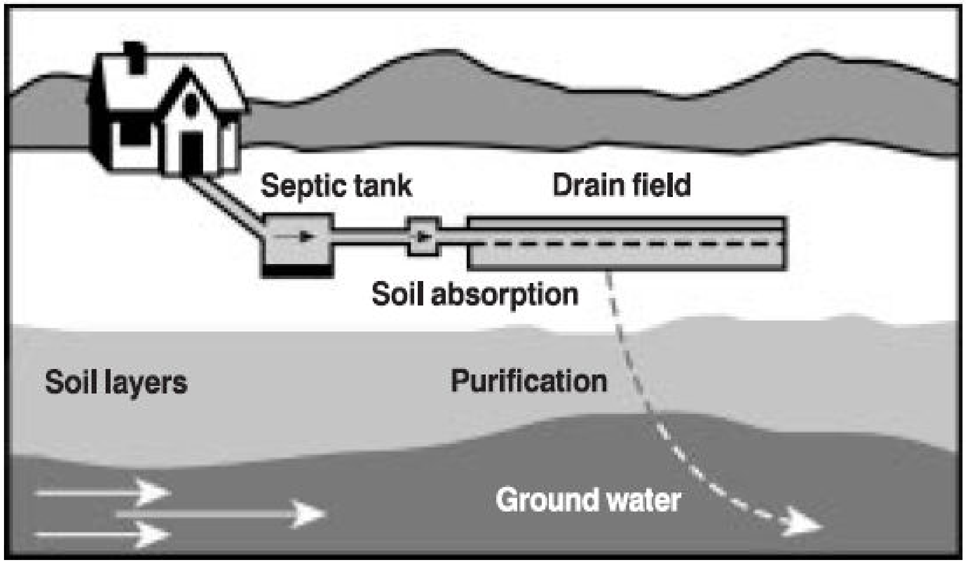

Figure 1.1. Typical onsite wastewater treatment system. (From USEPA Onsite Wastewater Treatment Manuel 2002).

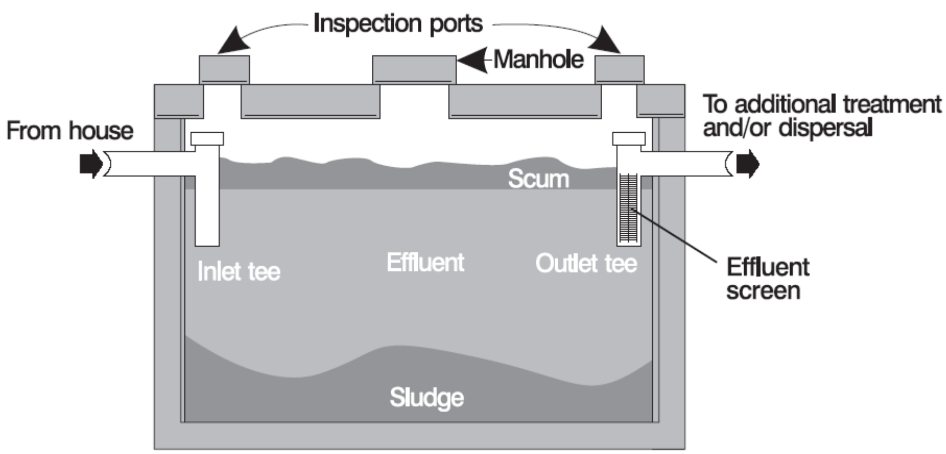

Figure 1.2. Typical single-compartment septic tank. (Picture courtesy of USEPA Wastewater Treatment Manuel 2002).

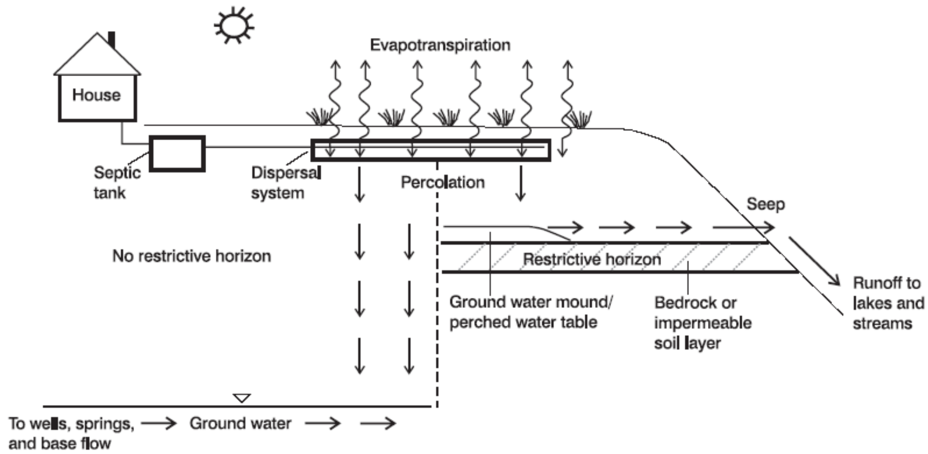

Figure 1.3. Fate and transport of septic system effluent. (From USEPA Wastewater Treatment Manual 2002).

Table 1.1. Soil type locations and limitations for septic systems

Soil Type	Location	Septic System Limitations
Clarksville	All sample sites	Severe: steep slopes, lateral seepage, chert beds 2-6 ft
Baxter	Duck Creek	Moderate: slow percolation rates
Elderado	Duck Creek Reference Site Hickory Cove	Moderate: stony subsoil, sloping topography, rapid infiltration rates resulting in diminished treatment
Dennis	Duck Creek	Severe: slow percolation rates
Nixa	Ketchum Cove	Severe: shallow depth to rock, slow percolation rates
Staser	Woodward Hollow	Moderate: flooding every 15-20 yrs, submerged septic systems during flood events
Sallisaw	Hickory Cove Woodward Hollow	Slight

Photo Courtesy of Dr. Bryan Carter, Oklahoma State University

Figure 1.4. Soil Profile from Grand Lake area showing Chert Bed.

Table 1.2. Pathogens in septic effluent (Macler and Merkle 1999).

Organism	Associated health effects
<E6> Viruses</E6>	
Coxsackie	Fever, pharyngitis (sore throat), rash, respiratory disease, diarrhea, hemorrhagic conjunctivitis, myocarditis, pericarditis, aseptic meningitis, encephalitis, reactive insulin-dependent diabetes, hand, foot and mouth disease
Echo	Respiratory disease, aseptic meningitis, rash, fever
Norwalk	Gastroenteritis (fever, vomiting, diarrhea)
Hepatitis A	Fever, nausea, jaundice, liver failure
Hepatitis E	Fever, nausea, jaundice, death
Rota	Gastroenteritis (fever, vomiting, diarrhea)
Enteric adeno	Respiratory disease, hemorrhagic conjunctivitis, gastroenteritis
Calici	Gastroenteritis (diarrhea)
Astro	Gastroenteritis (diarrhea)
<E6>Bacteria</E6>	
Escherichia coli	Gastroenteritis (diarrhea)
Salmonella spp.	Enterocolitis (fever, diarrhea, vomiting), endocarditis, meningitis, pericarditis, reactive arthritis, pneumonia
Shigella spp.	Gastroenteritis (diarrhea, fever, vomiting), reactive arthritis
Campylobacter jejuni	Gastroenteritis (diarrhea, fever, vomiting), Guillain-Barre syndrome
Yersinia spp.	Diarrhea, reactive arthritis
Legionella spp.	Legionnaires' disease, Pontiac fever, death
Vibrio cholera	Diarrhea, vomiting, death
<E6>Protozoa</E6>	
Cryptosporidium parvum	Diarrhea
Giardia lamblia	Chronic diarrhea

CHAPTER II

USE OF WATER QUALITY MEASUREMENTS TO DETECT POTENTIAL SEPTIC SYSTEM INPUT AT GRAND LAKE, OKLAHOMA

INTRODUCTION AND OVERVIEW

Wastewater production of the average household in the United States is approximately 7800 liters/person/month (USEPA 2002). While the majority of this is treated by municipal wastewater treatment plants (WWTP), approximately 25% of domestic wastewater is treated by onsite wastewater treatment systems in the U.S. (USEPA 2005). Under normal operating conditions, a properly functioning septic system (septic tank and associated lateral fields) should provide effective treatment of putrecible organics and other contaminants through the action of microorganisms in the septic tank and soil of the percolation field, and the chemical absorptive nature of the receiving soils (Hain and O'Brian 1979; US EPA 2002). However, failing septic systems may not provide efficient treatment and, thus, may allow contaminants to enter groundwater and/or be transported by other means to surface waters.

Septic system failures may occur due to hydraulic failure and/or treatment failure (USEPA 2002). Hydraulic failure is typically the most obvious with wastewater ponding above absorption field or backing up into the dwelling. Treatment failure occurs when effluent fails to be properly treated by soil and wastewater constituents reach ground or surface water. Improper site conditions, such as steep slopes, improper soil type, and site geology, result in a subsurface wastewater plume where partially treated sewage moves through soil pores, cracks, or ditches too rapidly for proper treatment (Swann 2001).

Septic system age is a significant factor that may lead to system failure. While most absorption fields have an expected life span of 15 to 30 years (Evans et al. 1999), Baffaut (2004) estimated failure rates of 40% for systems 37 years and older, 20% for systems 22-36 years old, and 5% failure for systems younger than 22 years. These failure rates were observed in southeastern Missouri and may not be applicable in some locations due to site differences in soil type and slope. The age at failure can also be influenced by improper placement of the septic system. For example, Day (2004) lists sites with slopes greater than 15% and less than 1.2m of usable soil depth as unsuitable for septic absorption fields and contributors to accelerated failure. Septic systems utilize soil as the medium to treat and transport effluent from the septic tank (Beal et al. 2005). Soil characteristics influence the effectiveness of effluent treatment during percolation and system efficiency is often directly related to site specific soil properties such as hydraulic conductivity, pore size between soil grains, and depth to impervious layer (Beal et al. 2005; Cheung et al. 2004; Sherlock et al. 2002). Tightly packed soil may cause hydraulic failure due to slow percolation of water causing ponding while soil with very high hydraulic conductivity may pass water too quickly to provide adequate treatment time.

Failing septic systems may be an important contaminant source; however, high numbers of properly operating systems can also be an important source of contamination. 15 septic systems per square kilometer has been quoted as the density at or above which catchment-scale impacts are likely to be observed (Whitehead et al. 2001). Lipp et al. (2001) found that the risk of wastewater contamination increased in areas of high on-site sewage disposal system densities. A number of studies have indicated septic system density as a factor in elevated concentrations of various wastewater constituents. For example, Cheung and Venkitachalam (2003) found that septic systems were an important source of P (94% as PO_4^{3-}) and Brendle (2004) found significant differences in concentrations of nitrate, chloride, and boron detected in samples taken from groundwater in areas of different septic system densities. Results of Brendle (2004) indicated a mean nitrate concentration 75 percent higher for samples from high-density areas as compared to low-density categories. Median chloride concentrations were 65 percent greater for high-density areas and median boron concentrations were 39 percent greater. A study done in the Florida Keys noted up to a 5000-fold increase in groundwater nutrients due to high densities of onsite wastewater disposal systems (Lapointe et al. 1990).

Other potential contaminants released by septic systems include a suite of chemicals used in most households. Cleaning products containing surfactants are used on a daily basis. These chemicals are then discharged with wastewater and introduced into the septic system. Detergents are one of a variety of chemicals which may be used as leak indicators and which may be used to delineate septic plumes (Geary 2003). Shimp et al. (1993) determined linear alkylate sulfonate (LAS) to be effectively removed in properly functioning septic tanks by biodegradation, while Rudel et al. (1998) found the detergent surfactant nonylphenol in treated septic system effluent.

Pharmaceuticals have been listed as indicators of wastewater infiltration from septic systems (Verstraeten et al. 2005; Kolpin et al. 2002). Both steroids and nonprescription drugs were detected in over 80% of samples collected downstream of intense urbanization during the Kolpin et al. study with surfactants and human steroids, such as coprostanol and cholesterol, among the analytes most often detected. Alvarez et al. (2004a) detected both hormones and antibiotics in surface water of Chesapeake Bay Tributaries. However, not enough sampling site information was available to determine a source.

Caffeine is also a potential indicator of domestic wastewater because it is clearly of anthropogenic origin and often has been detected in wastewater and surface water (Seiler et al. 1999). Municipal WWTP's are shown to efficiently eliminate caffeine (81-99% removal) (Buerge et al. 2003). However, caffeine is present in large amounts in coffee (346 mg/L avg); a household that consumes coffee can generate hundreds to thousands of mg of coffee daily and dispose of a large portion of this caffeine un-metabolized by pouring it down the sink (Seiler et al. 1999). This results in the discharge of caffeine by WWTP's and subsequent detection of the compound in ground and surface waters. Septic systems may also be an important source of caffeine in the environment. Sorption and sedimentation are not considered to be important sinks for caffeine and it has been shown that it is somewhat susceptible to degradation by microbes and fungi present in soil (Babu et al. 2005; Hakil et al. 1998). However, mobility of caffeine and associated geochemical processes are not well characterized and possibly allow caffeine to migrate from septic systems to surface waters.

Private on-site wastewater treatment systems can also be a major source of human enteric pathogens into the environment (Borchardt et al. 2003). Enteric viruses may be present naturally in the aquatic environment; however, these organisms are more commonly introduced through human sources such as leaking sewage and septic systems (Fong and Lipp 2005). Microbiological impairment of water can be assessed by monitoring; usually for the presence of indicator bacteria such as fecal coliform and *E. coli* (Simpson et al. 2002). These bacteria are present in all warm-blooded organisms. A study of the Savannah River Basin in Georgia reported that fecal coliform load might be attributed to failure of septic systems (Georgia DNR 2004).

Site Overview and Objectives

The Pensacola Hydroelectric Project impounds the Neosho, Elk, and Spring Rivers in northeastern Oklahoma to form Grand Lake O' the Cherokees (Grand Lake). This lake covers 18,818 hectares, holds $2.06X10^9$ m^3 of water, and is the third largest reservoir in Oklahoma with 2,092 km of shoreline. The lake spans a total of four Oklahoma counties including Ottawa, Mayes, Delaware, and Craig, while the entire drainage area of the lake covers over 2.5 million hectares and extends across state borders into Arkansas, Kansas, and Missouri.

Grand Lake is one of the most popular and fastest growing retirement locations in the United States (OSE 2004). While communities surrounding the lake have enjoyed both physical and economic growth due to the popularity of the lake, this popularity has led to extensive development of property close to the shoreline over the entire lake. For example, as of 1991, 8,093 homes were reported to have been built within 150m of the lake perimeter at flood pool elevation and 1,273 homes existed between 150 and 400m from this elevation (OWRB 1995). A number of the homes around Grand Lake are also known to be serviced by septic systems. Overall, this nearshore development has been considered a contributing factor to the lake's current state of cultural eutrophication, although the extent to which failing septic systems contribute to this eutrophication has not been specifically studied. A 1995 report on phosphorous loading into the lake estimated the input from septic systems to be 1,400-4,700 kg/yr. This study assumed housing occupancy to be 3.5 people per household for 60 days per year. However, personal observations and communication with Grand River Dam Authority (GRDA) has indicated that these assumptions may not be currently applicable with population growth and longer yearly occupancies resulting in more phosphorus input from septic systems. Karst topography (fractured limestone) present in the Grand Lake area (OWRB 1995) may be and important factor in rapid transport of marginally treated septic system effluent to surface water (Brendle 2004).

The objectives of this study were as follows:

1. Develop a geographic information system (GIS) database to identify localities along the Grand Lake shoreline that could pose a high risk for septic system failure and potential input to the lake. Key parameters considered in this database were the density and age of housing developments and the soil type the development occurred.
2. Determine if selected water quality parameters from lake samples collected near the sites characterized with GIS were significantly associated with the risk factors identified for septic system failure.

3. Compare water quality variables between sampling sites to determine any spatial trends (e.g. a longitudinal sequence from upper reservoir to lower near the Pensacola Dam).

MATERIALS AND METHODS

GIS Site Classification

ArcView 3.2 Geographic Information System (GIS) Software (ESRI, Redlands, CA) was used to classify several areas of Grand Lake (Figure 1.1) according to housing density, age, and soil type. These factors were used to determine sample sites that may have the greatest potential for impact from septic systems. Section, township, and range of each house along with the year built were obtained from tax records from local county tax assessor offices in Delaware, Mayes, and Craig County, Oklahoma. These records were compared to plat maps obtained from county clerk offices to determine the location of subdivisions. The combined records provided the location of each housing addition and year of construction of each house. Septic system age was estimated based on the assumption that the year of construction of the structure was the year the septic system was installed. It does not take replacement or maintenance of previously failed systems or use of a pre-existing system by new development into account. Based on the available records, the following age classes were established: >40years, 31-40 years, 21-30 years, 10-20 years, less than 10 years, and unaged.

Spatial boundaries of each housing addition were digitized into polygons and made into GIS layers. X-Tools feature of GIS software automatically calculated the area of these polygons and density of each addition was calculated by dividing the number of houses in the polygon by its area in hectares.

Soil classification was done using Soil Survey Geographic Database (SSURGO) maps provided by U.S. Natural Resources Conservation Service (NRCS), obtained from the University of Oklahoma (http://geo.ou.edu, Accessed: 1/10/2006.), and added to the areas classified by structure age and density. These digital maps were evaluated in conjunction with paper copy soil survey books for Mayes, Craig, and Delaware counties, OK (USDA 1973, 1972, and 1970). Soil survey books contained information on septic system limitations for each soil type. This information was merged with information contained in the GIS overlay maps to create a database in GIS overlay form that lists septic system limitations per soil type within each classified housing addition.

Water Quality

Five grab samples of lake water were taken monthly from each site by boat approximately 1m from the waterline. The first field season consisted of monthly samples taken from June-October 2006, while the second field season ranged from March-September 2007 for a total of 13 months of monthly sampling. The samples were collected in 500mL polyethylene bottles and placed on ice until analyzed within 24hrs. Temperature, pH, dissolved oxygen, chlorophyll-a, turbidity, and conductivity were measured in the field using a HACH Hydrolab Quanta multimeter with a DS-5X sonde (Hach Corporation, Loveland, CO).

GRDA Coal Fired Complex Laboratory analyzed monthly samples for total phosphorous, ortho-phosphate, nitrate, nitrite, and ammonium/ammonium. Ammonium was analyzed using an Orion Model 9512 ammonium probe (Thermo Fisher Scientific, Waltham, MA) via USEPA Method 350.3 and a 1.059 conversion factor was applied to results to convert to ammonium concentration. Nitrate and Nitrite were evaluated using

EPA Method 352.1 and HACH Method 8507, respectively. Ortho-hosphate and total phosphorus were analyzed using Hach Method 8048 and Standard Methods 4500-P E., respectively. Chloride ion content was analyzed for the last sample run of the first field season then monthly during the entire second season. Chloride determinations were done on a Lachat 8000 flow-injection analyzer (Hach Corporation, Loveland, CO) following USEPA method 325.2.

Samples for microbiological analysis were collected at each site in Whirl-Pak (Nasco, Fort Atkinson, WI) sterile bags and placed on ice until samples were analyzed the same day sample were taken. Determination of *E. coli*, total coliform, and fecal coliform densities were conducted at the GRDA Pensacola Dam Laboratory in Langley, OK. Total coliform, fecal coliform, and *E. coli* bacteria were evaluated using the membrane filter technique (Hach Method 10029). This procedure requires samples to be cultured on Advantec membranes (Toyo Roshi Kaisha, Ltd, Japan) in HACH m-ColiBlue 24 broth and incubated for 24 hrs in a Lab-Line Amphi-Hi-Low Incubator (Melrose Park, IL) at 35°C for total coliform and *E. coli*. Fecal coliform procedures were the same except M-FC broth was used and culture plates were placed in a Magniwhirl waterbath (St. Watertown, WI) and incubated at 44.5°C for 24 h. Between individual samples, all equipment used in the bacterial analyses was sterilized with a MilliPore UV sterilizer (Billerica, MA).

HACH Method 8028 for anionic surfactant detection was used the first field season to evaluate detergent levels in lake-water samples using a HACH DR/890 Colorimeter. This method detects alkyl benzene sulfonate (ABS) or linear alkylate sulfonate (LAS) (both components of household detergents) by extraction into benzene.

Boron levels were also evaluated using a HACH DR/890 Colorimeter via HACH Method 10061. Consistent results below minimum detection limit (MDL) resulted in the detergent tests being excluded from analysis during the second field season.

Hydrophilic wastewater compounds were evaluated by deploying Polar Organic Chemical Integrative Samplers (POCIS) obtained from Environmental Sampling Technologies (EST) Inc. (St. Joseph, MO). POCIS discs consist of Oasis HLB (Waters Corp., Milford, MA) sequestration medium enclosed within a micro porous membrane for the integrative sampling of hydrophilic organic chemicals (Alvarez et al. 2004b). POCIS samplers were deployed in Dripping Springs, Hickory Cove, an undeveloped cove, two sites on the main lake body, and the outfall of the City of Grove municipal waste water treatment plant for a total of fifteen samplers (Figure 2.1). Four samplers for each cove, with three samplers within the cove and one suspended from a buoy at the mouth of the cove. Deployment configuration consisted of three or six discs contained in a protective stainless steel perforated canister. Three-disc samplers (small canisters) were deployed at all but three sites while remaining sites had six-disc samplers (large canisters) with the additional three discs serving as replicates. A field blank of three discs was opened and exposed to air at each POCIS-deployment site to indicate any chemical exposure while discs were being handled and placed on-site. An additional set of three were used as a field blank at the waste water treatment plant outfall only.

Following a 48-d soak time, all samplers were retrieved and replaced with multi-plate samplers (discussed below). POCIS samplers were then transported from the site in air-tight metal cans (Alvarez et al. 2004b) which were held at -20°C until shipped overnight on ice to EST, Inc. for extraction. POCIS discs were individually cleaned with contaminant-free water and disassembled (EST Labs SOP 51). The Oasis HLB membrane was then filtered with 40 mL of methanol and the extract was collected in a 125 mL flask and evaporated under nitrogen to 1-2 mL (EST Labs SOP 52). The extract was then filtered again, 0.5 mL of methanol was added as rinse medium (EST Labs SOP 53), and the final sample was sealed and shipped for analysis. Analysis was conducted by Oklahoma Department of Environmental Quality Environmental Laboratory using liquid chromatography-mass spectroscopy (LC-MS) and gas chromatography-mass spectroscopy (GC-MS)

Stable Isotopes

Periphyton samples were collected from eleven sites during the first field season and midges (*Diptera:Chronomidae*) were collected from twelve sites during the second field season for nitrogen isotope analysis. Periphyton was collected (Figure 2.1) and placed in plastic canisters and frozen until processing. Macro-invertebrates were removed from periphyton and the samples were oven-dried at 60° C until dry. Multi-plate samplers were deployed for 52 days during second field season and Midges collected from the samplers were placed in plastic canisters and transported to lab on ice. Midges were dried by same method as periphyton. Samples were then ground to powder using a mortar and pestle, placed in 4x6mm tin capsules (Costech Inc., Valencia, CA), and shipped in 96-well plates to the University of California Stable Isotope Facility (Davis, CA) for analysis. Stable isotope ratios of carbon and nitrogen were measured by continuous flow isotope ratio mass spectrometry (IRMS) (20-20 mass spectrometer, Sercon, Crewe, UK) after sample combustion to CO_2 and N_2 at 1000° C in an on-line elemental analyzer (PDZEuropa ANCA-GSL). The gases were separated on a Carbosieve G column

(Supelco, Bellefonte, PA, USA) before introduction to the IRMS. Sample isotope ratios were compared to those of pure cylinder gases injected directly into the IRMS before and after the sample peaks and provisional delta 15N (AIR) and delta 13C (PDB) values calculated. Provisional isotope values were adjusted to bring the mean values of working standard samples distributed at intervals in each analytical run to the correct values of the working standards. The working standards are a mixture of ammonium sulfate and sucrose with delta 15N v Air 1.33 per mil and delta 13C v PDB -23.83. These are periodically calibrated against international isotope standards (IAEA N1, N3; IAEA CH7, NBS22). Total N and C were calculated from the integrated total beam energy of the sample in the mass spectrometer compared to a calibration curve derived from standard samples of known C & N content.

Data Analyses

SAS 9.1 (SAS Institute Inc., Cary, NC) statistical software was used for all data analyses. To evaluate the influence of housing development age and density on the water quality parameters, age was graphed against density and coves relatively close in age and density were placed into age/density classes resulting in five classes. Class 1 consisted of an undeveloped cove with minor agriculture in the area. Classes 2-4 were residential additions utilizing septic systems. Class 5 represents a residential area utilizing a waste water treatment plant (WWTP). An analysis of co-variance (ANCOVA, α=0.05) was applied to water quality parameters with applied co-variates: Temperature as a surrogate for season and year to investigate the variation due to increased lake inflow during the second field season. Logistical regression analyses were also performed to further characterize the relationship between age and density and the measured water quality parameters.

One-way analysis of variance (ANOVA) was conducted to determine significance of sample site, water temperature, and site-temperature interactive effects. All sites were then compared using Tukey-Kramer post-hoc adjustment and significance was determined at $\alpha=0.05$ (Freund and Wilson 2003).

Periphyton samples (n=3) analyzed in the 2006 field season were compared using an ANOVA model and also compared using Tukey-Kramer post-hoc adjustment and significance was determined at $\alpha=0.05$. Pearson correlation coefficient was applied to determine the strength of the correlation between nitrate and N^{15} isotope enrichment.

RESULTS

Age and Density Relationship to Water Quality

Samples sites (Fig. 2.1) were separated into five classes based on mean age and household density (Fig. 2.2). The undeveloped site (UD) was the sole location in Class 1. Class 2 included Woodward Hollow (WH) and Dripping Springs (DS) and Class 3 was comprised of Hickory Cove (HC) and Duck Creek (DC). Class 3 sites had similar housing density as compared to Class 2, but the mean age of the homes was slightly greater. Class 4 included Ketchum Cove (KC) and Cedar Cove (CC) which had the oldest homes of any of the sites. Class 5 was comprised of the Monkey Island site (MI). The average age of homes at this site was similar to that in Class 3, but the density of houses was the highest among the sample locations.

The results of comparisons of the water quality variables between the age/density classes are summarized in Table 2.1. This table includes the expected trend for a parameter that increases with higher age/density class (Ideal Increasing Parameter) and that for a parameter that decreases as the age/density class increases (Ideal decreasing parameter). No statistical difference between classes was observed for conductivity, boron, and nitrate, although differences between the groups were observed for the other parameters. Most notable is that Class 1 (the undeveloped site) had a significantly higher pH ($p=<0.0001$), dissolved oxygen ($p=<0.03$), total coliform ($p=<0.008$), nitrite ($p=<0.008$), and total phosphorous ($p=<0.007$) than all other classes. Fecal coliform and *E. coli* counts for Class 1 were no different than an Class 5 (a heavily developed site utilizing a WWTP) and both classes were significantly lower than all others ($p=<0.004$). The remaining differences between the age/density classes did not indicate any particular trend associated with the sampling site groupings and none of the parameters measured matched either of the "Ideal" scenarios with respect to differences between classes.

Results of linear regression analysis are listed in Table 2.2. Housing age was shown to be a significant source of variation in conductivity, dissolved oxygen, and ammonium only. However, r^2- values only explained a range of 1.1-3.7% of total variation in measured parameters due to housing age. Housing density was a significant source of variation in seven variables listed in Table 2.2. Although significant, r^2-values indicate a range of only 1.1-2.1% of variation observed in any parameter was due to housing density.

Temperature Relationship to Water Quality

Increased rainfall during the 2007 season resulted in higher lake levels and runoff than observed in 2006. Mean lake inflow for 2006 was $5.87 x 10^7 m^3$, while that for 2007 was 1.16x$10^9$$m^3$.Comparisons of data from both field seasons reflected this higher flow regime and water quality data are presented separately by each season as a result.

2006 Water Quality Data

The average temperature for each of the sites sampled in 2006 and 2007 is presented in Figure 2.3a and 2.3b. For 2006, site temperatures ranged from 20.3°C to 32.3°C (Fig. 2.3a). A significant site-temperature interaction was observed for most variables measured during both 2006 and 2007. In order to control for the effects of this interaction, 2006 data were separated into two temperature classes (Fig.2.4a); T1- October (>25.5°C) and T2- June-September (ranging from 25.5 °C to 32.3°C). When no site-temperature interaction was observed, data are presented graphically without temperature class groupings.

To facilitate presentation of the results, sites CC and HC will be referred to as "riverine" sites and WH, MI, DS, DC, and KC will be referred to as "reservoir" sites.

Average values and ranges for the 2006 water quality parameters are presented in Table 2.3. No significant between-site differences were observed for dissolved oxygen or pH during this sampling season. Due to equipment failure, no temperature, conductivity, chlorophyll-a, or dissolved oxygen data were recorded for sites DS, DC, and KC resulting in no T1 data presented for these sites. Conductivity (Fig. 2.5a) at riverine site HC was significantly greater than reservoir sites MI ($p=0.043$) and WH ($p=0.011$), and riverine site CC was greater than MI ($p=0.014$) only for T1. Conductivity at both CC and HC was significantly greater than reservoir sites (DC, DS, KC, WH, MI) for temperature T2 (all p-<0.0001). Chlorophyll-a levels (Fig. II.5b) were greater at riverine site HC than MI ($p=0.0014$) and WH ($p=0.0058$) for T1. Chlorophyll-a at riverine site CC was significantly greater than that at four of the reservoir sites (WH ($p=0.045$), DS ($p=0.03$), DC ($p=0.004$), KC ($p=0.03$)) for T2.

Total phosphorous (Fig. 2.6a) was significantly higher at riverine sites CC and HC than MI (all $p=<0.001$) for T1 and WH ($p=0.03$), MI ($p=0.02$), DS ($p=0.003$), DC ($p=0.005$), and KC ($p=0.006$) for T2. High variability in total P within CC at T2 resulted from one site at CC having much higher phosphorous levels than the others sampled within the cove.

Concentrations of ammonium (Fig. 2.6b), nitrite (Fig. 2.7a), and nitrate (Fig. 2.7b) were generally higher at riverine sites than at reservoir sites. A slight downstream longitudinal reduction in nitrite and nitrate was apparent although not significant. High variability for all three nitrogen species was observed in the riverine sites and in KC for ammonium only. Boron levels (Fig. 2.8a) exhibited a longitudinal effect within T1, but this trend was not observed for T2. *E. coli* densities (Fig. 2.8b) were generally higher at riverine sites, although variability was relatively high at all sampling locations. *E. coli* counts at HC were significantly greater than MI counts ($p=0.035$). Variability for fecal coliform counts was also high across all sites. The density of fecal coliforms at HC was significantly greater than coliform counts at CC, WH, MI, and KC (all $p=<0.0001$) (Fig. 2.9).

2007 Water Quality Data

For 2007, site temperatures ranged from 13.9°C to 35.2°C (Fig. 2.3b). Temperature classes for the 2007 field season (Fig. 2.4b) were as follows: T1- March and April (15°C-20.4°C), T2- May and September (ranging from 20.5 °C to 26.4°C) , T3- June, July, and August (>26.4°C) . The un-developed site (UD) was not added to the sample route until April and samples at this site were not taken until late afternoon when water temperatures were warmer. As such, there are no T1 data for this site.

Additional sites were added during the second field season to investigate water quality of the Elk and Neosho rivers that flow into Grand Lake as well as the confluence of the rivers referred to as "Combo". To facilitate presenting the data, Elk, Neo, and Combo sites will be referred to as "river" sites, CC and HC, which were just downstream of confluence, will be called "riverine" sites (still exhibited some influence of river flow but with a greater degree of development than river sites), and all down-lake sites will be referred to as "reservoir" sites.

Average values and ranges for all water quality variables measured during the 2007 field season are presented in Table 2.4 and depicted graphically in Figures 2.10–2.15. Dissolved oxygen (DO) (Figure 2.10a) exhibited increasing trends from river to reservoir sites for T1, although no significant differences were noted between site means within this temperature grouping.

As a result of equipment failure, T1 pH data were only collected for sites DC and KC (Figure 2.10b). For T2, pH was significantly lower at sites CC and MI ($p=<0.02$) than all river, riverine, and UD sites with the exception of Combo, and WH was significantly lower ($p=<0.04$) than all other sites. For T3, CC was significantly lower ($p=<0.01$) than all river, riverine, and reservoir sites UD and KC. DS was significantly lower than river sites and UD ($p=<0.008$) and WH was significantly lower than Elk ($p=0.028$) and Neo ($p=0.016$).

Significant between-site differences were observed in conductivity, although there were no clear spatial patterns within T1 or T2 (Figure 2.11a). However, for T3, Elk had significantly lower conductivity than Neo (p=0.012) and Combo (p=0.011), which could indicate the Neosho River is the greater contributor of dissolved solids. Chlorophyll-a levels (Figure 2.11b) increased from river to reservoir sites in T1, with DC having significantly higher (p=<0.03) levels than CC and WH, and levels at KC significantly greater than CC (p=0.047). No between-site differences were observed for T2, and KC had significantly higher chlorophyll-a levels than Neo (p=0.042) for T3.

Neo, Combo, HC, and CC had significantly greater (p=<0.0001) levels of total phosphorous than all reservoir sites for T1 and a general longitudinal decrease was indicated when moving from the river sites to reservoir sites near the Pensacola Dam (Figure 2.12a). However, Elk had significantly greater (p=<0.049) levels of phosphorous than DS, DC, and KC only. This longitudinal decrease was also seen in T3, with Neo, CC, and HC having significantly higher (p=<0.015) levels of total P than the reservoir sites with the exception of UD. Total P levels were also significantly greater at Combo and UD (p=<0.043) than at all reservoir sites but KC. However T3 site means indicate phosphorous levels at Combo and UD were much higher than KC and nearly significantly different (p=0.054 and 0.07, respectively) (Table 2.4). CC and HC had significantly higher (p=<0.0004) orthophosphate levels than all reservoir sites for T1, while Neo and Combo had significantly greater (p=<0.04) levels than DS, DC, and KC only for T1 (Figure 2.12b).

CC, HC, WH, and MI had significantly higher (p=<0.001) levels of nitrate than DS, DC, and KC for T1, while the river sites had significantly greater (p=<0.025) levels than KC and DC (Figure 2.13a). No significant between-site differences in nitrate was observed in T2, and CC had significantly higher nitrate levels than Combo (p=0.43) and

Elk (p=0.019) for T3. Nitrate site means for T1 were higher for all sites than T2 or T3. Nitrite and ammonium exhibited a similar trend as nitrate for T1 with higher levels in river and riverine sites than reservoir sites (Figures 2.13b and 2.14a).

Boron levels (Figure 2.14b) were greater in river and riverine sites than reservoir sites for T1, although the difference was not statistically different. Boron levels within T2 tended to increase from river to riverine sites then fall at sites down-lake from UD, with highest mean concentrations observed at HC and UD. No such patterns were observed for T3. Chloride levels (Figure 2.15a) were significantly higher (p=<0.03) at reservoir sites than riverine sites for T1, however; trends were not clear for T3 data and no significant differences were observed for T2. Total coliform counts were higher (p=<0.002) at WH than all other sites for T1 with no clear patterns observed for T2 or T3 (Figure 2.15b). Site means (Table 2.4) indicate that all riverine and reservoir sites had greater bacterial counts than rivers.

Stable Isotopes of Nitrogen

N^{15} isotope signature levels in periphyton samples ranged from 2.89‰ to 8.60‰ (Table 2.5). Periphyton samples from CC4 and DS3 were significantly more N^{15} enriched than HC4, WH1, MI1, ML1, and DC1 sites (p=<0.039), and HC1 values were significantly greater than those at DC1 (p=0.03, Figure 2.16). Isotope signature values were also graphed with nitrate levels to visually determine if any association existed between the two variables (Figure 2.17). The Pearson Correlation Coefficient for N^{15} versus nitrate levels was 0.45. Periphyton samples from CC4 had a higher N^{15} signature and higher nitrates than other sites sampled. Samples from DS3 were also more enriched, although had the lowest nitrate level of all samples.

In 2007, chironomid midge larvae were also collected for N^{15} isotope analysis and values ranged 13.42-16.87‰ (Table 2.5). Due to a small sample size (n=1), these data were not statistically analyzed. However, the midge sample from the mouth of HC was more enriched than other tissue samples, and a general downstream decrease in isotope levels was indicated (Figure 2.18).

Polar Organic Wastewater Compounds

Wastewater compounds that were detected in extracts from the POCIS samplers are listed in Table 2.6. Cholesterol was the most widely detected compound found at all sites except UD3 and HC4. The highest number of compounds was detected near the outfall of the Grove WWTP. In addition to cholesterol, coprostanol, tri (2-chloroethyl) phosphate, phenol, and nonyl-phenol were detected. Undeveloped sites had the same number of compounds detected as ML1, DS, and HC (Figure 2.19). Cholesterol and nonyl-phenol were detected at the mouth of HC and at two sites inside HC, while none were detected at HC4. Also, phenol was detected at the mouth of DS and a nearby main lake site with no detection of the compound at three sites within the same cove. These detections outside and at the mouths of sampled coves also indicate a presence of wastewater compounds in the main lake.

DISCUSSION

Relationship of Septic System Age and Density to Water Quality

Sample sites used in this study were selected by identifying heavily developed areas located near the waterline as shown on aerial photographs. In heavily wooded areas, such as Duck Creek, sites were initially selected by the number of boat slips visible off-shore. Land use was similar among all sites, with the exception of the undeveloped site, consisting of typical residential communities and associated impervious cover such as roadways, driveways, walkways to the lake, etc. This may have resulted in runoff heavily influencing local water quality. Land use at the Monkey Island site was similar to that described above except this site is also near a luxury resort with a large golf course. Aerial photography revealed more apparent impervious cover at this site than the others. Field observations indicated that all developed sites had comparable rates of housing occupancy by homeowners. Housing age did not factor into site selection and initial housing densities were estimated by sight. Age and density classification as described in the Materials and Methods was completed on 3% of the total Grand Lake shoreline which poses some limits on the range of age and density of housing developments that were available for analyses.

Previous studies indicate that age, density, and soil type are factors in septic system failure and subsequent contamination of ground and surface water. Septic systems have been noted to contaminate groundwater which can lead to a contaminant plume that reaches surface water (Winter et al. 1998). The state of Maine is currently investigating the use of records on septic system age to determine need for system replacement due to absorption field clogging or equipment failure (Dix and Hoxie 2001). Carle et al. (2005) determined that septic system age had a significant influence on nutrient, bacterial, and suspended solids in adjacent surface waters. Septic tank density has also been shown to be an important factor in nitrogen concentrations in surface waters (Hatt et al. 2004; Whitehead 2001). Soil type and useable soil depth play an important role in the degree of contamination by septic systems (Beal et al. 2005; Day 2004; Whelan and Barrow 1984). However, these studies do not indicate which factor (septic system age or density) is more or less important with respect to septic system failure and potential contaminant input. Soil types are nearly identical at all of the present study sites which meant this factor could be excluded from statistical models used to evaluate relationships between the water quality parameters and housing development attributes. Therefore, housing age and house density were given equal weighting in the development of the age/density classes used in models to investigate the potential relationship between these two factors and near-shore lake water quality.

Ideally, water quality parameters that could be positively related to the presence of septic input, such as nutrient concentrations, would be expected to increase with increasing housing age and/or density, while the opposite might be expected of parameters such as dissolved oxygen due to increased microbial respiration associated with increased nutrients. Lee et al. (2005) stated that septic system effluent can enter surface water and increase the biochemical oxygen demand (BOD) by adding nutrients that fuel primary production which reduces dissolved oxygen levels. In the case of a positively related variable such as nitrogen, the observed relationship with the age/density classes developed for use in this study would be 4>3>2>5>1, and a negatively related variable such as dissolved oxygen would be 1>5>2>3>4. Of the age/density classes that were created, Class 5 (Monkey Island) was unique in that while the site had the highest housing density (>4 houses/ha) and high age (>20 yrs), it was also served by a wastewater treatment plant (WWTP) so contamination from septic systems would be expected to be minimal when compared to other study sites.

Once variation due to seasonal flow differences and longitudinal gradients was removed, age/density class comparisons did not follow hypothesized trends for any measured water quality variable. For example, the undeveloped site (Class 1) had significantly higher levels of phosphorous and nitrite than sites with extensive development and was no different from Class 4 (which had the oldest and most dense development that uses septic systems) for ammonium, phosphate, and chloride. Reasons for this are not clear; however, this may be explained by higher than anticipated agricultural contributions of these contaminants at the undeveloped site and a strong longitudinal influence (to be explained later). Dissolved oxygen was the closest fit to the ideal hypothesis. Class 1 had significantly higher dissolved oxygen levels than all developed sites. However, this cannot be directly linked to septic system input. Class 4 did not have significantly lower dissolved oxygen levels than other developed sites of lesser age/density classes and was not different than Class 5. The undeveloped site is fairly shallow (<0.5m) and has a considerable fetch from the south and prevailing winds cause significant wave action and surface agitation which could at least partially explain the higher dissolved oxygen levels.

Carle et al. (2005) indicated that nitrogen, phosphorous, and fecal coliform bacteria in six urbanized streams subject to septic system effluent were strongly correlated to housing density. While this was not observed in nutrient concentrations for the present study, *E. coli* and fecal coliform counts exhibited a closer fit to the hypothesized relationship to age and density. Counts at the undeveloped site were not statistically different than counts at the developed site utilizing a WWTP (Class 5) and both site counts were significantly lower than all developed sites where septic systems were in use. Land use characteristics at the Class 5 site appears to be similar to all sites utilizing septic systems indicating similar sources of bacterial contributions. While septic systems may contribute to higher bacterial loading, other sources such as storm water and agricultural runoff, treated and untreated sewage input, waterfowl, and human body contact with water could also be factors (Boehm et al. 2003; Kullas et al. 2002; Paul et al. 1995; Hussong et al. 1979). Further investigation is needed to determine and quantify these sources.

As an alternative to equal weighting of age and density, regression analyses were also used to investigate the relationship between the individual water quality parameters and age and density as individual factors. Age significantly influenced variation of only three water quality variables (conductivity- r^2=0.015, dissolved oxygen- r^2=0.011, and ammonium- r^2=0.037), while density proved to be a significant source of variation in seven (conductivity- r^2=0.014, dissolved oxygen- r^2=0.021, temperature- r^2=0.013, *E. coli*- r^2=0.019, total and fecal coliform- r^2=0.019 and 0.012, respectively, and total phosphorus- r^2=0.014) indicating that, while neither of the factors explained more than 4% of the observed variation in the water quality variables, housing density may be the more important influence of the two.

Two variables, conductivity and dissolved oxygen, were slightly influenced by both age and density. Dissolved oxygen was lower with increasing age and density while conductivity increased. While no real cause and effect can be determined, the observed relationships could be due to heavier loading of organic material and other contaminants (i.e. yard clippings, fertilizer, metals) associated with large areas of impervious surface that usually co-occur with higher developmental densities (Hatt et al., 2004; Tong and Chen, 2002). Also, precipitation and local geology strongly influence lake conductivity and factors such as water temperature and primary production greatly influences dissolved oxygen (Mackie, 2004; Wetzel, 2001; Thornton, 1990).

As previously indicated, septic system failure increases with age and this may explain any observed influence of septic system age on water quality. It has been indicated that one in three septic systems built between 1950 and 2001 are in need of repair (Stout 2003). Evans et al. (1999) estimated septic system life span to be between 15 and 30 years after which failure may cause contamination to nearby waters. Septic system age classification that was performed in this study indicates many near shore septic systems fall into these age categories. However, the importance of septic system age is difficult to determine. A new system installed in unsuitable site conditions can fail to effectively treat wastewater while older well maintained systems in proper site conditions can operated effectively for many years. Septic system longevity also varies with design (USEPA, 2002). Given the observed soil and terrain conditions present at all sites in this study, it can be argued that all near shore septic systems investigated are possibly contributing to contamination of Grand Lake. In this case, septic system density should be considered the more important factor. Additional ground-truthing may result in more visual confirmation of septic system failures as was the case in the initial stages of this study in which a visibly failed system was draining into the lake. Analysis of such a site may be needed to adequately characterize the extent to which a failed system contributes to contamination. This may also involve investigation of the ground/surface water interface. Most studies involving septic systems target groundwater contamination (i.e. McCray 2005; Brendle 2004; McQuillan 2004; Geary and Whitehead 2001; Seiler et al. 1999; Rudel et al. 1998; Chen 1988). This may be due to dilution of contaminants to a level below detection limits or the high number of contaminant sources in the lake. Studies may need to assume a weight-of-evidence approach to link surface water contamination directly with near-shore septic systems.

Sample Site Relationship to Water Quality

Longitudinal differences

Reservoirs can be classified into three distinct zones, riverine, transition, and lacustrine, that occur longitudinally toward the dam. Wetzel (2001) describes these zones with respect to flow, lake morphology, and sedimentation characteristics. The riverine zone occurs where the impounded river enters the lake and normally has the highest flow velocity and suspended solid load. As flow progresses toward the dam, water velocity decreases as energy is distributed over wider areas of the transitional zone. Suspended solids entrained in the water column settle out due to the lower velocity. This process continues until water reaches the deepest portion of the reservoir near the dam. Reservoirs such as Grand Lake therefore exhibit characteristics of both rivers and lakes along the reservoir gradient. However, it is difficult to visually determine how Grand Lake fits into the described reservoir zonation. Lake width, observed from aerial photography, appears to be fairly consistent from the confluence of the Elk and Neosho rivers to the dam. Lake depth follows described longitudinal characteristics with lacustrine zone much deeper than the riverine zone.

In this study, at least some degree of a longitudinal gradient was observed for a number of the water quality variables. Both total phosphorus (TP) and orthophosphate decreased from the upper reservoir to the dam during both field seasons, although the decrease was not statistically significant for the later. Davis and Reeder (2001) observed similar decreases in ammonium and total phosphorus from the riverine to the lacustrine zone in a reservoir with this effect attributed to the association of the nutrients with suspended solids that settled out in the lacustrine zone.

All nitrogen species for the 2006 field season and T1 levels in 2007 also decreased along a longitudinal gradient. As indicated above, the effects on ammonium may be related to settling from the water column in association with suspended particles near the dam. These effects may also be due to the uptake of nutrients between the upper and lower reservoir with nitrogen introduced from the rivers being utilized by primary producers as it travels towards the dam.

The lack of a longitudinal gradient for T2 and T3 nitrate and nitrite in 2007 may have been related to water flow rates through the reservoir. It has been shown that increased flow rates allow dissolved and suspended constituents to penetrate deeper into the lacustrine zone (Wetzel 2001; Thornton 1990). Floodgates of the Pensacola dam were opened for relatively long periods during the second field season in order to relieve upstream flooding caused by high rainfall. This likely reduced reservoir retention time allowing for penetration of dissolved nitrogen species deeper into the lacustrine zone and, coupled with lake-wide nitrogen input from increased runoff, may have eliminated the longitudinal influence that was observed in the 2006 nitrogen analysis.

The lack of longitudinal trends in the 2007 nitrogen species may also be an artifact of how the data were analyzed. No site-temperature interaction was observed for the nitrogen variables in 2006 as it was in 2007 and so the 2006 data were not subdivided into temperature classes as they were in 2007. It may be that the combined data make the longitudinal effects more obvious while separation into temperature classes may mask the trend.

Seasonal Differences

A seasonal influence was also observed for a number of water quality variables. Nearly all nitrogen species in 2007 were higher within the T1 temperature category than for the other temperature classes. Nitrogen has been shown to increase from autumn until reaching a spring maxima when primary production utilizes nitrogen and returns it to lower summer levels (Pina-Ochoa 2006; Mackie 2004; Wetzel 2001). T1 chlorophyll-a levels mirrored nitrogen concentrations indicating nitrogen assimilation by phytoplankton. This may also demonstrate that nitrogen could be a seasonally limiting nutrient for primary production. In a review of previous lake nutrient experiments, Elser et al. (1990) reports that phosphorus is not always the primary limiting nutrient in fresh water and that nitrogen can be limiting on a seasonal basis. Downing and McCauley (1992) state that nitrogen was frequently the nutrient limiting primary production in lakes with total phosphorus concentrations greater than 0.03mg/L. Nutrient limitation has also been shown to differ on a spatial scale. Scott et al. (2005) determined that phosphorus was the limiting nutrient at the inflow of a wetland system while nitrogen was the limiting nutrient at the outflow.

After the spring decline in nitrogen, nitrogen-fixing blue-green algae could become the dominant phytoplankton species in the system. Nitrogen deficiencies result in increased nitrogenase activity in blue-green algal species allowing for N_2 fixation during conditions that limit other phytoplankton species (Horne and Commins 1987; Smith 1983). However, this may be offset by less than optimal water temperatures and photoperiod that exist at Grand Lake at the time of this decline. High water temperature appears to be an important factor in blue-green algae dominance (Roberts and Zohary 1987). Personal communication with GRDA personnel has indicated that no major blue-green algae blooms have occurred on Grand Lake in recent history and does not appear to be a major influence on nutrient dynamics.

Other Influences on Water Quality Variables

Influences of longitudinal gradient or seasonal variation were not observed for boron, chloride, or bacterial data. Boron concentrations of household wastewater range from 0.1-0.4 mg/L (McQuillan 2004). A study by the United States Geological Survey (USGS) (2000) determined two groundwater wells contaminated by septic systems to have boron levels of 0.258 mg/L and 0.284 mg/L. The same study indicated that boron was a good indicator of septic system contamination because background levels are relatively low relative to septic system effluent, it was not biologically removed by treatment, and was not retained in the subsurface Boron concentrations in the range described above were observed at most of the sampling sites used in the present study for at least one of the temperature classes. However, no clear differences were observed between the undeveloped site and sites using septic systems for wastewater treatment. Boron is a micronutrient required by algae and is present in lakes at much higher concentrations than other minor elements (Wetzel 2001). Fluctuating flow regimes and lake mixing may make the connection between boron and septic system input difficult to characterize in a surface water system the size of Grand Lake.

Chloride content in 2007 was higher for T1 and concentrations dropped after April. However, this may have been related to increased lake flow more than seasonal influences. June 2007 is also when the area experienced heavy precipitation that resulted in a large inflow to the lake and dam release to relieve upstream flooding. This may have resulted in a dilution or flushing effect that lowered or more evenly distributed chloride longitudinally along the lake. Chloride content of natural fresh water is approximately 8.3 mg/L (Wetzel 2001) and measured chloride ranged form 2.4-15 mg/L for this study. Septic systems are known contributors of chloride to groundwater and soil is not considered efficient in its removal (Brendle 2004; McQuillan 2004; Chen 1988). This may indicate that a portion of this chloride measured in Grand Lake may be originating from shoreline septic systems; however, other sources of chloride include most forms of agriculture, road salting, deicing products, and water softeners (Kaushal 2005; Brendle 2004; Honisch 2002; Ritter et al. 2002; Chen 1988).

Developed sites had consistently higher bacterial counts than river sites with HC having the highest fecal coliform counts in 2006 and WH having the highest mean total coliform counts in 2007. However, personal communication with a landowner in HC indicated that one sample site within the cove (HC1) was frequently occupied by Canada Geese (*Branta Canadensis*) and several species of domestic and wild waterfowl were observed to occupy all sample coves. As such, these waterfowl could be contributing to observed bacterial counts. Waterfowl are known contributors of bacterial contamination to surface water (Kullas et al. 2002; Hussong et al. 1979) along with other sources such as poultry and animal feeding operations (Baffaut 2004), septic systems and WWTP's (Beal et al.2005; Verstraeten et al. 2005; Baffaut 2004; Wicklein 2004), and urban runoff (Wicklein 2004; Tong and Chen 2002; Paul et al. 1995).

Finally, there were statistical differences in conductivity and pH among sample sites that indicated no obvious trends observed. All measurements were similar to water quality studies (pH range: 6-11; conductivity: 0.230-0.509 mS/cm) previously conducted at Grand Lake by the Oklahoma Conservation Commission (1998-1999) (Canty 1999) and Oklahoma Water Resources Board (1993-2001) (OWRB 2001) indicating no major changes in these parameters during fourteen years of water quality monitoring.

Stable Isotope Analysis

Average δN-15 values for septic system effluent vary widely among literature sources, from 7.3‰ to 10.3‰ (Fogg et al. 1998), 10‰ to 20‰ (McKinney et al. 2002), and 7.6‰ to 12.1‰ (McQuillan 2004). Fertilizer nitrogen has been shown to have values ranging from -3.0‰ to 5.0‰ and chicken litter has been shown to have a δN-15 value of 7.9‰ (McQuillan 2004; McKinney et al. 2001; Graening and Brown 1999; Fogg et al. 1998; McClelland and Valiela 1997).

Cabana and Rasmussen (1996) found that primary producers and primary consumers had mean δN-15 values of 0‰ and 7.5‰, respectively under pristine conditions, and 3.3‰ and 11.0‰, respectively in aquatic environments with anthropogenically dominated inputs. Periphyton in streams passing through housing development using septic systems were found to have mean δN-15 values of 9.6‰ while periphyton in areas of development served by a WWTP had mean values of 6.7‰ (Steffy and Kilham 2004). δN-15 values observed in 2006 periphyton samples ranged from 2.89‰ to 8.39‰ with the majority of samples (n=9) above the value listed for pristine systems. While some sites on Grand Lake may have had periphyton utilizing enriched nitrogen sources; these values fall into all listed δN-15 value ranges and determining the degree that any one source contributes to the observed isotopic values presents difficulties beyond the scope of this study.

Comparison of periphyton δN-15 values to mean nitrate levels at corresponding sample sites indicate a site within CC had a relatively enriched isotope signature and high nitrate levels and a site in DS had a relatively enriched isotopic signature and low nitrate levels. Given the variability in periphyton δN-15 values (explained below) and seasonal variation in nitrate levels observed in this study, it is also difficult to determine any trends or implications of this comparison.

Studies have shown high variability in periphyton δN-15 values due to season and flow regime. Yoshioka and Wada (1994) observed periphyton δN-15 to increase from 3‰ to 8‰ during spring and then decrease to 6‰ in August. MaCleod and Barton (1998) noted a seasonal drop from 6.7‰ in the summer to 2.7‰ in autumn. Both studies indicated that increased respiration during the higher light and temperature conditions of spring and summer as compared to autumn resulted in less cellular nitrogen and less discrimination of N^{14} over N^{15} causing a higher enrichment of the heavier nitrogen isotope. Flow velocity has also been shown to produce variability in periphyton δN-15 values. Lower water velocity tends to maximize the boundary layer along the cell membrane that inhibits nutrient diffusion into algal cells and makes the cell more isotopically indiscriminate due to lower nutrient levels available for respiration. This results in more enriched signatures in lower water velocities (Trudeau and Rasmussen 2003; MaCleod and Barton 1998).

In order to eliminate this variability, chironomid midge larvae were analyzed for δN-15 in 2007. Other studies have also used primary consumers over producers to eliminate this variation (Vander Zanden et al. 2005; McKinney et al. 1999). Chironomid δN-15 values ranged from 13.42‰ to 16.87‰. It was expected that these signatures would be higher than those observed to periphyton due to the 3-4‰ increase due to trophic level increase of primary consumer over producer (Steffy and Kilham 2004). However, adding the highest value observed for periphyton and the upper range value for increase in trophic positions give a δN-15 value of 12.39‰. If it is assumed that this represents the concentration for a primary consumer in this system without any isotopic enrichment from external sources, all δN-15 values observed in chironomid tissue indicate δN-15 enrichment. This may indicate that the organisms are utilizing a source of human derived nitrogen. However, once again, characterization of a particular source is difficult and separating a single source of influence on isotopic signature from many input sources may not be possible using current techniques (Phillips and Gregg 2003).

Water Soluble Organic Wastewater Contaminants

Organic wastewater compounds were detected at all sites where POCIS samplers were deployed with the exception of HC4 and UD3. A large Bryozoan colony was found growing inside the protective sampler cage at HC4 which restricted flow to the sampler and may have lead to the lack of any detectable analytes. It is unclear why no compounds were detected at UD3. In order to actually quantify the concentration of analytes that are accumulated by a POCIS sampler, compound-specific partitioning coefficients must be determined under conditions that closely match those in the environment the samplers are deployed in. Calculation of these partitioning coefficients was beyond the scope of this study. Therefore, contaminants that were detected will only be discussed on a presence/absence basis.

Wastewater input from municipal WWTP's is a major contributor of organic wastewater contaminants to surface water (Bradley et al. 2007; Gagne et al. 2006; Sandstrom et al. 2005; Thomas and Foster 2005; Alvarez et al. 2004b; Petty et al. 2004; Kolpin et al. 2002; Schulman et al. 2002). Therefore, it is not surprising that samples from the outfall of the Grove WWTP resulted in the highest number of compounds detected at any site sampled. Detected compounds included cholesterol- a human and plant sterol, coprostanol- a human fecal steroid, tri (2-chloroethyl) phosphate (TCEP)- a common flame retardant, phenol- an anti-microbial agent, and nonylphenol- a detergent surfactant. Cholesterol was found at all sites in which contaminants were detected, and is one of the most commonly detected analytes in surface waters (Mudge and Duce 2005; Brown and Wade 1984). Both plants and animals may be sources of cholesterol. For example, Mudge and Duce (2005) listed periphyton as a source of cholesterol in surface water. Because of the ubiquity of cholesterol, coprostanol may be a better tracer of human fecal contamination (Conn et al. 2006; Sandstrom et al. 2005; Kolpin et al. 2002; Leeming and Nichols 1996; Brown and Wade 1984). Conn et al. (2006) found coprostanol present in septic tanks at concentrations up to 700µg/L. In the present study, coprostanol was only detected at the WWTP outfall.

Despite being banned from use due to toxicity (Andresen et al. 2004), TCEP was detected at the WWTP outfall. Other studies have found TCEP in WWTP effluent at varying concentrations with one facility discharging TCEP at concentrations of 33.8µg/L (Kim et al. 2007; Martinez-Carballo et al. 2007). It is surprising that TCEP was not found at any other site given its many sources in surface water. Other studies have detected TCEP in surface water, groundwater, rain, and surface runoff with only moderate degradability in surface water (Andresen et al. 2004; Fries and Puttmann 2003; Fries and Puttmann 2001). Phenol was detected at the WWTP outfall and at sites nearly 20km away at a main lake site (ML1) and at the mouth of Dripping Springs Cove (DSM).

This common anti-microbial agent has been found in onsite wastewater treatment systems, although its fate and transport through soil and into surface water is not well known (Conn et al. 2006). Phenol can be degraded in freshwater environments through photodegradation and microbial utilization (Farrell and Quilty 2002; Hwang and Hodson 1986). This may explain why phenol was not found at sites closer to the WWTP (Hickory Cove and the undeveloped site). However, the presence of this compound far down-lake may indicate another source of the contaminant.

Nonylphenol was detected at the outfall of the WWTP and in nearby Hickory Cove and the undeveloped site. This compound is common in septic system effluent and may actually be concentrated to levels that have endocrine disrupting effects on aquatic organisms by septic system treatment processes (Conn et al. 2006; Jobling et al. 1996). If the detected nonylphenol was originating from near shore septic systems, it would likely be found at sites far down lake in areas of more urban development, although studies have shown that nonylphenol is susceptible to degradation by sunlight and microbial utilization (Fujii et. al 2000; Ahel et al. 1994).

Caffeine was not detected at any site sampled in this study. This is surprising since caffeine is a widely used substance, is a common wastewater contaminant, and is easily detectable in surface water (Kim et al. 2007; Bradley et al. 2007; Gagne et al. 2006; Conn et al. 2006; Babu et al. 2005; Sandstrom et al. 2005; Thomas and Foster 2005; Kolpin et al. 2004; Petty et al. 2004; Buerge et al. 2003; Kolpin et al. 2002; Hakil at. al. 1998). Literature reviewed for this study indicates that caffeine has been detected in septic tanks at concentrations up to 450µg/L (Conn et al. 2006), WWTP effluent up to 22µg/L (Gagne et al. 2006), and surface waters up to 6.0µg/L (Kolpin et al. 2002). Studies have shown that localized microbial and fungal assemblages exposed to caffeine-containing wastewater can be conditioned to utilize the compound as an energy source and drastically reduce its concentration in surface water (Bradley et al. 2007; Babu et al. 2005; Hakil et al. 1998). This is a possible explanation for why this study did not detect caffeine in Grand Lake, although this is purely speculative.

The POCIS portion of this study was conducted during a high flow period on Grand Lake which may have resulted in detection of fewer targeted wastewater contaminants than would be found under normal or low-flow conditions. Contaminant dilution during high flow has been shown to reduce the number of wastewater contaminants detected and lower levels of detected contaminants (Kolpin et al. 2004). Further research of soil processes and contaminant transport in the subsurface is needed to better characterize the role of near shore septic systems as contamination sources on Grand Lake.

CONCLUSIONS

Based on the high density of houses in close proximity to the water line and marginal soil types, it is possible that effluent from near-shore septic systems reaches the surface water of Grand Lake. While the water quality parameters evaluated in the study could all be related to input from septic systems and/or wastewater treatment plants, they were also potentially influenced by factors such as land use, flow regime, season, reservoir morphology, and endemic biota. No clear relationship was observed between water quality and the categories used to differentiate housing age and density. This may simply indicate that if there is any input from septic systems reaching Grand Lake, the volume is not large enough to be detected with the parameters used for the study. Other influences may have made the relationship of the age/density classes unclear. For example, total phosphorus was significantly higher at the undeveloped site (Class 1) than classes with higher age and developmental density. This was likely due to the decreasing longitudinal gradient observed for total phosphorus. It is also possible that the approach used to categorize and weight the factors of age and density of houses did not accurately reflect the role these factors play in causing septic system input to the lake, and/or that the range of housing age and density used was not sufficient to capture any relationship that may have existed.

Longitudinal effects (from upper reservoir to the Pensacola Dam) were observed for total phosphorus, orthophosphate, all 2006 nitrogen species, and T1 nitrogen species in 2007, which followed a decreasing gradient towards the dam. This trend was not observed for most nitrogen species in 2007, possibly due to significantly increased lake inflow from heavy precipitation events that began in June, 2007 and continued for the remainder of the second field season.

Significantly higher nitrate levels in 2006 and high variability in the nutrient parameters were observed within Cedar Cove. The second site within the cove (CC2) contributed a disproportionately large amount of nitrogen and phosphorus to the overall cove mean. This could indicate that a smaller-scale focus (i.e. between areas within a specific cove) is warranted in future studies trying to link shoreline activities with water quality conditions in the lake.

Targeted organic wastewater contaminants were found at most sites sampled. These contaminants likely originated from a number of sources such as WWTP's, inflow from rivers, surface runoff, and possibly near-shore septic systems. Determining the magnitude of contaminant contribution that can be individually attributed to each of these sources will likely prove difficult with no clear link to any one source.

With respect to stable isotopes, chironomid midges were used as a representative consumer species in this study since they were relatively easy to obtain. While this component of the work was preliminary in nature, relatively high δN-15 values observed in Cedar Cove and Dripping Springs may indicate a source of nitrogen derived from wastewater input and suggests that further investigation of these areas may be warranted.

Future Research

The results of this study reveal many influences affecting the water quality of Grand Lake. This may suggest the need to focus future studies on indicators more firmly associated with wastewater input such as emerging chemical pollutants and stable isotopes of nitrogen. Passive sampling conducted for this study demonstrated the presence of some emerging chemical pollutants in Grand Lake. Research that provides a means to determine actual *in-situ* water concentrations of these compounds as well as a better understanding of which of these compounds are the most appropriate indicators of septic system input is necessary. Further passive sampling is needed to better understand the role of the Elk and Neosho rivers in contributing organic wastewater compounds to Grand Lake and may allow a better understanding of the importance of near shore development as a potential contributor. Increased replication of sites with near shore septic systems and main lake sites with minimal development may reveal differences not observed with a single sampling event. A detailed comparison of analytes found at the WWTP outfall and nearby Hickory Cove could provide some degree of source discrimination for the organic wastewater compounds.

As discussed previously, input of human-derived wastewater can cause enrichment of N^{15} isotope in surface water biota. The many nitrogen inputs into a freshwater lake can also cause system-wide variation in primary producer δN-15 isotope signatures which can make source differentiation difficult. The determination and utilization of a suitable primary consumer species may provide a better understanding of the influence of anthropogenic development and activities on water quality on Grand Lake. McKinney et al. (1999) found that freshwater mussels had tissue turnover rates slow enough to control spatial and temporal variation. Common freshwater mussel species are also relatively easy to identify in the field. Freshwater mussels may therefore be a better choice as a primary consumer for future stable isotope studies at Grand Lake.

Bacterial source tracking has the potential to detect the source of bacterial input to surface waters. In this approach, repetitive element PCR uses DNA primers that attach to interspersed repetitive DNA elements that produces a unique fingerprint to determine species of origin (Scott et al. 2002). Other methods classify viruses that only infect *E.* coli into sources of origin (Griffin et al. 2000). Wicklein (2004) describes a method that allows for the determination of bacterial sources based on the fact that humans are exposed to different types of antibiotics, and exposed more often than pets or wild animals. Bacteria from collected samples are exposed to antibiotics and the reaction is compared to a library of known bacterial samples (i.e. human, chicken, cow, dog, deer, etc.).

In addition to different indicators of septic system input, different techniques to identify plumes of failing septic systems also exist. On average, sewage effluent is warmer than the ambient ground temperature and will exhibit a different thermal signature than surrounding ground or water (USEPA 2000). The United States Environmental Protection Agency (USEPA) and Arkansas Department of Health have used infrared imaging to find wastewater input into surface water (Eddy 2000; USEPA 1999). Scientists at Macomb County Health Department in Michigan have also used infrared technology to show warmer areas of Lake St. Claire due to the possible input from failed septic systems or illegal sewage discharge (USEPA 1999). Through infrared aerial photographs that detect temperature differences between wastewater and surface water, theses agencies have determined that wastewater can move to surface water in a plume with defined borders. These infrared photographs can be generated using a Forward Looking Infrared (FLIR) imaging system that can be mounted on an airplane or helicopter. As of this writing, the Grand River Dam Authority has this imaging equipment and the approach could be used to locate areas of wastewater input as test areas to concentrate sampling efforts and evaluate indicators of input.

A better understanding of the groundwater/surface water interface may also be necessary to determine the effects of contaminant dilution as it enters surface water from the subsurface. For example, a failed septic system located in the early stages of this study was found to contribute detectible levels of surfactants at the water line of Grand Lake with none being detected one meter from the water line at the surface or near the bottom. This illustrates that contaminant dilution has a major effect on the detectability of chemical pollutants present in surface water. Comparisons of wastewater contaminants detected in groundwater with those detected in adjacent surface water may be valuable in characterizing the extent of contaminant dilution. This involves the use of sampling wells installed near the waterline of Grand Lake. However, rocky soils with chert beds that surround Grand Lake may make the installation of sampling wells unfeasible and landowners may be unwilling to participate in the study.

Characterizing the role of any one contaminant source in a system such as Grand Lake is a difficult undertaking. Many factors must be considered and addressed in a study of this nature. The first field season of this study occurred during drought conditions and the second in unusually high flow conditions. It is not clear how this affected the results, although one would assume that this had some influence on the water quality of Grand Lake observed in this study. Water quality of Grand Lake is also influenced by past and present land use practices not only in the immediate area but throughout the entire watershed as well as natural factors such as local geology and topography. As indicated above, the methods used in this study did not confirm that effluent from nearshore septic systems influence water quality of Grand Lake and it may be necessary to use different methods and parameters to determine this relationship. A larger scale (i.e. cove wide or targeted effluent plume) may also be a more appropriate approach. However, this study is an important first step in this process and will provide a solid foundation for future research.

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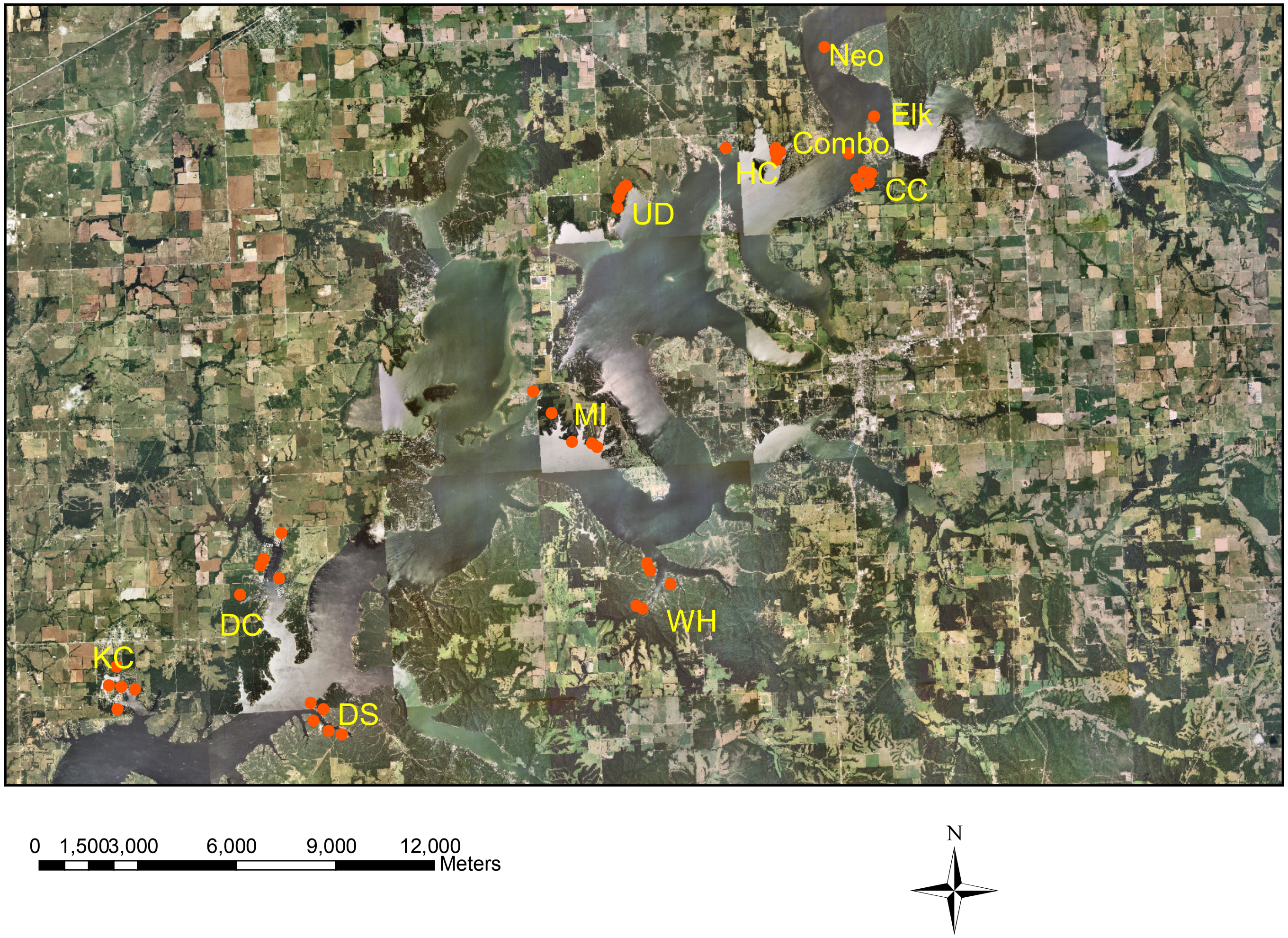


Figure 2.1. Grand Lake sample sites. Neo:Neosho River; Elk:Elk River; Combo:Confluence of Elk and Neosho Rivers; CC:Cedar Cove; HC:Hickory Cove; UD:Undevelope Site; MI:Monkey Island; WH:Woodward Hollow; DC:Duck Creek; DS:Dripping Springs; KC:Ketchum Cove.

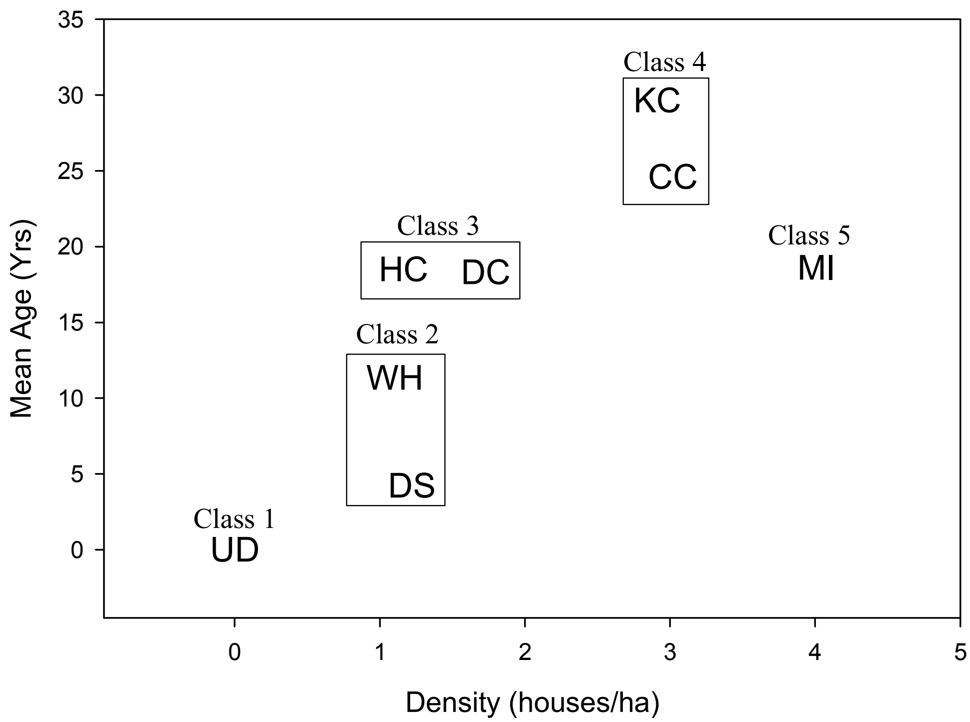
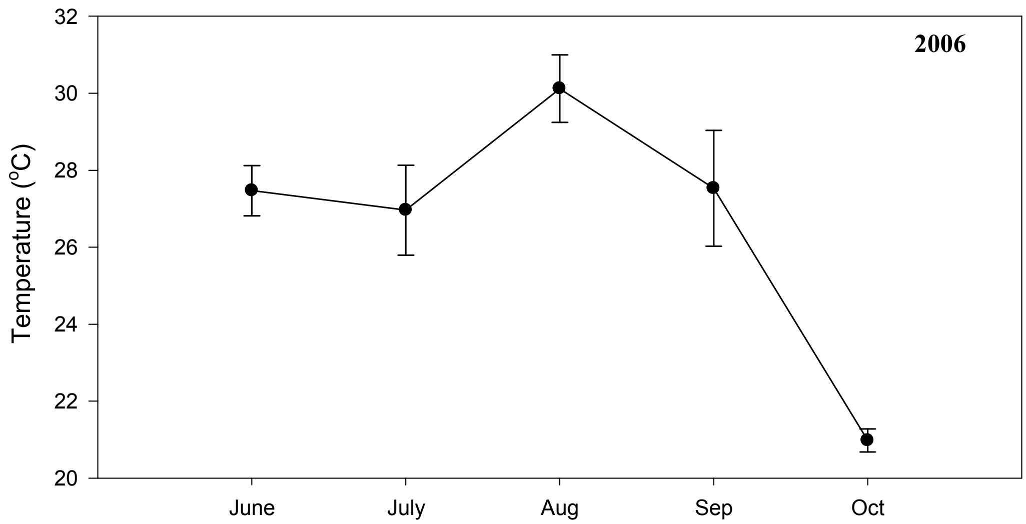


Figure 2.2. Age/density classes for sampling sites on Grand Lake. UD:Undeveloped site; WH:Woodward Hollow; DS:Dripping Springs; HC: Hickory Cove; DC:Duck Creek; KC:Ketchum Cove; CC:Cedar Cove; MI:Monkey Island (served by a WWTP).



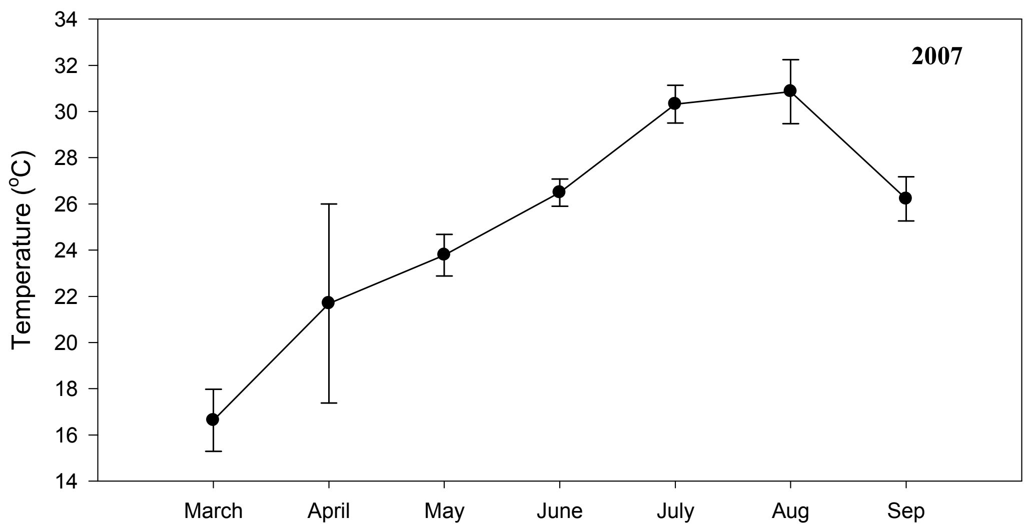


Figure 2.3. Average monthly temperatures across all of the Grand Lake sampling sites for 2006 (a) and 2007 (b). Error bars represent one standard deviation from mean.

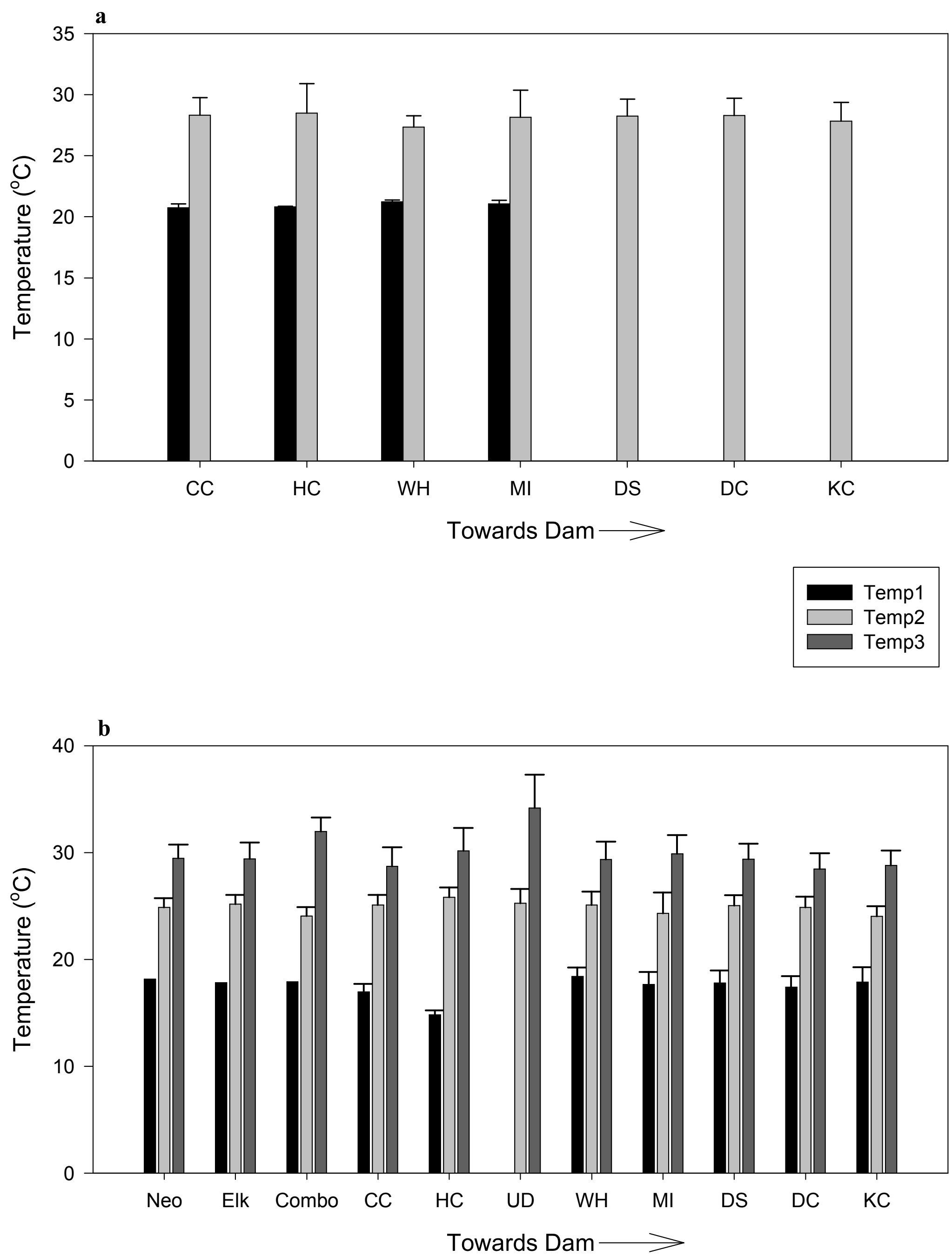


Figure 2.4. Mean temperature values for each temperature class for 2006 (a) and 2007 (b) field seasons at the Grand Lake sites. Error bars represent one standard deviation from mean. Neo:Neosho River; Elk:Elk River; Combo:Confluence of Elk and Neosho; CC:Cedar Cove; HC:Hickory Cove; UD:Undeveloped Site; WH:Woodward Hollow; MI:Monkey Island; DS:Dripping Springs; DC:Duck Creek; KC:Ketchum Cove.

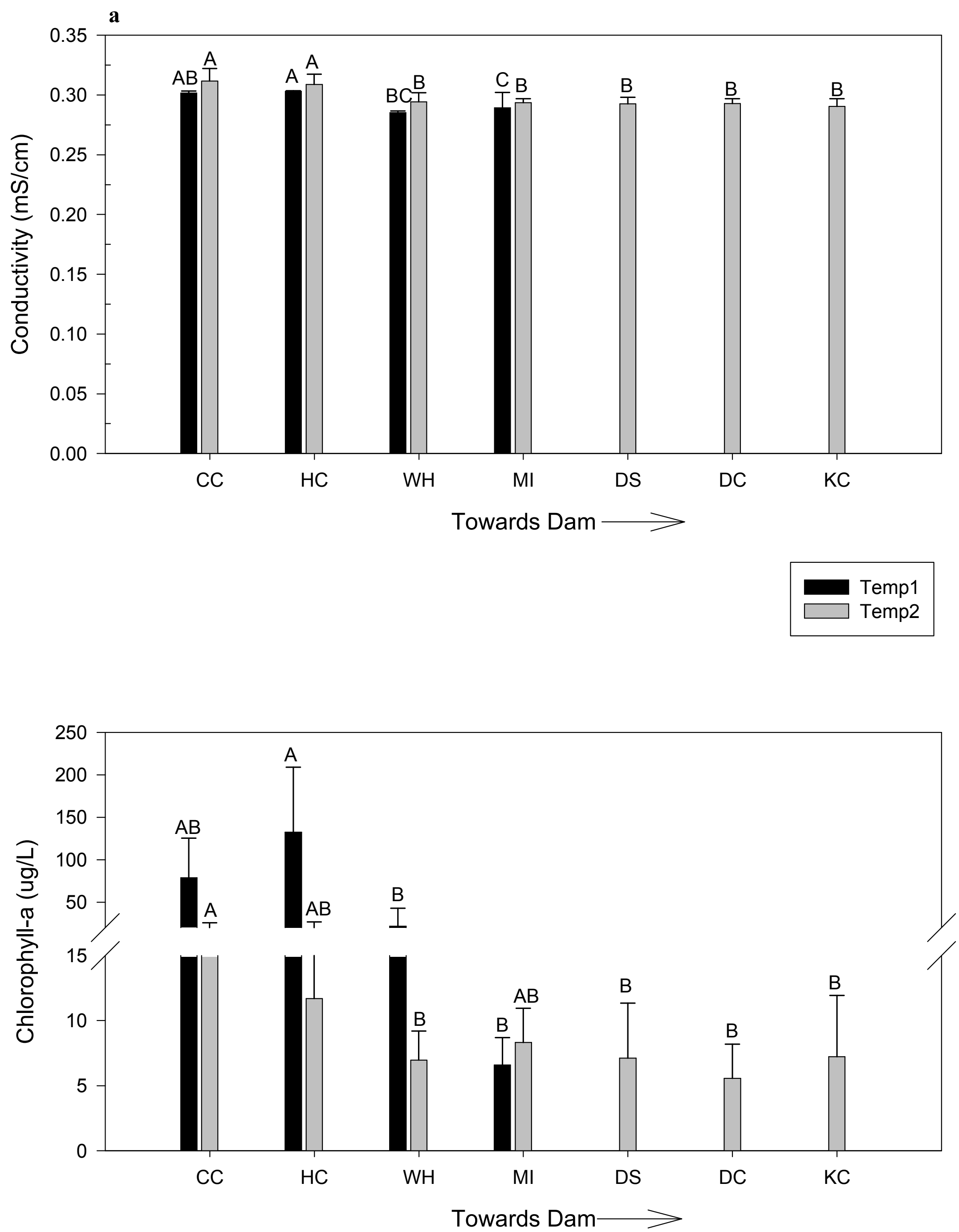


Figure 2.5. Mean values of conductivity (a) and chlorophyll-a (b) concentrations in water samples from Grand Lake for each temperature class in 2006. Bars that do not share a common letter are significantly different at α=0.05. Error bars represent one standard deviation from mean. CC:Cedar Cove; HC:Hickory Cove; UD:Undeveloped Site; WH:Woodward Hollow; MI:Monkey Island; DS:Dripping Springs; DC:Duck Creek; KC:Ketchum Cove.

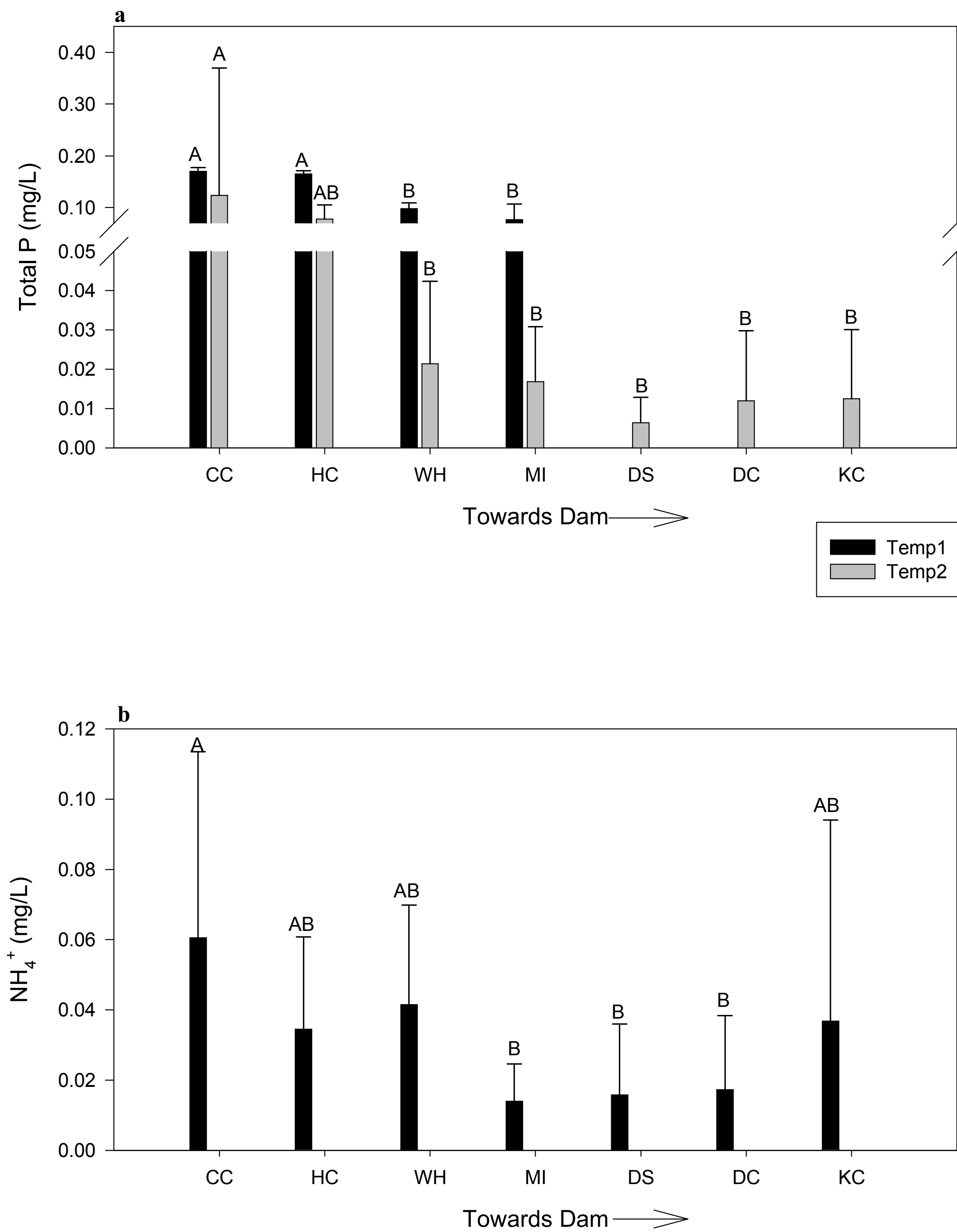


Figure 2.6. Mean values of total phosphorous (a) and ammonium (no temperature classes shown) (b) concentrations in water samples from Grand Lake for each temperature class in 2006. Bars that do not share a common letter are significantly different at α=0.05. Error bars represent one standard deviation from mean. CC:Cedar Cove; HC:Hickory Cove; UD:Undeveloped Site; WH:Woodward Hollow; MI:Monkey Island; DS:Dripping Springs; DC:Duck Creek; KC:Ketchum Cove.

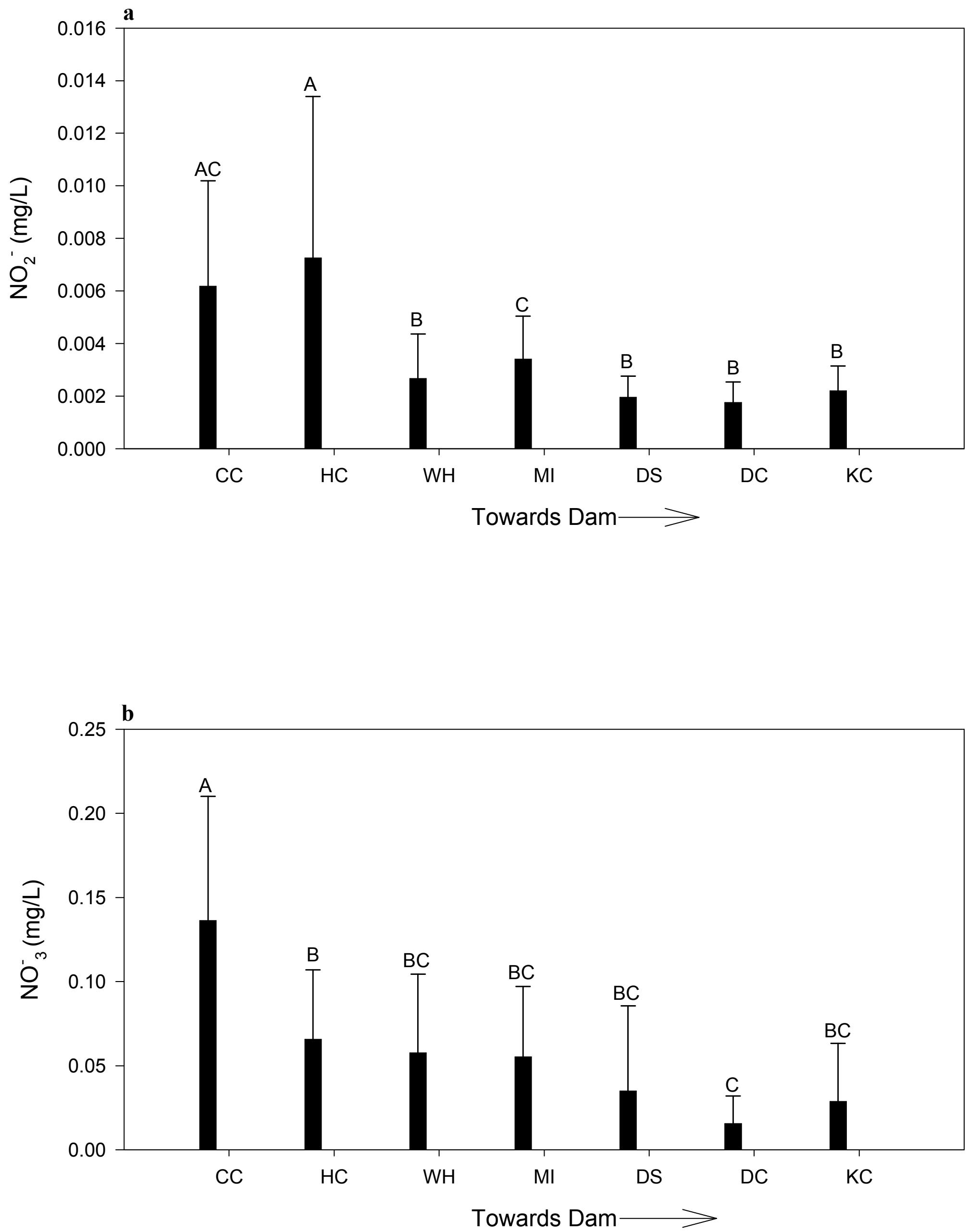


Figure 2.7. Mean values of nitrite (a) and nitrate (b) concentrations in water samples from Grand Lake in 2006. Bars that do not share a common letter are significantly different at α=0.05. Error bars represent one standard deviation from mean. CC:Cedar Cove; HC:Hickory Cove; UD:Undeveloped Site; WH:Woodward Hollow; MI:Monkey Island; DS:Dripping Springs; DC:Duck Creek; KC:Ketchum Cove.

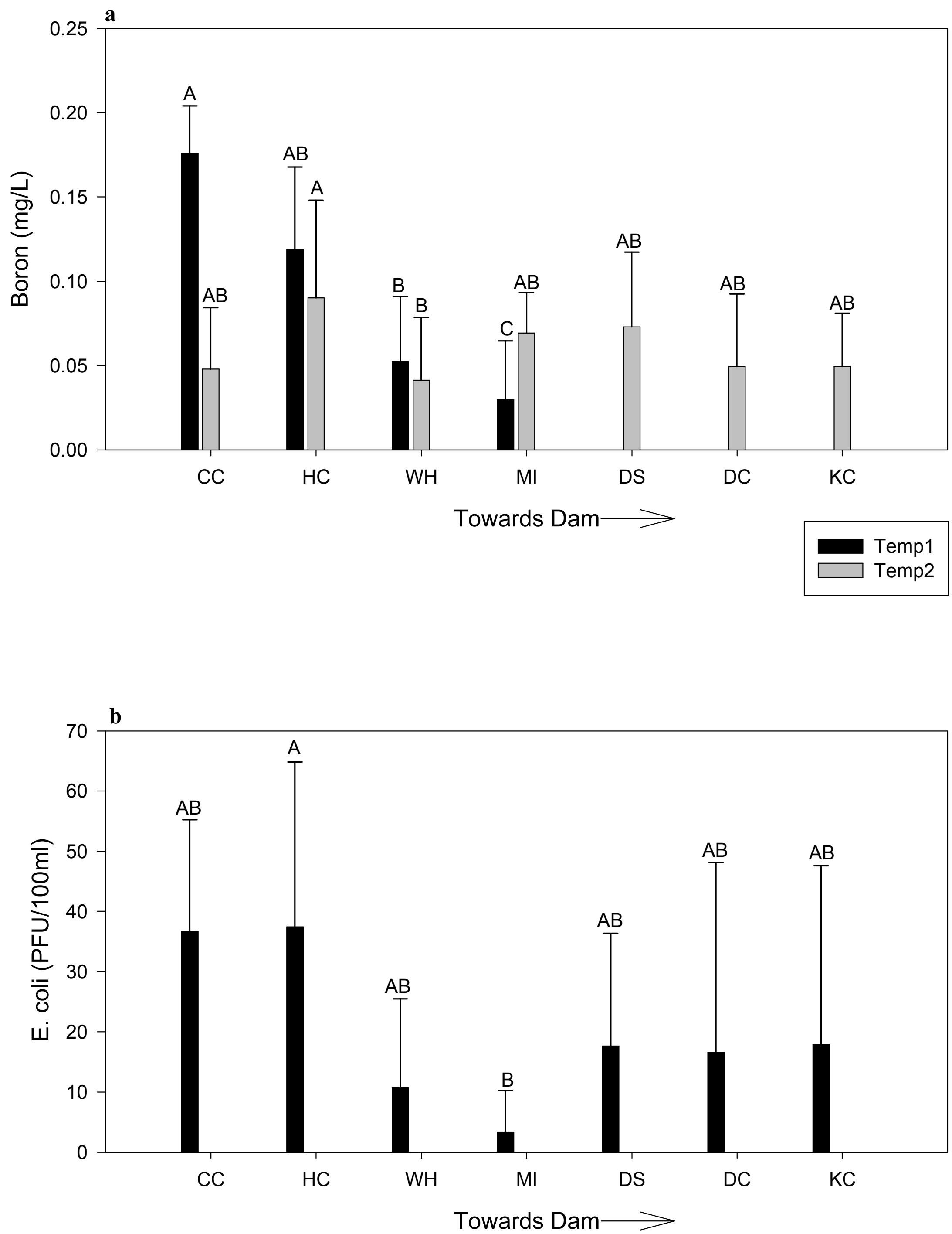


Figure 2.8. Mean values of total boron (a) and *E. coli* (no temperature classes shown) (b) concentrations in water samples from Grand Lake for each temperature class in 2006. Bars that do not share a common letter are significantly different at α=0.05. Error bars represent one standard deviation from mean. CC:Cedar Cove; HC:Hickory Cove; UD:Undeveloped Site; WH:Woodward Hollow; MI:Monkey Island; DS:Dripping Springs; DC:Duck Creek; KC:Ketchum Cove.).

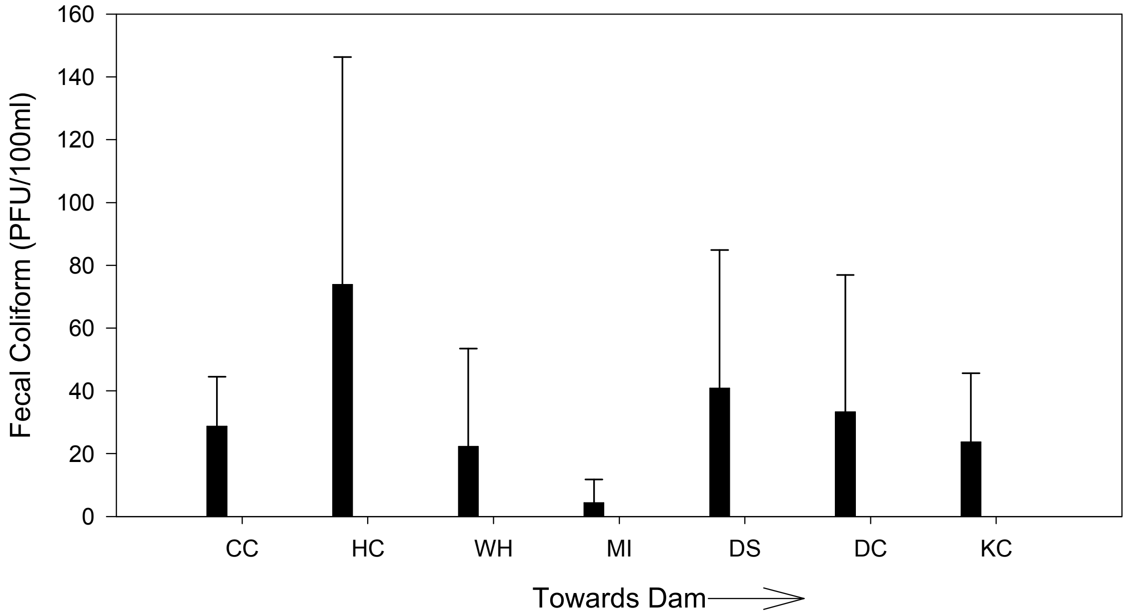
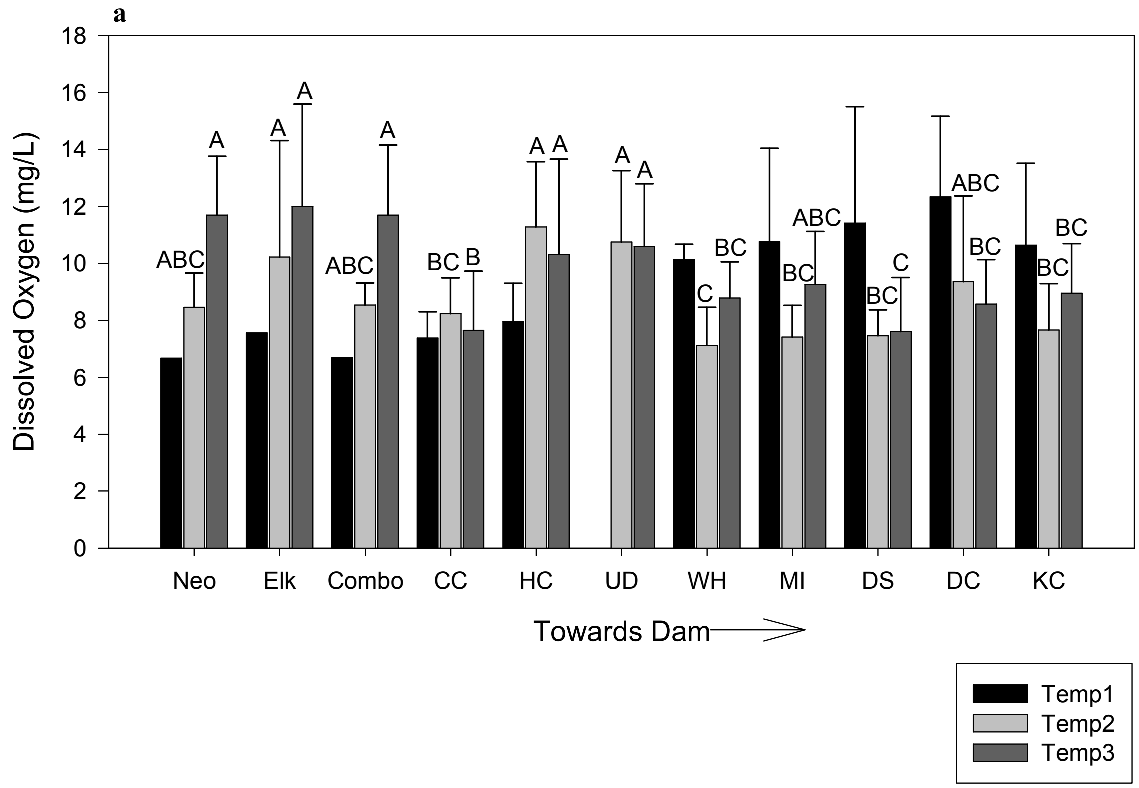


Figure 2.9. Mean values of fecal coliform counts in water samples from Grand Lake in 2006. Bars that do not share a common letter are significantly different at α=0.05. Error bars represent one standard deviation from mean. CC:Cedar Cove; HC:Hickory Cove; UD:Undeveloped Site; WH:Woodward Hollow; MI:Monkey Island; DS:Dripping Springs; DC:Duck Creek; KC:Ketchum Cove.



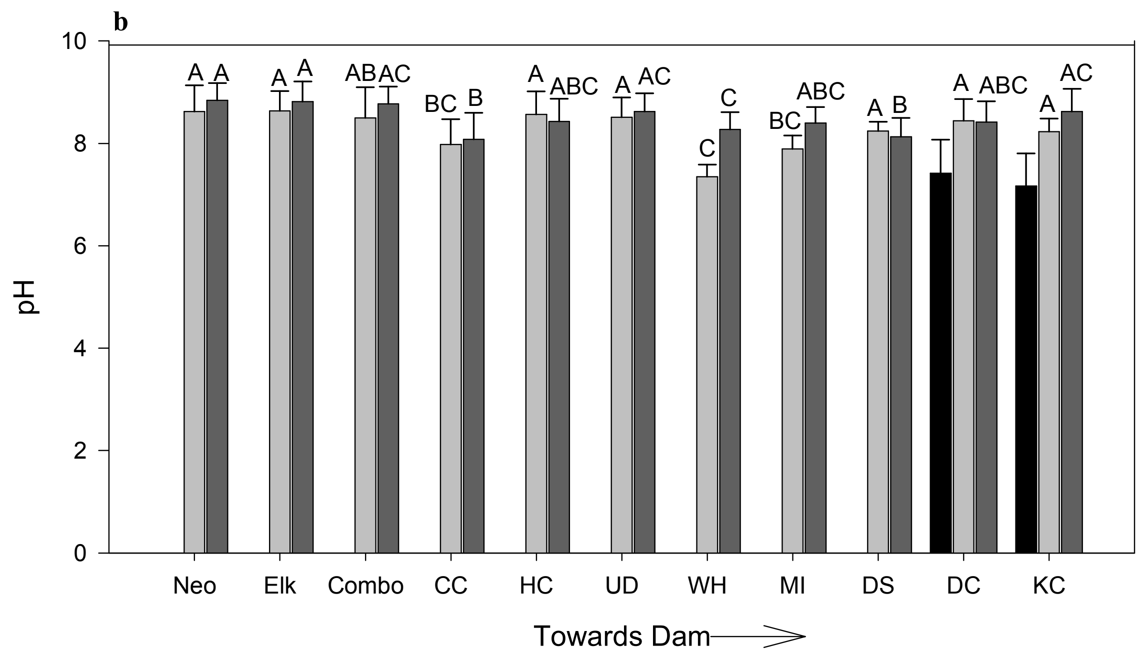


Figure 2.10. 2007 Mean values of (a) dissolved oxygen and (b) pH concentrations in water samples from Grand Lake and river sites for different temperature classes. Bars that do not share a common letter are significantly different at α=0.05. Error bars indicate one standard deviation from mean. Neo:Neosho River; Elk:Elk River; Combo:Confluence of Elk and Neosho; CC:Cedar Cove; HC:Hickory Cove; UD:Undeveloped Site; WH:Woodward Hollow; MI:Monkey Island; DS:Dripping Springs; DC:Duck Creek; KC:Ketchum Cove.

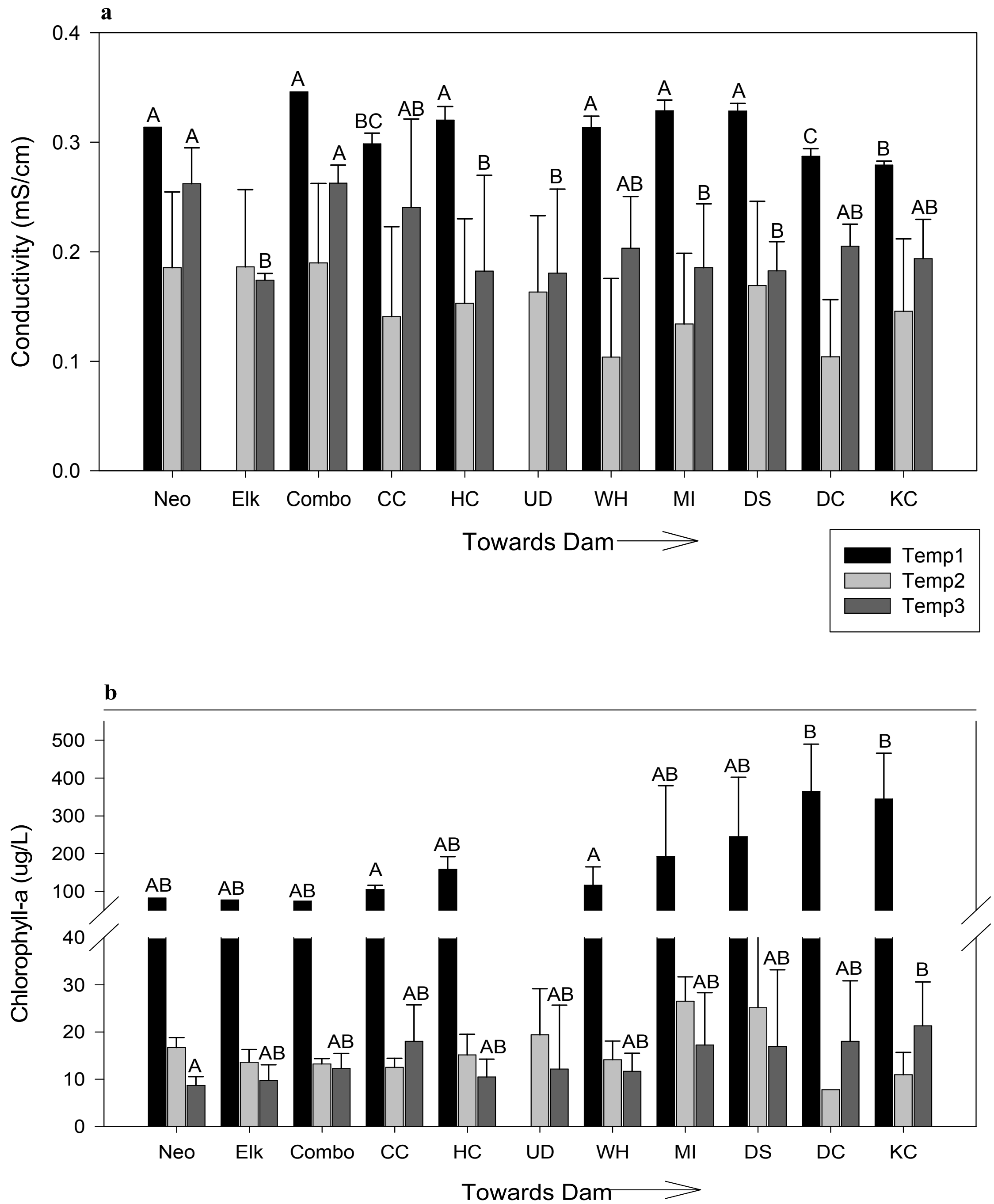


Figure 2.11. 2007 Mean values of (a) conductivity and (b) chlorophyll-a concentrations in water samples from Grand Lake and river sites for different temperature classes. Bars that do not share a common letter are significantly different at α=0.05. Error bars indicate one standard deviation from mean. Neo:Neosho River; Elk:Elk River; Combo:Confluence of Elk and Neosho; CC:Cedar Cove; HC:Hickory Cove; UD:Undeveloped Site; WH:Woodward Hollow; MI:Monkey Island; DS:Dripping Springs; DC:Duck Creek; KC:Ketchum Cove.

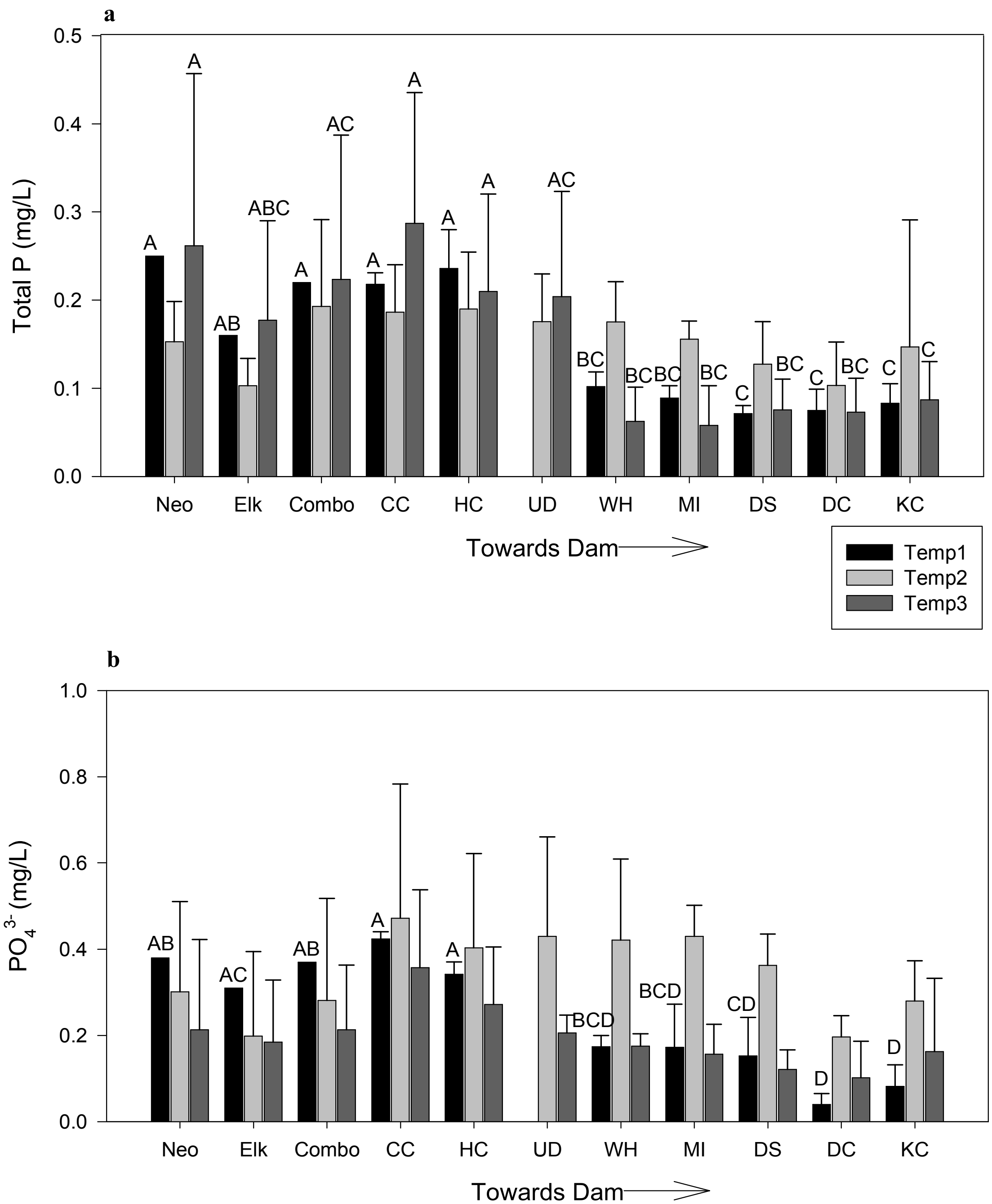


Figure 2.12. 2007 Mean values of (a) total phosphorous and (b) orthophosphate concentrations in water samples from Grand Lake and river sites for different temperature classes. Bars that do not share a common letter are significantly different at α=0.05. Error bars indicate one standard deviation from mean. Neo:Neosho River; Elk:Elk River; Combo:Confluence of Elk and Neosho; CC:Cedar Cove; HC:Hickory Cove; UD:Undeveloped Site; WH:Woodward Hollow; MI:Monkey Island; DS:Dripping Springs; DC:Duck Creek; KC:Ketchum Cove.

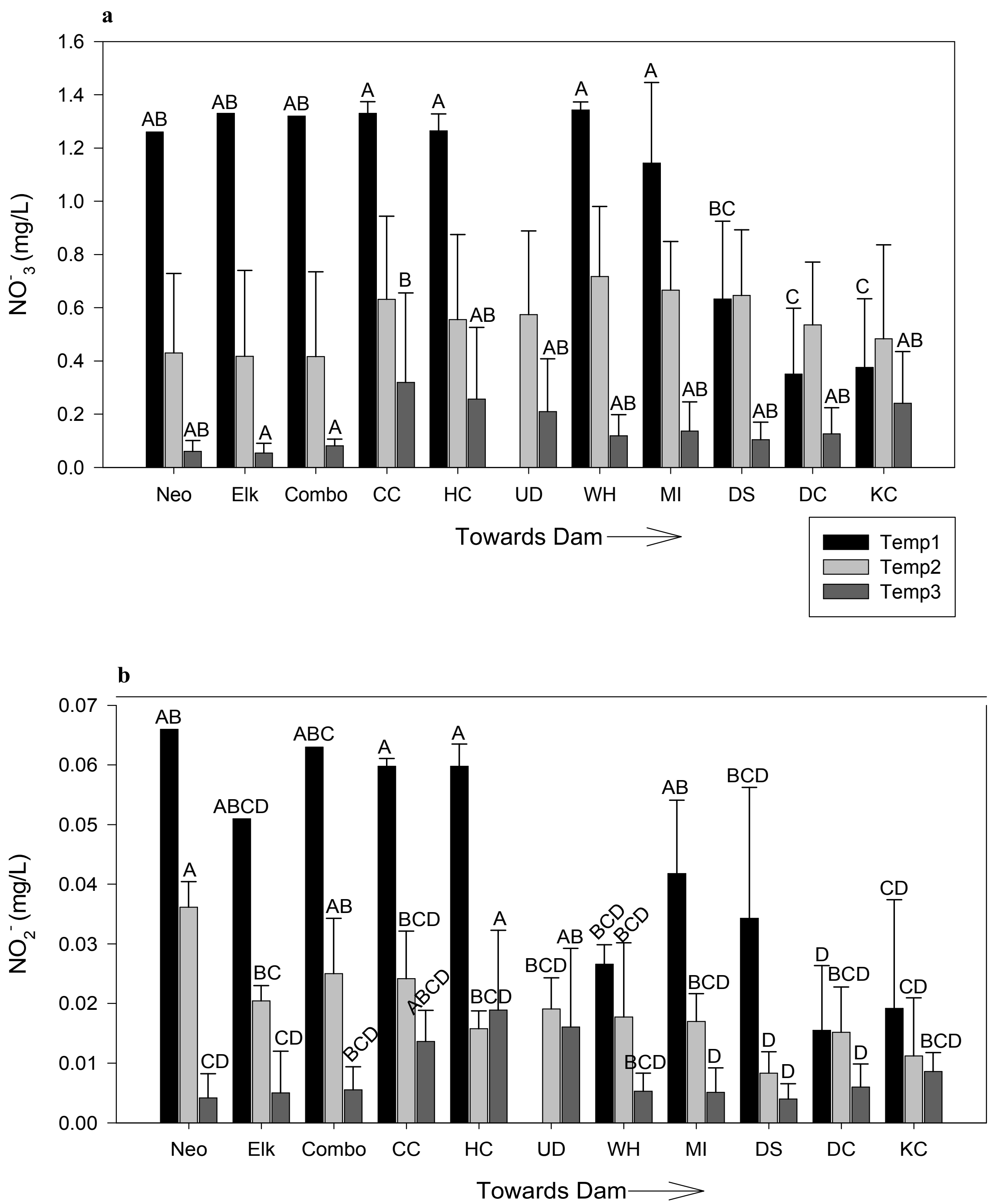


Figure 2.13. 2007 Mean values of (a) nitrate and (b) nitrite concentrations in water samples from Grand Lake and river sites for different temperature classes. Bars that do not share a common letter are significantly different at α=0.05. Error bars indicate one standard deviation from mean. Neo:Neosho River; Elk:Elk River; Combo:Confluence of Elk and Neosho; CC:Cedar Cove; HC:Hickory Cove; UD:Undeveloped Site; WH:Woodward Hollow; MI:Monkey Island; DS:Dripping Springs; DC:Duck Creek; KC:Ketchum Cove.

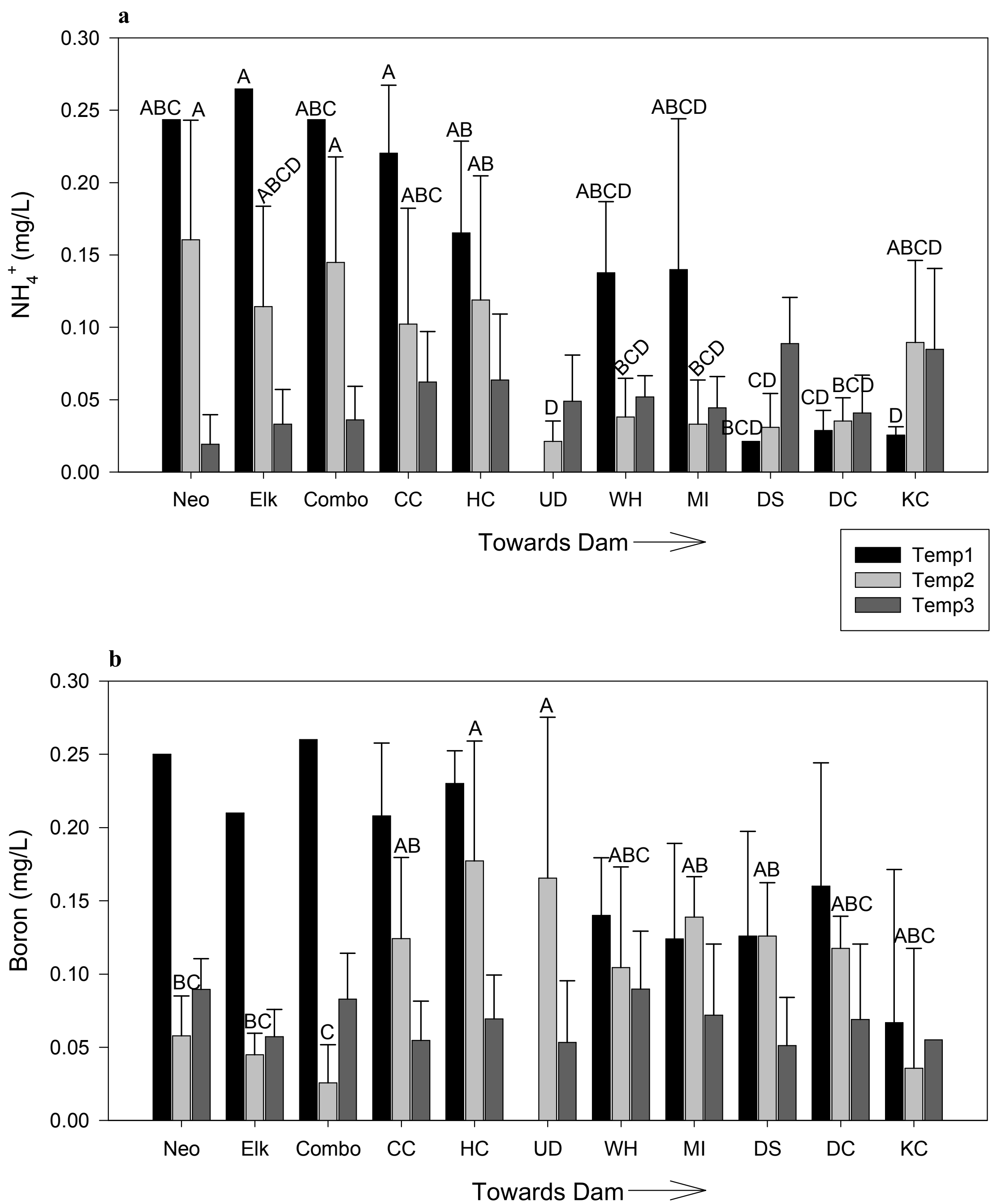


Figure 2.14. 2007 Mean values of (a) ammonium and (b) boron concentrations in water samples from Grand Lake and river sites for different temperature classes. Bars that do not share a common letter are significantly different at α=0.05. Error bars indicate one standard deviation from mean. Neo:Neosho River; Elk:Elk River; Combo:Confluence of Elk and Neosho; CC:Cedar Cove; HC:Hickory Cove; UD:Undeveloped Site; WH:Woodward Hollow; MI:Monkey Island; DS:Dripping Springs; DC:Duck Creek; KC:Ketchum Cove.

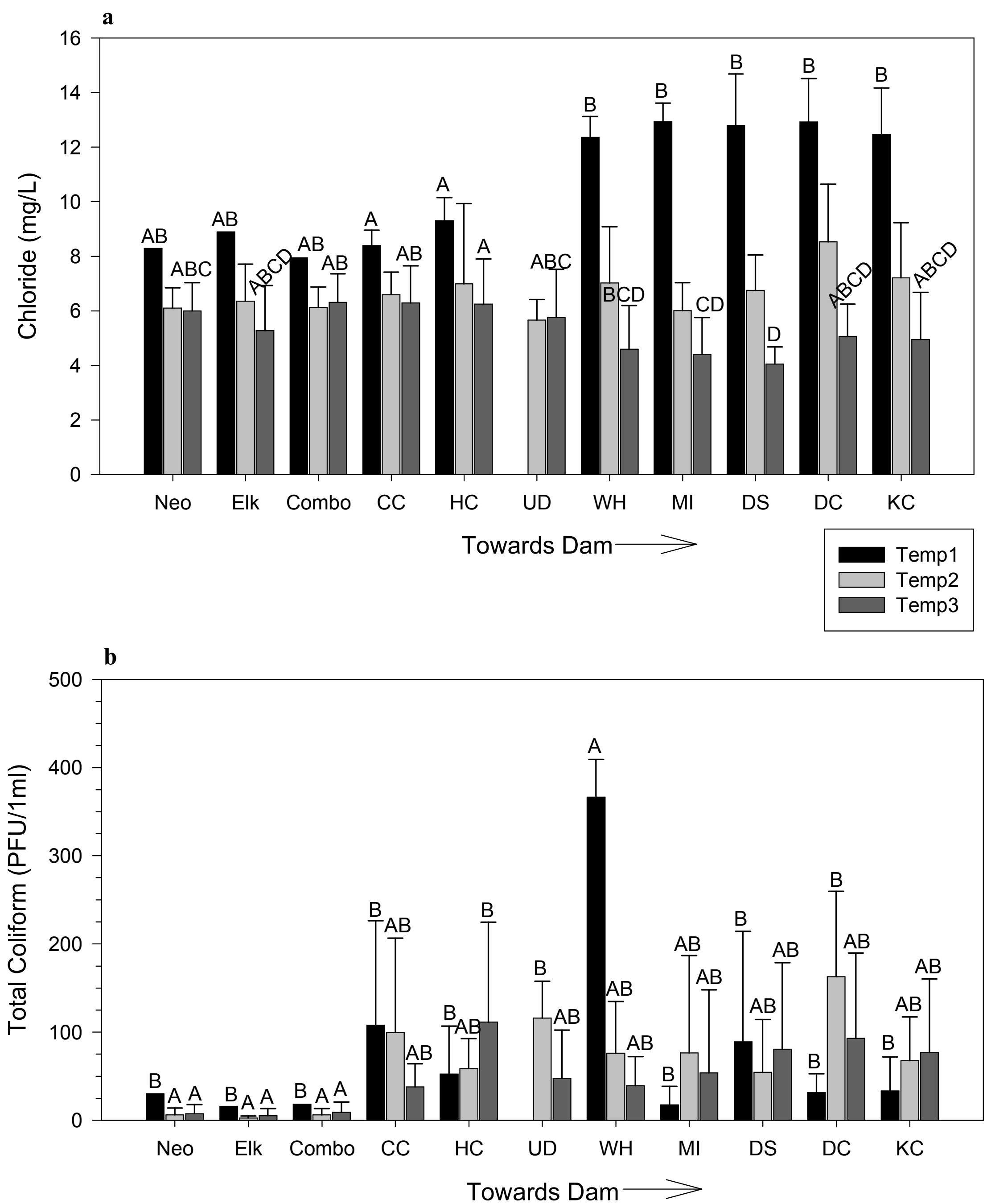


Figure 2.15. 2007 Mean values of (a) chloride and (b) total coliform concentrations in water samples from Grand Lake and river sites for different temperature classes. Bars that do not share a common letter are significantly different at α=0.05. Error bars indicate one standard deviation from mean. Neo:Neosho River; Elk:Elk River; Combo:Confluence of Elk and Neosho; CC:Cedar Cove; HC:Hickory Cove; UD:Undeveloped Site; WH:Woodward Hollow; MI:Monkey Island; DS:Dripping Springs; DC:Duck Creek; KC:Ketchum Cove.

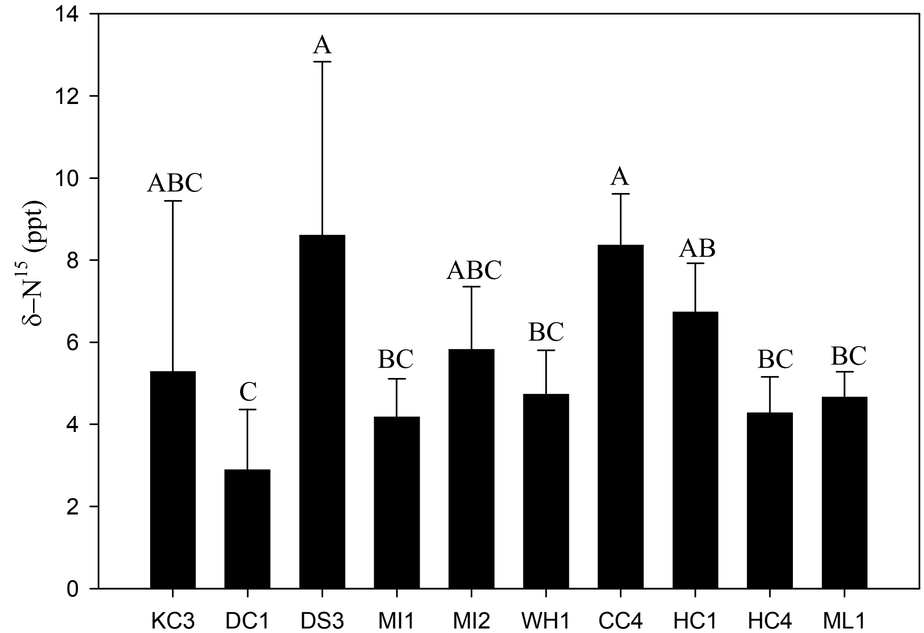


Figure 2.16. Mean δN^{15} values in periphyton samples (n=3) from Grand Lake sites in 2006. Bars that do not share a common letter are significantly different at $\alpha=0.05$. Error bars indicate one standard deviation from mean. KC:Ketchum Cove; DC:Duck Creek; DS:Dripping Springs; MI:Monkey Island; WH:Woodward Hollow; CC:Cedar Cove; HC:Hickory Cove; ML:Main Lake. Numbers designate site within each cove the sample was taken from.

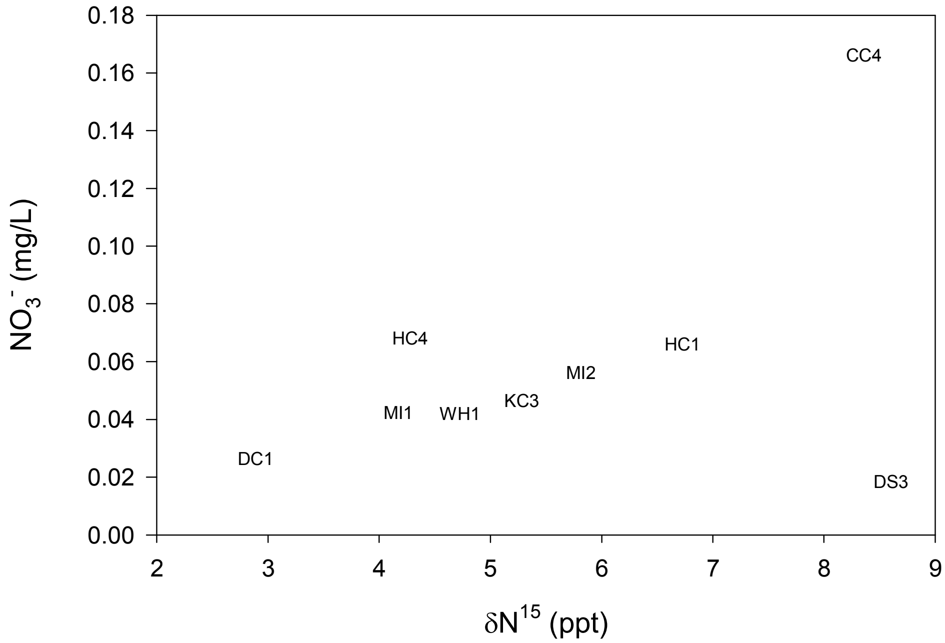


Figure 2.17. Mean nitrate vs. periphyton δN^{15} values from samples collected at the Grand Lake sites. KC:Ketchum Cove; DC:Duck Creek; DS:Dripping Springs; MI:Monkey Island; WH:Woodward Hollow; CC:Cedar Cove; HC:Hickory Cove; ML:Main Lake. Numbers designate site within each cove the sample was taken from.

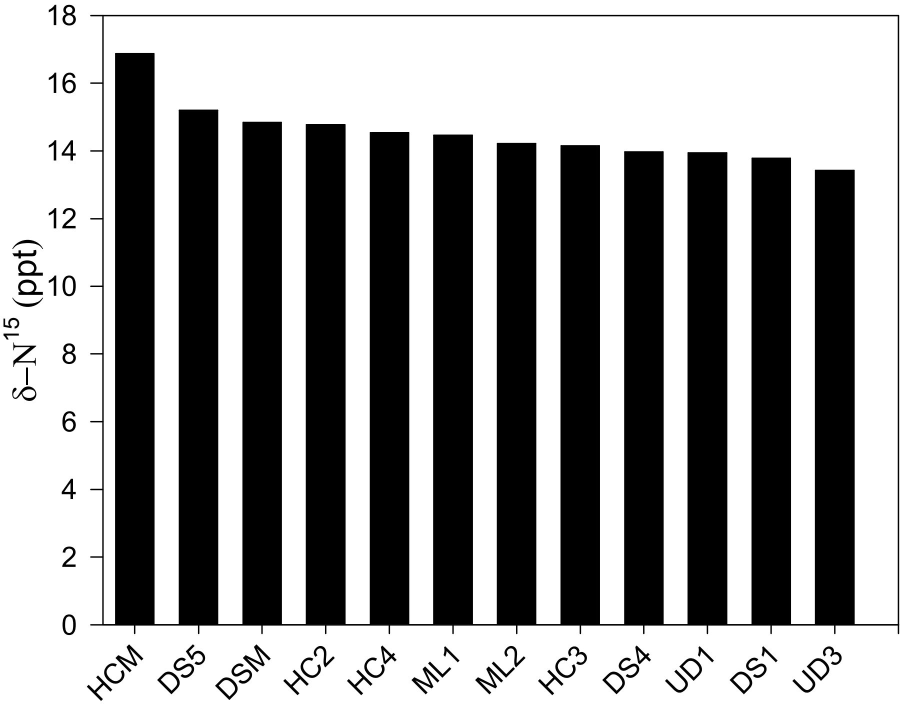


Figure 2.18. δN^{15} levels from chironomid midge samples (n=1) from Grand Lake sites. HC:Hickory Cove; DS:Dripping Springs; ML:Main Lake; UD:Undeveloped Cove. Numbers designate site within each cove sample was taken from. (M) designates sample taken from the mouth of corresponding cove.

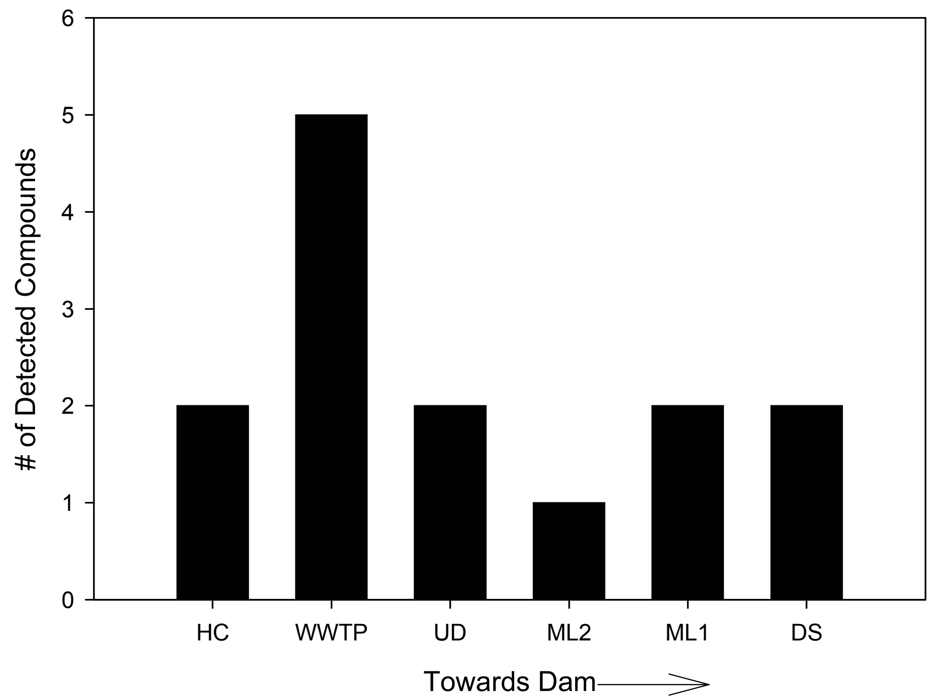


Figure 2.19. Number of organic wastewater contaminant detections at Grand Lake sites.HC:Hickory Cove; WWTP:Waste Water Treatment Plant outfall; UD:Undeveloped Cove; ML:Main Lake; DS:Dripping Springs. . Numbers designate site within each cove sample was taken from.

Table 2.1. Comparison of age/density classes and water quality parameters. Classes that do not share a common letter are significantly different at α=0.05. X= No significant differences.

	Age/Density Class Order				
Water Quality Variable	1	2	3	4	5
Conductivity	X	X	X	X	X
pH[1]	A	C	B	B	BC
Chlorophyll-a	C	C	AB	A	BC
Dissolved Oxygen[1]	A	C	B	C	BC
E. coli	C	B	A	B	C
Total Coliform	A	B	B	B	C
Fecal Coliform	C	B	A	B	C
Boron	X	X	X	X	X
Nitrate	X	X	X	X	X
Nitrite	A	C	B	B	BC
Ammonium	AB	BC	B	A	C
Total Phosphorous	A	C	B	B	C
Phosphate	AB	ABC	C	A	BC
Chloride	A	B	A	AB	B
Ideal increasing parameter			4>3>2>5>1		
Ideal decreasing parameter			1>5>2>3>4		

Table 2.2. Results of age and density linear regressions. X=no significant differences at α=0.05.

Dependent Variable	**Development Age**				**Development Density**			
	F	**p**	**slope**	**r^2**	**F**	**p**	**slope**	**r^2**
Conductivity	6.12	0.014	0.001	0.015	5.35	0.021	0.008	0.014
Dissolved Oxygen	4.39	0.037	-0.029	0.011	8.7	0.003	-0.295	0.021
Ammonium	15.46	<0.0001	0.001	0.037	x	x	x	x
Temperature		x		x	5.41	0.021	-0.433	0.013
E. coli		x		x	7.64	0.006	-5.29	0.019
Total Coliform		x		x	8	0.005	-9.57	0.019
Fecal Coliform		x		x	4.78	0.029	-8.4	0.012
Total Phosphorous		x		x	6.14	0.014	-0.01	0.014

Table 2.3. Mean values for water quality parameters measured at Grand Lake sites during 2006. Numbers in parenthesis are ranges.

	Site						
	CC		HC		WH		MI
Temperature Class	2	3	2	3	2	3	2
# Samples	5	15	4	16	5	15	6
Temperature (°C)	20.8 (20.3-21.1)	28.3 (26.67-30.7)	20.8 (20.8-20.9)	28.5 (21-31)	21.2 (21.1-21.4)	27.3 (26.2-28.9)	21.1 (20.5-21.3)
Conductivity (mS/cm)	0.30 (0.299-0.302)	0.31 (0.300-0.328)	0.30 (0.303-0.304)	0.31 (0.298-0.323)	0.29 (0.284-0.288)	0.29 (0.279-0.304)	0.29 (0.280-0.315)
pH	8.92 (8.04-9.82)	8.41 (8.17-8.71)	8.15 (7.38-9.46)	8.49 (7.39-8.83)	8.31 (6.87-11.23)	8.31 (7.78-8.69)	7.44 (6.19-8.65)
Chlorophyll-a (mg/L)	78.7 (1.7-130)	15.1 (1.7-39.6)	132.4 (75.9-243)	11.7 (2.9-67)	21.6 (5-58.9)	7.0 (3.8-11.5)	6.6 (3.5-9.2)
Dissolved Oxygen (mg/L)	8.41 (8.2-8.56)	7.15 (6.21-10.42)	5.89 (5.27-6.5)	8.22 (6.21-12.19)	8.60 (8.08-8.95)	6.97 (3.31-8.77)	6.11 (0.63-7.88)
ORP	382 (361-408)	16.1 (-29-54)	373.5 (354-396)	51.5 (-3-399)	361.8 (346-405)	39.1 (9-78)	312 (50-383)
***E. coli* (pfu/100mL)**	25 (21-32)	43 (16-69)	31 (4-67)	39 (5-96)	6 (1-16)	13 (0-59)	3.00 (0-7)
Total Coliform (pfu/1mL)	10 (8-15)	64 (12-113)	3 (0-5)	72 (17-145)	65 (6-180)	92 (10-174)	33 (4-161)
Fecal Col. (pfu/100mL)	434 (33-59)	21 (2-37)	108 (43-214)	61 (4-245)	10 (3-28)	28 (1-130)	2 (0-6)
Boron (mg/L)	0.18 (0.13-0.20)	0.05 (0.00-0.12)	0.12 (0.08-0.19)	0.09 (0.00-0.19)	0.05 (0.00-0.08)	0.04 (0.00-0.11)	0.03 (0.00-0.08)
Nitrate (mg/L)	0.14 (0.12-0.15)	0.14 (0.01-0.34)	0.04 (0.02-0.06)	0.07 (0.001-0.14)	0.01 (0.03-0.06)	0.07 (0.001-0.15)	0.05 (0.03-0.06)
Nitrite (mg/L)	0.01 (0.004-0.009)	0.01 (0.001-0.012)	0.01 (0.006-0.007)	0.01 (0.001-0.028)	0.00 (0.001-0.004)	0.00 (0.00-0.004)	0.00 (0.001-0.004)
Ammonium (mg/L)	0.05	0.07	0.04	0.03	0.03	0.05	0.02

Table 2.3 continued. Mean values for water quality parameters measured at Grand Lake sites during 2006. Numbers in parenthesis are ranges.

	Site		
	DS	**DC**	**KC**
Temperature Class	3	3	3
# Samples	25	20	20
Temperature (°C)	28.3 (27-31.1)	28.3 (26.9-30.8)	27.8 (25.2-30.6)
Conductivity (mS/cm)	0.29 (0.282-0.299)	0.29 (0.25-0.298)	0.29 (0.278-0.299)
pH	8.45 (7.93-8.7)	8.44 (7.98-8.71)	8.51 (8.24-8.74)
Chlorophyll-a (mg/L)	7.1 (2.3-17.1)	5.6 (2.1-12.7)	7.2 (2.3-18.8)
Dissolved Oxygen (mg/L)	8.17 (5.39-9.64)	7.95 (5.77-9.08)	8.18 (6.92-9.80)
ORP	31.9 (-12-81)	36.4 (-14-86)	42.3 (-6-85)
***E. coli* (pfu/100mL)**	18 (0-74)	17 (2-123)	18 (0-121)
Total Coliform (pfu/1mL)	46 (0-120)	55 (4-224)	56 (0-229)
Fecal Col.(pfu/100mL)	41 (0-173)	33 (4-180)	24 (0-71)
Boron (mg/L)	0.07 (0.00-0.145)	0.05 (0.00-0.15)	0.05 (0.00-0.1)
Nitrate (mg/L)	0.04 (0.001-0.21)	0.02 (0.001-0.06)	0.03 (0.001-0.15)
Nitrite (mg/L)	0.002 (0.001-0.003)	0.002 (0.001-0.003)	0.002 (0.001-0.004)
Ammonium (mg/L)	0.02 (0-0.06)	0.02 (0.00-0.07)	0.04 (0.00-0.23)
Total Phosphorous (mg/L)	0.01 (0.00-0.020	0.01 (0.00-0.06)	0.01 (0.001-0.07)
Orthophosphate (mg/L)	0.05 (0.001-0.3)	0.03 (0.001-0.06)	0.05 (0.00-0.18)

Table 2.4. Mean values for water quality parameters measured at Grand Lake sites during 2007. Numbers in parenthesis are ranges.

Site	Neo			Elk		
Temperature Class	1	2	3	1	2	3
Number of Samples	1	7	11	1	7	11
Temperature (°C)	18.2	25.2 (23.3-25.7)	30.2 (26.6-31.1)	17.8	25.1 (23.1-25.6)	30.4 (26.7-31.9)
Conductivity (mS/cm)	0.31	0.19 (0.074-0.226)	0.26 (0.230-0.336)	x	0.19 (0.077-0.228)	0.17 (0.163-0.180)
PH	x	8.62 (7.82-8.96)	8.84 (8.02-9.13)	x	8.63 (7.83-8.87)	8.81 (8.24-9.27)
Chlorophyll-a (mg/L)	83.1	16.7 (15.7-20.4)	8.7 (5.3-11.2)	77.6	13.6 (0-16.7)	9.7 (5.9-14.8)
Dissolved Oxygen (mg/L)	6.67	8.46 (7.12-10.99)	11.69 (6.85-13.58)	7.56	10.23 (8.14-19.18)	12.01 (7.66-16.23)
ORP	459	288.9 (135.8-348)	368 (338-404)	449	284.6 (131.7-342)	363 (329-405)
E. coli (pfu/100mL)	7	5 (0-22)	1 (0-7)	12	23 (1-129)	1 (0-1)
Total Coliform (pfu/1mL)	30	6 (0-18)	7 (0-34)	16	2.50 (0-6)	5 (0-25)
Fecal Coliform(pfu/100mL)	34	9 (0-30)	1 (0-7)	14	7 (0-29)	2 (0-18)
Boron (mg/L)	0.25	0.06 (0.01-0.1)	0.09 (0.06-0.12)	0.21	0.05 (0.02-0.06)	0.06 (0.03-0.09)
Nitrate (mg/L)	1.26	0.43 (0.2-1.04)	0.06 (0.02-0.14)	1.33	0.42 (0.2-1.04)	0.05 (0.02-0.15)
Nitrite (mg/L)	0.07	0.04 (0.031-0.044)	0.00 (0.00-0.016)	0.05	0.02 (0.018-0.025)	0.01 (0.001-0.026)
Ammonium (mg/L)	0.24	0.16 (0.03-0.275)	0.02 (0.005-0.04)	0.26	0.11 (0.00-0.233)	0.03 (0.005-0.053)
Total Phosphorous (mg/L)	0.25	0.15 (0.12-0.25)	0.26 (0.07-0.59)	0.16	0.10 (0.04-0.13)	0.18 (0.04-0.34)
Orthophosphate (mg/L)	0.38	0.30 (0.15-0.74)	0.21 (0.09-0.82)	0.31	0.20 (0.02-0.63)	0.18 (0.07-0.55)
Chloride (mg/L)	8.3	6.1 (5.7-6.1)	6.0 (4.9-7.4)	8.9	6.34 (6.0-9.4)	5.27 (3.5-7.0)

Table 2.4 continued. Mean values for water quality parameters measured at Grand Lake sites during 2007. Numbers in parenthesis are ranges.

Site	Combo			CC		
Temperature Class	1	2	3	1	2	3
Number of Samples	1	7	11	5	16	14
Temperature (°C)	17.9	25.0 (23.2-25.5)	30.3 (26.7-31.2)	17.9 (16.8-18.6)	24.4 (23.2-26.5)	29 (26.5-31.5)
Conductivity (mS/cm)	0.35	0.19 (0.074-0.233)	0.26 (0.247-0.280)	0.30 (0.284-0.307)	0.14 (0.069-0.291)	0.24 (0.001-0.297)
PH	N/D	8.50 (7.5-8.85)	8.77 (8.24-9.14)	N/D	7.98 (7.5-8.76)	8.08 (7.27-8.77)
Chlorophyll-a (mg/L)	74.5	13.2 (12.3-14.5)	12.3 (8.4-16.7)	105 (87.5-119)	12.5 (10-14.9)	18 (6.9-36.1)
Dissolved Oxygen (mg/L)	6.68	8.53 (8.04-10.27)	11.70 (7.66-14.26)	7.38 (6.28-8.57)	8.23 (6.34-10.1)	7.65 (3.52-10.39)
ORP	431	286 (150-339)	362 (328-410)	421 (418-423)	220 (107-399)	375 (328-412)
E. coli (pfu/100mL)	31	17 (6-28)	0	13 (5-23)	50 (8-385)	6 (0-34)
Total Coliform (pfu/1mL)	18	6 (0-21)	9 (0-32)	108 (19-280)	100 (8-346)	38 (4-90)
Fecal Coliform(pfu/100mL)	33	20 (5-27)	1 (0-2)	22 (15-30)	37 (2-154)	17 (0-127)
Boron (mg/L)	0.26	0.03 (0.00-0.08)	0.08 (0.05-0.15)	0.21 (0.15-0.26)	0.12 (0.05-0.235)	0.05 (0.00-0.095)
Nitrate (mg/L)	1.32	0.42 (0.18-1.05)	0.08 (0.05-0.14)	1.33 (1.28-1.4)	0.63 (0.22-0.98)	0.32 (0.05-1.11)
Nitrite (mg/L)	0.06	0.03 (0.015-0.035)	0.01 (0.003-0.017)	0.06 (0.058-0.061)	0.02 (0.014-0.037)	0.01 (0.006-0.02)
Ammonium (mg/L)	0.24	0.14 (0.011-0.212)	0.04 (0.011-0.053)	0.22 (0.169-0.297)	0.10 (0.02-0.275)	0.06 (0.01-0.127)
Total Phosphorous (mg/L)	0.22	0.19 (0.12-0.4)	0.22 (0.07-0.58)	0.22 (0.21-0.24)	0.19 (0.12-0.28)	0.29 (0.12-0.61)
Orthophosphate (mg/L)	0.37	0.28 (0.16-0.8)	0.21 (0.110.63)	0.42 (0.41-0.45)	0.47 (0.17-0.97)	0.36 (0.16-0.65)
Chloride (mg/L)	7.9	6.1 (5.6-7.7)	6.3 (5.2-8.4)	8.4 (7.7-9.2)	6.6 (5.6-8.0)	6.3 (4.3-8.5)

Table 2.4 continued. Mean values for water quality parameters measured at Grand Lake sites during 2007. Numbers in parenthesis are ranges.

Site		DS			DC	
Temperature Class	1	2	3	1	2	3
Number of Samples	8	12	13	10	6	19
Temperature (°C)	15.5 (14.1-17.7)	25.3 (23.2-26.4)	29.4 (26.6-30.8)	17.4 (15.6-18.7)	24.9 (23.8-26.3)	28.5 (26.6-30.1)
Conductivity (mS/cm)	0.33 (0.322-0.339)	0.17 (0.082-0.244)	0.18 (0.15-0.22)	0.29 (0.277-0.297)	0.10 (0.081-0.210)	0.21 (0.181-0.239)
PH	x	8.24 (8.01-8.52)	8.13 (7.71-8.78)	7.41 (6.65-8.18)	8.44 (8.02-8.96)	8.41 (7.83-9.01)
Chlorophyll-a (mg/L)	245 (75.4-429)	25.1 (0-46.3)	16.9 (5.83-56.8)	364 (28.2-429)	7.77 (7.77-7.77)	18.0 (5.7-55.5)
Dissolved Oxygen (mg/L)	11.42 (5.8-18.5)	7.46 (6.03-8.88)	7.60 (5.72-11.32)	12.34 (9.14-16.5)	9.36 (6.54-14.8)	8.57 (6.54-11.9)
ORP	447 (389-480)	286.6 (126-429)	379 (339-409)	470 (450-518)	166 (117-362)	384 (337-442)
E. coli (pfu/100mL)	81 (3-211)	11 (2-20)	11 (1-52)	48 (4-220)	31 (0-156)	23 (2-93)
Total Coliform (pfu/1mL)	89 (0-405)	55 (8-228)	80 (1-289)	31 (9-76)	163 (73-297)	93 (5-362)
Fecal Coliform(pfu/100mL)	90 (5-293)	25 (5-75)	30 (1-88)	217 (3-700)	35 (4-121)	44 (0-202)
Boron (mg/L)	0.13 (0.04-0.26)	0.13 (0.04-0.17)	0.05 (0.00-0.09)	0.16 (0.06-0.3)	0.12 (0.1-0.15)	0.07 (0.00-0.16)
Nitrate (mg/L)	0.63 (0.22-0.92)	0.65 (0.21-0.94)	0.10 (0.04-0.21)	0.35 (0.07-0.66)	0.54 (0.19-0.78)	0.13 (0.04-0.37)
Nitrite (mg/L)	0.03 (0.007-0.057)	0.01 (0.001-0.011)	0.004 (0.001-0.007)	0.02 (0.005-0.031)	0.02 (0.001-0.022)	0.01 (0.001-0.012)
Ammonium (mg/L)	0.02 (0.02-0.022)	0.03 (0.011-0.085)	0.09 (0.053-0.13)	0.03 (0.011-0.048)	0.04 (0.021-0.064)	0.04 (0.011-0.11)
Total Phosphorous (mg/L)	0.07 (0.06-0.08)	0.13 (0.05-0.25)	0.08 (0.02-0.14)	0.08 (0.05-0.13)	0.10 (0.04-0.19)	0.07 (0.005-0.15)
Orthophosphate (mg/L)	0.15 (0.02-0.23)	0.36 (0.21-0.43)	0.12 (0.07-0.22)	0.04 (0.02-0.1)	0.20 (0.12-0.26)	0.10 (0.01-0.24)
Chloride (mg/L)	12.8 (8.4-15)	6.7 (4.5-8.4)	4.0 (3.0-4.9)	12.9 (11.0-14.9)	8.5 (4.4-10)	5.1 (3.8-7.1)

Table 2.4 continued. Mean values for water quality parameters measured at Grand Lake sites during 2007. Numbers in parenthesis are ranges.

Site	WH			UD		MI		
Temperature Class	1	2	3	2	3	1	2	3
Number of Samples	5	15	15	12	18	10	9	16
Temperature (°C)	14.8 (13.9-15.6)	23.7 (22.1-25.4)	29.8 (27.1-32.2)	25.3 (23.5-26.4)	30.3 (26.5-35.2)	17.3 (16.3-20.1)	24.3 (21.9-26.3)	29.4 (26.6-31.1)
Conductivity (mS/cm)	0.31 (0.306-0.33)	0.10 (0.001-0.198)	0.20 (0.138-0.245)	0.16 (0.084-0.232)	0.18 (0.070-0.267)	0.33 (0.32-0.344)	0.13 (0.078-0.210)	0.19 (0.160-0.244)
PH	N/D	7.34 (6.87-7.62)	8.27 (7.46-8.74)	8.51 (8.14-9.15)	8.62 (7.88-9.13)	x	7.89 (7.56-8.27)	8.40 (7.93-8.81)
Chlorophyll-a (mg/L)	117 (70.8-169)	14.1 (10.2-20.1)	11.7 (6.85-20.9)	19.4 (3.86-32.1)	12.1 (5.58-55.7)	192 (11.5-430)	26.5 (21.2-33.4)	17.2 (3.88-35.8)
Dissolved Oxygen (mg/L)	10.13 (9.6-10.72)	7.12 (5.17-9.82)	8.78 (6.37-10.38)	10.75 (7.36-13.87)	10.60 (6.5-16.49)	10.76 (7.95-18.45)	7.42 (5.63-8.68)	9.26 (6.61-13.32)
ORP	436 (425-448)	251 (161-429)	378 (341-403)	288 (141-426)	302 (129-433)	448 (403-474)	253 (115-422)	380 (339-409)
E. coli (pfu/100mL)	3 (1-4)	34 (0-139)	8 (1-30)	48 (21-94)	19 (0-76)	16 (5-53)	13 (1-23)	10 (0-48)
Total Coliform (pfu/1mL)	367 (324-420)	76 (9-230)	39 (4-100)	116 (43-158)	47 (2-167)	17 (3-71)	76 (7-281)	54 (0-364)
Fecal Coliform(pfu/100mL)	6 (1-16)	57 (5-151)	65 (0-695)	36 (18-55)	19 (0-66)	18 (3-52)	13 (2-41)	53 (0-312)
Boron (mg/L)	0.14 (0.08-0.18)	0.10 (0.02-0.29)	0.09 (0.015-0.15)	0.17 (0.005-0.31)	0.05 (0.00-0.16)	0.12 (0.04-0.27)	0.14 (0.1-0.19)	0.07 (0.01-0.18)
Nitrate (mg/L)	1.34 (1.3-1.37)	0.72 (0.24-1.06)	0.12 (0.04-0.25)	0.57 (0.15-0.96)	0.21 (0.03-0.55)	1.14 (0.75-1.51)	0.67 (0.45-0.91)	0.14 (0.03-0.48)
Nitrite (mg/L)	0.03 (0.023-0.03)	0.02 (0.002-0.039)	0.005 (0.002-0.009)	0.02 (0.012-0.027)	0.02 (0.004-0.043)	0.04 (0.023-0.058)	0.02 (0.01-0.024)	0.01 (0.001-0.012)
Ammonium (mg/L)	0.14 (0.095-0.201)	0.04 (0.01-0.1)	0.05 (0.032-0.074)	0.02 (0.011-0.053)	0.05 (0.011-0.106)	0.14 (0.021-0.233)	0.03 (0.011-0.11)	0.04 (0.011-0.085)
Total Phosphorous (mg/L)	0.10 (0.08-0.12)	0.18 (0.110.25)	0.06 (0.005-0.13)	0.18 (0.12-0.25)	0.20 (0.1-0.59)	0.09 (0.07-0.11)	0.16 (0.13-0.18)	0.06 (0.005-0.15)
Orthophosphate (mg/L)	0.17 (0.14-0.21)	0.42 (0.17-0.74)	0.18 (0.12-0.23)	0.43 (0.15-0.97)	0.21 (0.15-0.28)	0.17 (0.05-0.32)	0.43 (0.32-0.55)	0.16 (0.04-0.36)
Chloride (mg/L)	12.4 (11.7-13.6)	7.0 (4.40-10.4)	4.6 (2.4-7.5)	5.7 (4.6-6.6)	5.8 (3.4-8.0)	12.9 (11.7-13.9)	6.0 (4.8-7.3)	4.4 (2.6-7.7)

Table 2.4 continued. Mean values for water quality parameters measured at Grand Lake sites during 2007. Numbers in parenthesis are ranges.

Site	KC			HC		
Temperature Class	1	2	3	1	2	3
Number of Samples	10	10	15	5	9	20
Temperature (°C)	17.1 (15.2-19.2)	25.3 (23.6-26.5)	29 (26.7-30.8)	17.7 (17.4-18.4)	25 (24-26.3)	28.9 (26.6-33.1)
Conductivity (mS/cm)	0.28 (0.275-0.284)	0.15 (0.082-0.211)	0.19 (0.159-0.244)	0.32 (0.305-0.334)	0.15 (0.087-0.235)	0.18 (0.001-0.307)
PH	7.17 (6.23-7.60)	8.23 (7.75-8.64)	8.62 (7.41-9.05)	N/D	8.56 (8.12-9.09)	8.43 (7.64-9.29)
Chlorophyll-a (mg/L)	344 (26.1-430)	10.9 (7.1-18.6)	21.3 (8.86-42.1)	158 (108-183)	15.1 (9.12-19.4)	10.4 (3.89-17.1)
Dissolved Oxygen (mg/L)	10.63 (6.22-13.99)	7.66 (4.93-10.25)	8.96 (6.28-11.81)	7.95 (5.81-9.13)	11.28 (7.91-14.63)	10.31 (5.01-16.22)
ORP	486 (457-578)	226 (92.3-360)	371 (317-430)	441 (429-451)	232 (141-346)	320 (110-424)
E. coli (pfu/100mL)	14 (5-40)	6 (1-11)	19 (0-160)	7 (1-11)	39 (4-86)	58 (1-340)
Total Coliform (pfu/1mL)	33.50 (7-138)	67.60 (21-162)	76.67 (15-350)	52.20 (14-148)	58.56 (7-111)	111.5 (1-360)
Fecal Coliform(pfu/100mL)	35 (0-163)	55 (6-153)	29 (0-92)	28 (13-47)	28 (10-53)	117 (0-700)
Boron (mg/L)	0.16 (0.07-0.27)	0.10 (0.03-0.15)	0.08 (0.015-0.22)	0.23 (0.2-0.26)	0.18 (0.09-0.31)	0.07 (0.02-0.12)
Nitrate (mg/L)	0.38 (0.1-0.69)	0.48 (0.06-0.87)	0.24 (0.04-0.55)	1.26 (1.18-1.35)	0.56 (0.18-0.91)	0.26 (0.03-1.02)
Nitrite (mg/L)	0.02 (0.003-0.048)	0.01 (0.001-0.021)	0.01 (0.005-0.014)	0.06 (0.055-0.064)	0.02 (0.012-0.019)	0.02 (0.005-0.046)
Ammonium (mg/L)	0.03 (0.021-0.032)	0.09 (0.011-0.159)	0.08 (0.032-0.21)	0.17 (0.116-0.275)	0.12 (0.021-0.254)	0.06 (0.011-0.148)
Total Phosphorous (mg/L)	0.08 (0.07-0.14)	0.15 (0.06-0.55)	0.09 (0.005-0.14)	0.24 (0.2-0.31)	0.19 (0.12-0.32)	0.21 (0.09-0.55)
Orthophosphate (mg/L)	0.08 (0.03-0.17)	0.28 (0.15-0.38)	0.16 (0.02-0.55)	0.34 (0.31-0.38)	0.40 (0.14-0.64)	0.27 (0.12-0.54)
Chloride (mg/L)	12.5 (10.5-14.3)	7.2 (4.5-10.4)	5.0 (3.3-8.7)	9.3 (8.4-10.1)	7.0 (5.8-14.8)	6.3 (3.3-8.8)

Table 2.5. Mean δN-15 values for periphyton (n=3) and chironomid midge samples (n=1) from Grand Lake. HC:Hickory Cove; DS:Dripping Springs; ML:Main Lake; UD:Undeveloped Cove. Numbers designate site within each cove sample was taken from. (M) designates sample taken from the mouth of corresponding cove

Site	Periphyton	Chironomid
DS1	x	13.79
DSM	x	14.83
DS3	8.60	x
DS4	x	13.96
DS5	x	15.19
CC4	8.36	x
HC1	6.73	x
HCM	x	16.87
HC2	x	14.77
HC3	x	14.15
HC4	4.28	14.54
DC1	2.89	
ML1	4.66	14.46
ML2	x	14.22
KC3	5.28	x
MI1	4.17	x
MI2	5.81	x
WH1	4.73	x
UD1	x	13.94
UD3	x	13.42

Table 2.6. Wastewater contaminants detected in POCIS study. HC:Hickory Cove; WWTP:Waste Water Treatment Plant outfall; UD:Undeveloped Cove; ML:Main Lake; DS:Dripping Springs. . Numbers designate site within each cove sample was taken from.

Site	Analytes Detected
DSM	Phenol
	Cholesterol
DS1	Cholesterol
DS4	Cholesterol
DS5	Cholesterol
ML1	Phenol
	Cholesterol
ML2	Cholesterol
WWTP	Tri (2-chloroethyl) phosphate
	Coprostanol
	Cholesterol
	Phenol
	Nonyl-phenol
HCM	Nonyl-phenol
	Cholesterol
HC2	Nonylphenol
	Cholesterol
HC3	Nonylphenol
	Cholesterol
HC4	none
UDM	Cholesterol
UD1	Cholesterol
UD2	Cholesterol
	Nonylphenol
UD3	None
WWTP Blank	Cholesterol
Field Blank	None

Kevin Wayne Burgess

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Master of Science

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Major Field: Environmental Science

Scope and Method of Study: The Pensacola Hydroelectric Project impounds the Neosho, Elk, and Spring Rivers in northeastern Oklahoma to form Grand Lake O' the Cherokees (Grand Lake). Grand Lake has extensive near-shore development with the majority of homes using septic systems to treat household wastewater. The soils that surround much of the lake are only marginally adequate to retain/treat septic leachate, so it is possible that effluent from some septic systems could reach the surface water of Grand Lake. This study investigated the relationship between factors that could increase the potential for septic system failure (housing density and age) and common water quality parameters that could indicate the presence of septic effluent in surface waters. A geographic information system (GIS) database was used categorize selected sites along the Grand Lake shoreline based on housing density and age. Water samples were taken at these sites through late spring, summer and early fall of 2006 and early spring, summer, and early fall of 2007, and a suite of water physical, chemical and bacterial parameters were measured. Flow regimes between the two field seasons were significantly different, with 2006 having low flow conditions and 2007 with record-breaking precipitation and flooding.

Findings and Conclusions: No clear relationship was observed between water quality and the categories used to differentiate housing age and density, although site comparisons indicated a decreasing longitudinal gradient towards the dam for parameters such as total phosphorous, orthophosphate, nitrate, nitrite and ammonium. High variability was also observed for some water quality parameters within sample coves. Sampling with Polar Organic Chemical Samplers (POCIS) indicated organic wastewater compounds at most sites sampled with the most detected at the outfall of a wastewater treatment facility. Stable isotope (δN-15) indicates some potential nitrogen enrichment of some sites, but a more definitive investigation with increased replication is needed to better evaluate this relationship. Overall, no clear links between risk factors for septic system failure and water quality indicators of this failure were indicated in this study. Methods such as microbial source tracking and groundwater investigation may be necessary to characterize the relationship of septic systems and water quality.

ADVISER'S APPROVAL Joseph R. Bidwell