LAKE SHASTINA LIMNOLOGY

Prepared for the

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Department of Environmental Science & Policy
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by
Elizabeth Vignola
Michael Deas

Watercourse Engineering, Inc.
133 D Street, Suite F
Davis, CA, 95616

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1. Introduction

Lake Shastina, also known as Dwinnell Reservoir, is located on the Shasta River at approximately River Mile (RM) 40. Impounded by Dwinnell Dam, the reservoir and associated facilities were constructed in the late 1920's as a water supply project for the Montague Water Conservation District (MWCD). Although a relatively small reservoir, with a capacity of approximately 50,000 acre-feet, the reservoir only fills in above normal runoff years due to the relatively modest yield from upstream watershed areas, seasonal water use, and appreciable seepage loss from the reservoir.

Montague Water Conservation District (MWCD) was organized May 5, 1925. There are approximately 11,000 irrigable acres in the Montague District (Booher et al. 1959). Besides the dam and the reservoir, the district owns 60 miles of canals and lateral ditches to direct water from Lake Shastina to water rights owners during the irrigation season. The main canal is approximately 35 miles long. The MWCD owns the rights to 35,000 acre-feet of water from the Shasta River and 15,000 acre-feet of water in Parks Creek plus other, incidental, water rights.

2. Previous Studies

Lake Shastina has had one detailed limnological study, completed by Dong et al (1974), wherein a comprehensive examination of the physical, chemical, and biological characteristics of the reservoir were examined using data from water year 1972 (October 1, 1971 to September 30, 1972). In addition, the California State Water Resources Control Board (SWRCB) sampled for a variety of physical and chemical parameters during the 1970s and 1980s; this data can be found in the STORET database. Subsequent efforts included a sampling study by the California Department of Water Resources (DWR) between February 1998 and April 2002, with the majority of samples being collected between November 2000 and April 2002. This effort consisted primarily of thermal and dissolved oxygen profiles as well as discrete nutrient samples and flow monitoring. The most recent work at Lake Shastina was completed by the NCRWQCB and includes September 2003, July 2004, and August 2004 sampling consisting of additional physical profiles (temperature, dissolved oxygen, pH, and specific conductance), as well as discrete samples for phytoplankton, chlorophyll a, nutrients, and biochemical oxygen demand. The NCRWQCB also completed sonde profiling of Lake Shastina during this time (NCRWQCB 2004a).

3. Geology

The Shasta River basin lies in the Shasta Valley in Northern California. The landscape is dominated by the 14,162 foot peak of Mt. Shasta. To the east lies the rugged topography of the Cascade Range, including several volcanic cones. On the west, the valley is bordered by the Klamath Mountains which prevent moist ocean air from reaching the valley. The valley is underlain with thick layers of sandstone, greywacke, shale, and basaltic lava. Additionally, on the eastern side, where Lake Shastina is located, there is alluvium composed mostly of sand, gravel and clay, most of which is water bearing (DWR, 1986). Throughout the valley are small hillocks that are deposited debris from a huge avalanche and debris flow that occurred more than 300,000 years ago (Crandall, 1989).

3.1. Land Use

Land use adjacent to Lake Shastina is dominated by the urban development associated with Lake Shastina Properties. In addition, upstream land use practices and conditions - in both the Shasta River and Parks Creek – can potentially contribute nutrients that impact the water quality of Lake Shastina.

Land use practices in the Shasta River watershed include agricultural, urban, and industrial uses. Agricultural practices include pasture and hay (and/or alfalfa) production, as well as stock water and rural urban uses. The City of Weed is the primary municipal use; however, increasing urbanization of the upper watershed has been occurring steadily over the past years. Finally, the upper portion of the watershed lies within the boundaries of the Shasta National Forest where land use practice includes timber management and harvest.

3.1.1. Shastina Properties

The community of Lake Shastina surrounds much of the lake. Shastina Properties began as a second home recreation community in 1968 and today is home to retirees and families. Currently, the Shastina Properties housing development consists of 4,052 building lots on approximately 1,800 acres. The community of Lake Shastina has paved roads and water, sewer, police and fire departments. There is also a golf resort with 27 holes. Lake Shastina's wastewater treatment facility is a secondary treatment plant that utilizes evaporation-percolation ponds as its treated water disposal site. These ponds are located downstream of Lake Shastina near Big Springs Road. Development of lands adjacent to the reservoir and in upstream areas of the watershed can impact the water quality of the reservoir through changes in land use, and point and non-point source discharges.

3.1.2. City of Weed

The city of Weed lies in the upper portion of the Shasta River basin. Weed, with a population of approximately 3,000 people, has low, medium, and high density housing, businesses located around a downtown core, a 9-hole golf course, and typical city services such as a water supply and sewer system. Wastewater from the city of Weed goes through secondary treatment at the wastewater treatment facility. Disposal of treated wastewater is via evaporation-percolation ponds adjacent to Boles Creek; monitoring in Boles Creek upstream and downstream of the ponds indicates that there is no significant leakage from the ponds into Boles Creek (pers. comm. M. St. John). The J.H. Baxter Superfund site, located at the northeastern border of Weed, is an EPA listed Superfund site. This timber milling facility also treats wood. Wood treatment, to guard against fungi, insects, and other nuisances, involves the use of arsenic, creosote, chromium, copper, zinc, and pentachlorophenol: all of which are hazardous materials. Beaughton Creek runs through the Superfund site (pers. comm. M. St. John).

3.1.3. Other Land

Other land uses in the Shasta River basin above Lake Shastina include rural residential use and agricultural lands (pasture lands) and logging and timber in the Klamath and Shasta-Trinity National Forests. Crystal Geyser taps water from the headwaters of Beaughton Creek for drinking water; there is a bottling facility nearby. In addition, there is a resort, Stewart Mineral Springs, located at the headwaters of Parks Creek.

4. Reservoir Morphology

The basin occupied by Lake Shastina is shallow near the inflow of the Shasta River and in the vicinity of Garrick Creek, broadens at mid-reservoir, and narrows to the deepest site near the dam. Contour maps of Lake Shastina are available in the Dong et al (1974) limnological survey. Additionally, transverse profiles of the lake bottom elevation presented by Dong et al (1974) provide further representation of the gradual deepening towards the dam. Lake Shastina has a full pool operating elevation of approximately 2,800 feet above mean sea level (ft msl). Table 1 provides geometric parameters and values for Lake Shastina, including area, width, and depth of the reservoir (Dong et al 1974).

The outlet works for the reservoir include an 8.5 foot diameter tower with 4 gates each with an invert elevation of 2757 ft msl (approximate capacity 700 cfs). There is an unregulated spillway tower with two openings at 2,805 ft msl, as well as an upper invert elevation of 2816 ft msl (pers comm. W. Bennett). The reservoir outlet works were originally designed for a larger reservoir and have been retrofitted to accommodate lower maximum storage.

Table 1 Geometric parameters for Lake Shastina (recreated from Dong et al (1974)).

Parameter	Value
Drainage Area	117 square miles
Surface Area	2.85 square miles
Main axis (extends from south-southwest to north-northeast)	3.80 miles
Maximum width	1.50 miles
Mean width	0.80 miles
Maximum depth	65 feet
Mean depth	22 feet
Volume	50,000 acre-feet
Length of shoreline	13.50 miles

5. Hydrology

The Dwinnell Dam was constructed to form Lake Shastina (Dwinnell Reservoir) in 1927. The reservoir was constructed with a theoretical total storage capacity of 74,000 acre-feet, but due to seepage and possible safety issues associated with the dam, Dwinnell Reservoir originally had an effective storage capacity of approximately 34,000 acre-feet. The addition of a berm at the toe of the dam in 1956 allowed for the storage limitation to increase from 34,000 acre-feet to approximately 50,000 acre-feet, where it remains today. The Montague Water Conservation District holds the rights to 35,000 acre-feet of water from the Shasta River. Additionally, they hold rights to approximately 15,000 acre-feet of water from Parks Creek that is diverted into the Shasta River above Lake Shastina for storage in the reservoir. Following the increase in allowable storage to 50,000 acre-feet in 1956, Lake Shastina did not attain full pool until February 1958 (DWR Watermaster Service Reports).

5.1. Inflow

Lake Shastina inflow is primarily derived from the Shasta River; however, inflows from Garrick Creek, other smaller intermittent streams and other surface and subsurface inflow, as well as precipitation contribute as well. Additionally, water from Parks Creek is diverted into the Shasta River above Dwinnell Dam for storage in Lake Shastina under an existing MWCD water right. The Parks Creek diversion enters the river upstream of the stream gauge at Edgewood and is thus included in the annual flow records for the Shasta River at Edgewood.

5.1.1. Shasta River

The mean annual flow for Shasta River at Edgewood was estimated from available data from USGS and the DWR Watermaster Service reports between 1937 and 1967. Although the record is discontinuous, as shown in Figure 1, examining annual average precipitation in Yreka (Figure 2) indicates that the years when flow data is available were neither particularly wet nor dry. For the years of available flow data that are presented in Figure 1 the mean annual precipitation at Yreka is 20.25 inches, which is comparable to the long term mean (1872 to 1992) of 18.0 inches. This suggests that the series of years were not particularly wet nor particularly dry. Thus the average of these years was assumed to represent average conditions. Based on USGS and the DWR Watermaster reports the mean annual flow for the Shasta River at Edgewood is approximately 60,000 acre-feet based.

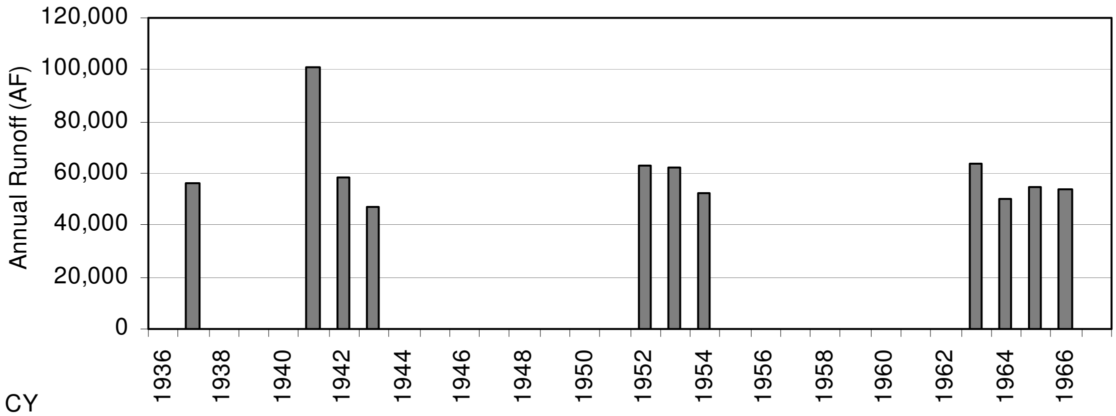

Figure 1 Mean annual flow at Edgewood (calendar year).

5.1.2. Other Surface Water

There are no formal records for Garrick Creek inflow to Lake Shastina, but Watermaster Service records detail seasonal water use in the creek. During the summer period much of this water is diverted for use in the lands adjacent to the creek. However, being a spring creek, with spring streams tending to have relatively stable year-round flows compared to streams with precipitation supported base flows, one could assume that during non-consumptive use periods (October through March) these flows reach the lake. Examination of the available records during the period 1936 and 1955 (when detailed records are available), Garrick Creek above diversions averaged 6.4 cfs, with a minimum of 5.5 cfs and maximum of 7.7 cfs for the months of April through September (DWR Watermaster records). Dong et al (1974) estimated Garrick Creek contributed 4,800 acre-feet for the 1972 water year.

There are no formal records for other minor tributaries, diffuse surface runoff (e.g., storm water inflows), or return flows from local developed areas. However, with regard to inflow, it is important to note the MWCD water rights for Parks Creek result in a significant augmentation of inflow to Lake Shastina. Although records of actual delivery are unavailable, existing water rights allow up to 15,000 acre-feet per year of water to be transferred from the Parks Creek watershed to the Shasta River watershed above Lake Shastina between the dates of November 1 and June 1.

5.1.3. Groundwater

No records of groundwater inflow to Lake Shastina are available. However, Dong et al (1974) estimated groundwater inflow as 300 acre-feet for the 1972 water year.

5.1.4. Precipitation

Based on the mean annual precipitation of 13.9 inches at Weed for the period 1992 through 2003, (see Figure 2) inflow due to precipitation can be estimated based on the surface area of Lake Shastina at full pool operations (2.85 square miles or 1,823 acres). This approximation yields 2,112 acre-feet of annual precipitation inflow. Because of annual variability in inflow and operations of Dwinnell Reservoir, this value does not represent conditions when reservoir storage is lower and/or precipitation varies from the long-term mean. Nonetheless, the value does provide a ballpark of inflow due to precipitation. This estimated direct precipitation value differs significantly from the number used in the water budget reported by Dong et al (1974). Dong et al. gathered their precipitation data from the 1971 California Region Framework Study Committee and it is unclear if this was a multi-year estimate or simply the data for a single year. The 1974 USGS report also used 2.85 square-miles as the area of Lake Shastina, but according to recent estimates (based on DeLorme 1999) the area of Lake Shastina is approximately 1.41 square miles. This significant difference in lake size highlights the need for a fully developed and updated water budget for Lake Shastina.

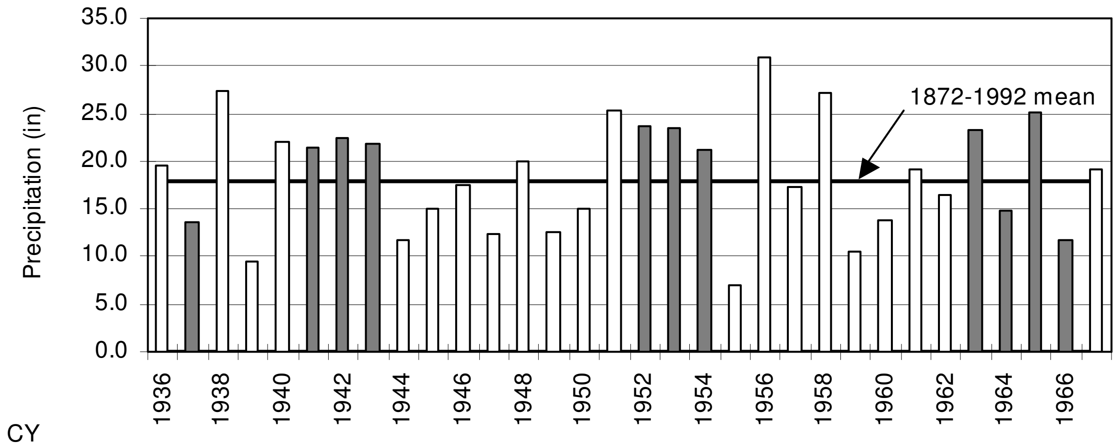

Figure 2. Mean annual precipitation at Yreka (calendar year). Gray bars represent years in which flow data is also available (see Figure 1).

5.2. Outflow

Outflow from Lake Shastina includes controlled and uncontrolled (spill) releases from Dwinnell Dam to the Shasta River, releases to the MWCD canal, evaporation, and losses to groundwater (seepage). The one reservoir outlet leads to the MWCD canal and releases to the Shasta River are made from the canal via a channel gate with a 700 cfs capacity. The reservoir outlet is an 8.5 foot pipe that is at 2757 feet above mean sea level (msl). The outlet tower spillway has two holes at 2805 feet msl as well as an emergency spillway a rim over pour (pers. comm. M. St. John). Records of uncontrolled spill volumes or rates to the Shasta River were unavailable.

Recent records of diversion to the MWCD were not reviewed for this study, but review of Watermaster service records from the 1950’s indicates that the amount of water released into the MWCD canal (including the “in-lieu of natural flows” releases) were typically in the range of 10,000 acre-feet to 17,000 acre-feet per irrigation season. The “in-lieu” releases to the Shasta River below Dwinnell Dam are required to provide water to landowners whose diversion points were inundated during the construction of Dwinnell Dam. These rights amount to up to approximately 10 cfs and are delivered on an as needed basis from the MWCD canal via a slide gate. Explicit records (e.g., daily, weekly, or monthly) of these releases are unavailable; however older water master records often report a total volume of water delivered for the season on the order of 1,500 to 3,000 acre-feet.

5.2.1. Evaporation

Evaporation values from Dwinnell Reservoir are largely based on estimates. Dong et al (1974) estimated evaporation losses at 6,000 acre-feet per year or approximately 5 to 10 percent of the annual outflow. Based on Dunne and Leopold (1978), annual average evaporation from a Class A evaporation pan is approximately 60 inches per year for the area of northern California where the Shasta Basin is located. Class A pan coefficients range 0.40 to 0.85 based on wind conditions, local vegetation size and type, and relative humidity (Shuttleworth, 1993). For locations where pan coefficients have not been determined, Dunne and Leopold (1978) recommend an average annual value of 0.70 to 0.75. Multiplying the annual evaporation pan rate by 0.7 provides an approximate annual free water surface evaporation in the Shasta Valley of approximately 42 inches per year. For a surface of approximately 1824 acres, the annual evaporation loss is roughly 6,380 acre-feet per year. This value is consistent with Dong et al (1974).

5.2.2. Losses to Groundwater

Records of groundwater losses from Lake Shastina only consist of estimates from older DWR Watermaster service records and Dong et al (1974). Estimated losses due to seepage and evaporation in the DWR Watermaster service records from the 1950's are in the range of 6,500 acre feet to as much as 42,000 acre feet and are largely a function of storage. Periods with more storage tend have larger seepage losses. Dong et al (1974) estimated groundwater seepage losses as 30,100 acre-feet for the 1972 period.

5.3. Storage

Lake Shastina storage varies in response to annual hydrology, MWCD demand, and carryover storage from previous years. Storage information is readily available from DWR and is presented for water years 1937 to 2004 in Figure 3 (CDEC and DWR Watermaster service reports). Data obtained from the Shasta Valley Watermaster Service is the reported storage in the reservoir for the first of the month. The CDEC database has average monthly storage values available. Individual missing months that were unavailable in either data set were filled by linear interpolation for March 1943 and April 1946. However, there is one gap in the data that was not approximated due to the difficulty in making an accurate estimation. There were no data points given for all of October 1981 in the Watermaster service records and the CDEC records indicated an error in the data for this month. Under some circumstances a linear interpolation between the two closest points (in this case September and November storage) would have been a sufficient estimation; however, October is often the month with the lowest storage levels of the year and linear interpolation using September and November storage values would probably not provide an accurate storage estimate. Additionally, early Watermaster reports (in the 1930s and 1940s) often only recorded storage in the reservoir during the irrigation season, April through September. While the rest of the year cannot be properly estimated, due to the large data set available, these early data gaps do not negatively affect the analysis.

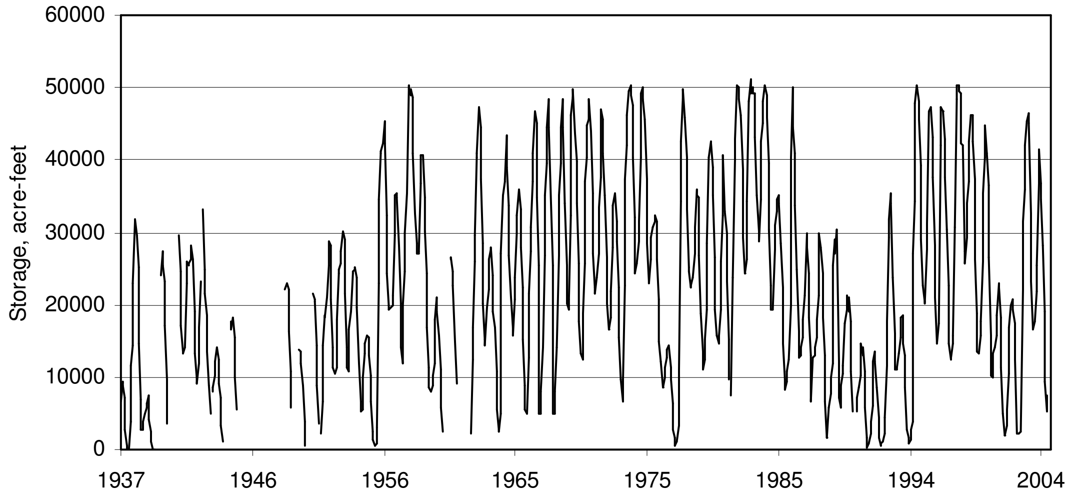

Figure 3 Monthly storage in Lake Shastina for April 1937 through October 2004. Note that the maximum storage levels in the reservoir increase in the 1950s from just over 30,000 AF to 50,000 AF with the addition of a berm to the toe of the dam.

The elevation and storage capacity curve in Figure 4 provides useful information for assessing Lake Shastina. Basin contours were taken in 1926 before the reservoir was filled. These contours, coupled with three transverses taken with sounding techniques during 1971, indicate that the minimum lake elevation (storage is equal to zero) is approximately 2,735 feet above mean sea level (ft msl), but that little volume is available below approximately 2,760 ft msl.

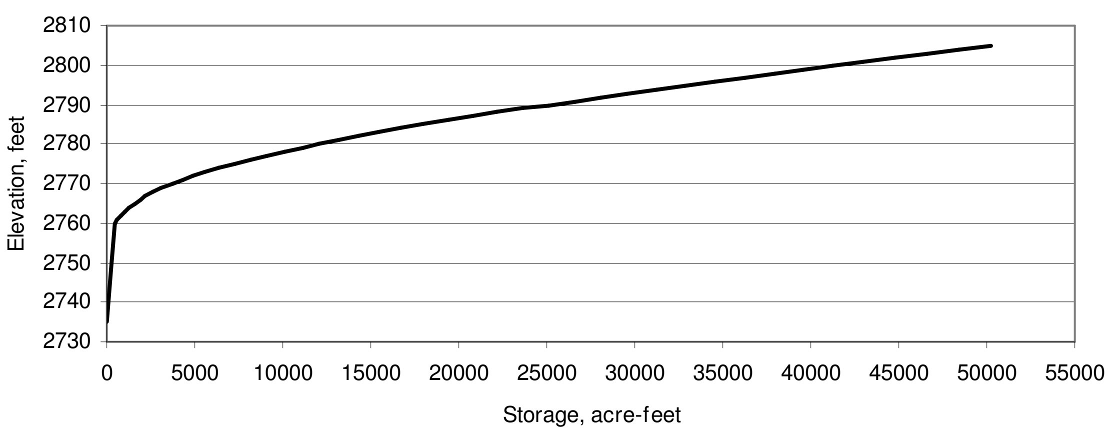

Figure 4 Storage and elevation capacity curve for Lake Shastina. Data derived from Montague Irrigation District information (1958-1993).

Storage and elevation data derived from Montague Irrigation District data during the period of 1958 to 1993 indicate that approximately 400 acre-feet is the reservoir dead storage. This is confirmed in that the minimum reported storage in Lake Shastina, according to the Shasta Valley Watermaster Service reports is 400 acre-feet.

Storage data (used to interpolate elevation data) prior to 1993 is available from Shasta Valley Watermaster Service Reports. After 1993 (namely 2000 to 2003) data was downloaded from CDEC. All CDEC data is treated as provisional.

5.4. Water Balance

Dong et al (1974) completed a water balance for the reservoir using 1972 water year data. Although only a single year was employed, the exercise is useful in identifying the potential magnitude of the various hydrologic components. A water balance is based on the conservation of mass and in its simplest form is given as

$$\Delta S = I - O$$

For Lake Shastina, detailed annual inflows and outflows can be included:

$$\Delta S = I_S + I_G + I_P - (O_S + O_E + O_L)$$

Where

- ΔS = *annual change of lake storage*
- I_S = surface water inflow (Garrick Creek, Shasta River (including Parks Creek diversion into Shasta River), runoff)
- I_G = ground water inflow
- I_P = direct precipitation inflow on lake surface
- O_S = surface outflow (Montague Main Canal, Shasta River)
- O_E = evaporation loss (outflow) from lake surface
- O_L = ground water outflow (including leakage).

The magnitudes of the various terms and the percentage of total inflow and outflow from the Dong et al (1974) report are shown in Figure 5.

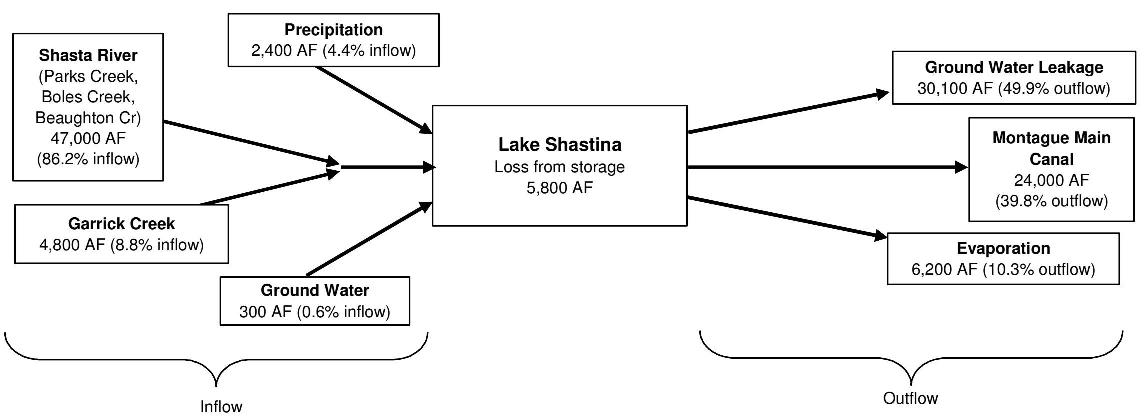

Figure 5 Diagram of the Lake Shastina water budget for the 1972 water year. Notice the percentage of inflows and outflows that each box represents. Adapted from Dong et al. (1974).

5.4.1. Summary

The Shasta River originates in the Eddy Mountains as a precipitation based stream. Its primary natural inflows are Boles Creek and Beaughton Creek, plus a diversion ditch running from Parks Creek also delivers water into the Shasta River for storage in Lake Shastina. The Shasta River is the largest inflow into the reservoir. Lake Shastina also receives water from Garrick Creek, a spring creek, from direct precipitation and groundwater.

Lake Shastina currently is designed to hold up to 50,000 acre-feet of water for storage and use by the MWCD. Since 1956, the reservoir has reached its capacity of 50,000 acre feet on approximately 10 occasions or an average of twice in every ten year period. These numbers are based on the daily mean storage in acre-feet found it the Watermaster service reports and CDEC. According to the detailed 1950s Watermaster service reports, the typical release during the irrigation season to the MWCD canal is between 10,000 and 17,000 acre-feet, which includes the in-lieu of natural flows water rights and the MWCD water rights.

The Lake Shastina surface outflow is regulated during the summer period as deliveries to meet the needs of water rights holders are released into the MWCD canal. A large component of the outflow, nearly 50% in 1972, is a result of groundwater seepage and loss: the reservoir is essentially losing water out the bottom, like a leaking sink or basin.

6. Limnological Considerations

6.1. Water Temperature

Water temperature is a critical parameter in the assessment of any lake or reservoir due to the unique thermal properties of water with respect to density. Not only does seasonal thermal loading affect the mixing characteristics of such waters, but temperatures also impact the rate of chemical reactions.

6.1.1. Thermal Properties

One of the important properties of water is its change in density in relationship to temperature. Water is the densest at about 4^{o}C (39.2^{o}F) with decreasing density as it gets either warmer or cooler while in liquid form. However, the relationship between temperature and density is not linear. Rather, the change in density per unit temperature change increases as water temperatures rise. Thus, the density difference between 10^{o}C (50^{o}F) and 11^{o}C (51.8^{o}F) is less than the density difference between 20^{o}C (68°F) and 21^{o}C (69.8°F). This concept is illustrated in Figure 6. Differences in water temperature and density have a great impact on the physical, chemical, and biological properties of bodies of water.

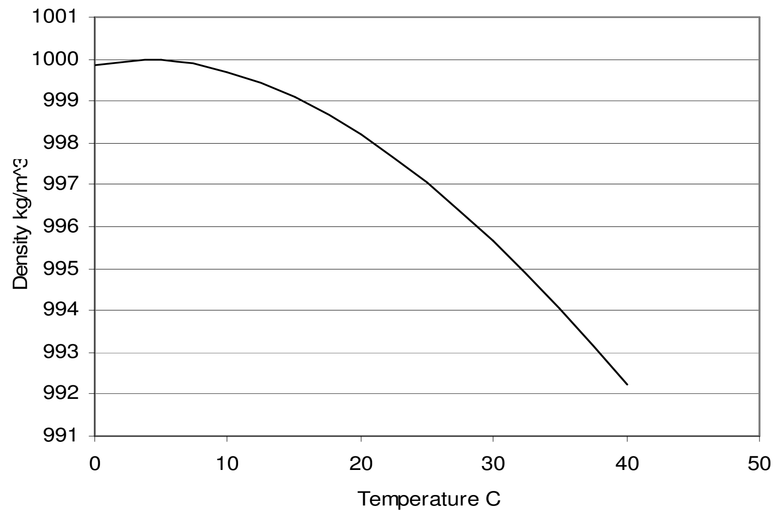

Figure 6 The relationship between temperature and density for fresh water.

Deep reservoirs and lakes in temperate climates are subject to seasonal stratification due to solar thermal loading and wind induced mixing. During summer periods, as the day length increases with increased solar altitude, thermal loading of lakes and reservoirs significantly increases the thermal energy in the surface waters. Early in the season, wind often mixes these waters throughout the reservoir, but as the day length becomes progressively longer and solar thermal loading increases, wind generated mixing cannot thoroughly mix the solar warmed surface waters throughout the entire water body. The result is thermal stratification with warm, less dense waters on the surface, and cool denser waters at the bottom. Stratification typically results in three distinct vertical zones in the water: the hypolimnion, the epilimnion, and the metalimnion.

The hypolimnion is the deep water zone, filled with cold, dense water. It is often relatively quiescent because it is separated from contact with the atmosphere. Oxygen consumption due to decomposition of organic matter can lead to depressed levels of dissolved oxygen. Long periods of stratification can lead to hypolimnetic anoxia. The epilimnion is the well-mixed surface layer that is in contact with the atmosphere and is thereby influenced by solar and wind energy. Primary production is a large component of the epilimnion because it generally corresponds with the photic zone and light availability. Between the hypolimnion and the epilimnion lies a transition layer called the metalimnion. The metalimnion is also called the thermocline because it is typified by a temperature gradient. The thermocline is an important physical factor for all biota because it impedes vertical mixing and exchange between the epilimnion and the hypolimnion, acting to restrict nutrient and gas circulation due to the kinetic energy needed to pass from less dense, warm water thru the gradient to denser, colder water.

Stratification in temperate zones is mostly seasonal. The stratification process is often reversed in the fall when cooler atmospheric temperatures reduce the surface temperatures of the reservoir. Differences in hypolimnion and epilimnion water temperatures and associated densities diminish allowing wind produced currents (as well as energy associated with inflows or outflows) to mix the whole reservoir in a process known as turnover.

Turnover has several important impacts on reservoir and lake systems. Top to bottom mixing in the fall redistributes the biotic and chemical components. This includes the reoxygenation of anoxic or depressed hypolimnetic waters, nutrient redistribution, and thermal homogeny.

In addition to the traditional stratification schemes, there may also be secondary thermoclines in the epilimnion and/or metalimnion. These may not be stable seasonal thermoclines, but rather are intermittent and may be present for relatively short periods. These secondary thermoclines may be created during warm, calm periods when the surface waters are not thoroughly mixed with depth. Secondary thermoclines are not stable and they often disappear during the following wind event.

6.2. Dissolved Oxygen

The distribution of oxygen in reservoirs is closely related to photosynthesis and respiration, thermal stratification, and organic matter. Oxygen is important in that its presence or absence influences not only the ability of organisms to occupy a particular zone of the reservoir, but also its bearing on the type of chemical reactions that can occur with regard to nutrient cycling. Plants and animals consume oxygen during respiration but, when the appropriate amount of light and nutrients are available, plants can produce oxygen through photosynthesis. Another main source of oxygen in the reservoir is diffusion from the atmosphere (reaeration). Oxygen in water is typically measured as dissolved oxygen (DO) and the loss of DO due to biological activity in the breakdown of organic matter is called the biological oxygen demand (BOD); loss due to chemical activity is the chemical oxygen demand (COD). A dynamic equilibrium exists between oxygen gas in the atmosphere and DO levels in aquatic systems that is a function of atmospheric pressure, temperature, and oxygen levels in water. This theoretical equilibrium is termed saturation DO. Temperature and saturation oxygen concentration have an inverse relationship: cold water has a higher saturation DO than warm water. Oxygen saturation levels are often useful to consider because oversaturation may be an indicator of high levels of photosynthesis while undersaturation can indicate an increase in organic matter and/or pollutants or phytoplankton respiration.

Both seasonal and diel variations in oxygen can affect the components of a reservoir system. Seasonally, oxygen may remain at or close to equilibrium levels throughout the depth of the reservoir under isothermal (un-stratified) conditions. Once thermal stratification occurs, the oxygen levels decrease, especially in the aphotic zones where significant respiration takes place. Anoxia can occur in the hypolimnion. At the same time that the hypolimnion is undersaturated, the epilimnion may be supersaturated due to intense photosynthesis. The reservoir will not reach equilibrium across the oxygen gradient due to the thermocline barrier. On a daily basis, oxygen in the photic zone may fall below saturation levels in the early morning, increase as the light and therefore photosynthesis increases until supersaturation is reached late in the afternoon, only to be depleted again once nighttime respiration begins.

6.2.1. Impacts of Hypolimnetic Anoxia on Eutrophication: Sediment Nutrient Release

Hypolimnetic anoxia leads to reducing conditions near the bed and can lead to the release of nutrients (e.g., orthophosphate, PO_4^{-3}, and ammonia, NH_4^+) from deposited sediments to overlaying water. This phenomenon, known as internal nutrient loading, exacerbates eutrophication because it acts as a positive feedback on itself – increased internal nutrient loading leads to greater algal productivity, which in turn leads to more severe hypolimnetic anoxia, which in turn leads to an increase in internal nutrient loading.

In addition to the release of nutrients, anoxic sediments tend to release other problematic reduced compounds including iron, manganese (both produce what is called "yellow water"), and sulfide. Anoxic sediments rich in organic matter can release gaseous hydrogen sulfide (H_2S) – noticeable by the familiar rotten-egg smell. In lakes this H_2S is oxidized to SO_4^{2-} at the sediment-water interface so long as there is an oxygenated condition at the sediment-water interface. Once the sediment-water interface becomes anoxic (e.g., through bacterial respiration and chemical oxygen demand), H_2S is released into the overlying water (Horne and Goldman, 1994).

6.3. pH, Alkalinity and Electrical Conductivity

The pH of a water body describes how acidic or alkaline the water is based on an exponential scale of 1 to 14. A pH reading of less than 7 indicates acidity, greater than 7 indicates alkalinity, and a pH of 7 is considered neutral. Most lakes and reservoirs have a pH between 6 and 9. Changes in the pH of a lake may influence other nutrients and minerals including iron, ammonia, phosphate and carbon dioxide. pH may vary with depth in a reservoir, especially during summer months when phytoplankton abundance and production may lead to higher pH values in areas of photosynthesis (e.g., the epilimnion), and lower in areas where respiration or decay of organic matter is occurring (e.g., the hypolimnion). pH may be affected by photosynthesis and respiration because pH is closely related to the amount of and form of carbon in the water column.

During growth (most prominent during photosynthesis) algae require carbon for cell growth. Although carbon may be present in the water column, during periods of peak growth, algae may deplete readily available forms of carbon in weakly buffered systems. When dissolved forms are depleted, carbon will enter the water column via the air-water interface as carbon dioxide: $CO_{2(g)} \rightarrow CO_{2(aq)}$. However, this process is often insufficient to keep up with algal demands. Under such conditions certain algae species are able to utilize (remove) CO_2 from bicarbonate ion: $HCO_3^- \rightarrow CO_2 + OH^-$. The result is an increase in hydroxyl (OH^-) concentration and an associated increase in pH. It is not uncommon to see diurnal variation in pH ranging from 0.5 to 1.5 pH units as a result of algal productivity

The increase in pH, if accompanied by elevated water temperature can cause a dramatic shift in unionized ammonia concentrations in aquatic systems. Unionized ammonia (NH_3 vs. NH_4^+) is toxic to fish in small quantities and lethal exposure periods are on the order of hours. Ammonia, and other nitrogen species, will be further discussed in later sections of this report.

Alkalinity is intimately related to pH in that it describes how many hydrogen ions the water can hold (preventing acidification): it is the buffering capacity of the lake, reducing changes in pH as photosynthesis and respiration occur. Carbon dioxide dissolves in water and forms carbonic acid, which then forms bicarbonate and then carbonate. These reactions are reversible and the equilibrium between the three types of inorganic carbon is associated with changes in pH. The reservoir of carbonate and bicarbonate in an aquatic system acts as buffer against changes in pH. Alkalinity values less than 100 mg/L indicate a weakly buffered system and the pH of the lake may follow a diurnal cycle during the growth season when photosynthetic activity is abundant. Alkalinity values greater than 200 are found in strongly buffered systems where the pH is not affected by photosynthetic productivity. Values between 100 and 200 are in a transitional zone and some movement of the pH value may occur.

EC is a measure of a body of water's ability to conduct electricity, and therefore a measure of the water's ionic activity and content. Higher concentrations of ionic (dissolved) constituents in a lake or stream lead to a higher measured conductivity of that lake or stream. Conductivity of the same water changes substantially as its temperature changes. Electrical conductivity (EC) normalized to a temperature of 25 ºC is specific conductance (SC), allowing direct comparison of various waters.

6.4. Nutrients

The following sections include a brief description of essential nutrients and their roles in temperate lakes and reservoirs. For a complete discussion of general nutrient roles in lakes and reservoirs please refer to Wetzel 2001.

6.4.1. Nitrogen

Nitrogen enters reservoirs as nitrate in rainfall and atmospheric deposition in the watershed, moving freely through soils and streams. In addition to that from inflowing water sources, decomposing bacteria break down organic matter and release ammonia (Figure 7).

Nitrogen in aquatic systems may exist in several forms: dissolved nitrogen gas (N_2), organic nitrogen incorporated into organic matter, ionized (NH_4^+) and undissociated ammonia (NH_4OH+NH_3), nitrite ion (NO_2^-), and nitrate ion (NO_3^-). The prevalence of ionized versus undissociated ammonia is primarily pH-dependent, such that the fraction as unionized species (NH_4OH + NH_3) increases significantly above pH=8. As noted, previously, unionized ammonia is toxic to aquatic life (with the concentration limit dependent upon the organism and life stage). Under aerobic conditions, bacteria mediate the oxidation of ammonia to nitrate (with an intermediate step as nitrite) in a process called nitrification. In the nitrification reaction, 1.0 g of ammonia consumes 4.57 g of oxygen (Chapra 1997). In the absence of oxygen, bacteria mediate the reduction of nitrate to nitrogen gas.

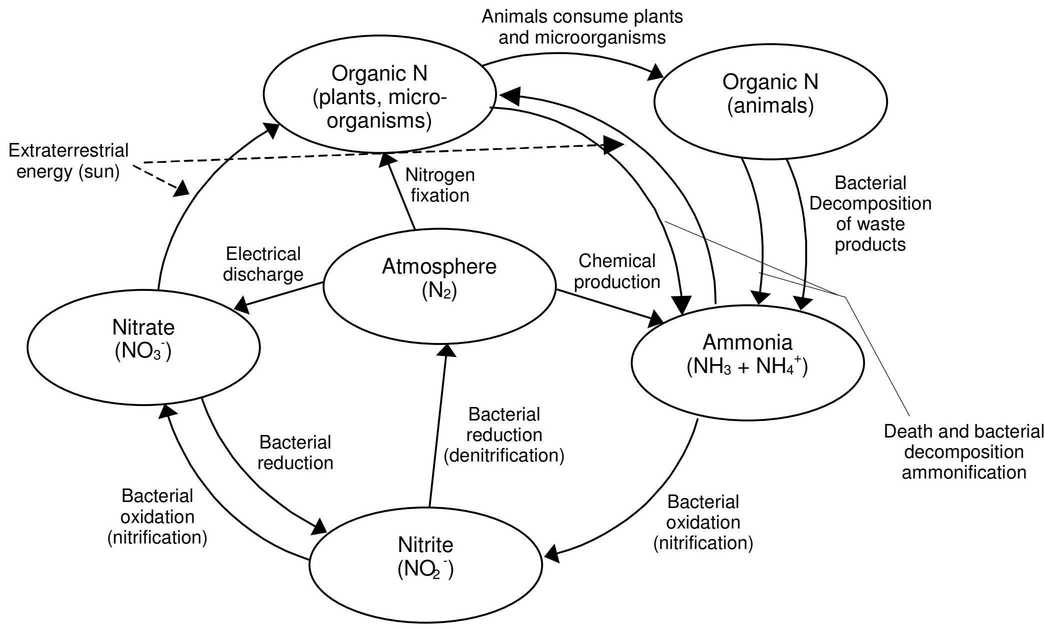

Figure 7 Nitrogen cycle

6.4.2. Phosphorus

Phosphorus is a common growth limiting nutrient in phytoplankton even though the quantity needed by plants is much smaller than other nutrients, such as carbon or nitrogen. Its critical role as a growth limiting nutrient is partially due to the fact that the only biologically available form of phosphorus is soluble phosphate (PO_4). Most phosphorus is unavailable to biota because it tends to strongly adsorb to particles in the water, exceptions include anoxic conditions. Phosphate may accumulate over the winter when primary production is at a minimum, but by late spring the levels of available phosphate may be reduced to less than 2.0 µg/L through algal uptake. This lack of available phosphate results in a system where phosphate is continuously recycled back to phytoplankton after excretion from fish, zooplankton, or bacterial activity. Internal loading from the sediments and lake mixing are important methods of returning phosphates that have left the photic zone and return them to the epilimnion for use. External sources of phosphorus include domestic and agricultural wastes that enter the water body through runoff.

6.4.3. Other Nutrients

Silicon, calcium, magnesium, potassium, iron, and other minor metals are also essential for plant and animal health and growth in aquatic ecosystems. Diatoms, a type of algae, require silica (SiO_2) in large quantities to form rigid cell walls. Because the cycle of silica in aquatic systems is predominately unidirectional, unlike nitrogen and phosphorus which can be recycled, silica is often the limiting nutrient in diatom growth and abundance (biomass). A shortage of silica combined with an abundance of other nutrients, may lead to a change in the dominant algal species in a lake or reservoir from diatoms to blue-green algae (Horne and Goldman, 1994). Other nutrients, such as calcium, magnesium, potassium, and iron, are essential in metabolism, composition and structure, and other cellular components. Please refer to Horne and Goldman (1994) for a complete discussion.

6.5. Algae

Phytoplankton are microscopic, free floating plants that commonly inhabit reservoirs as well as other water bodies. These photosynthetic plants are the primary producers of aquatic habitats. There are thousands of species of phytoplankton that are placed into general classifications by their mobility, noxious emissions, growth rates, seasonal cycles, and spatial variation in the water column. Common temperate lake algae include Chlorophyta (greens), Bacillariophyceae (diatoms), Cryptomonads (which do not have a common name), and Cyanophyta (blue-greens).

Green algae are a diverse group that, like higher plants, uses chlorophyll *a* and *b* to produce food for themselves via photosynthesis. They are found in a wide variety of aquatic environments, including those bodies of water that are nutrient rich. They may be single celled or colonial and some may have flagella--long hair like structures that allow for movement in the water.

Diatoms are possibly the most widespread algae in the world. They typically bloom first in the spring, quickly depleting the nutrients available. Diatoms are often limited by silica (SiO_2) that forms their cell walls; however, silica walls require less energy expenditure than cellulose walls so they often have a growth rate advantage over other types of phytoplankton when an abundance of silica is present. Generally, diatoms are non-toxic and do not form colonies. They are typically associated with undisturbed, aesthetically pleasing bodies of water.

Cryptomonads are single-celled algae that typically contain chlorophyll *a* and other pigments that are utilized in photosynthesis. Their cells are extremely delicate and sensitive to algal preservation techniques which results in their being underrepresented in collected samples. They are common and typically quite diverse in temperate regions although they have been reported in nearly every type of water body throughout the world.

Blue-green algae are not truly algae at all. Instead they are actually photosynthesizing bacteria known as cyanobacteria. Blue-greens are commonly found during the summer and fall in warm, eutrophic waters. During the winter they survive in a spore like resting stage in order to bloom and grow again once conditions are more favorable. These algae are successful in that they can regulate their position in the water column by means of either gas vacuoles or by producing dense carbohydrates that act as a ballast to cause sinking. Some blue green algae can also fix dissolved atmospheric nitrogen gas. In addition, cyanobacteria are of particular concern because they emit noxious chemicals that discourage grazing by zooplankton and their blooms can be detrimental in recreational waters. For further descriptions of phytoplankton please refer to Wehr and Sheath (2003).

6.6. Trophic Status

Lakes and reservoirs can be categorized into a trophic status or level based on their productivity, the amount of nutrients present, clarity, and availability of oxygen.

Eutrophic lakes are high in both winter nutrient levels and in primary productivity. Eutrophic lakes often reside in large, especially fertile watersheds (due either to natural or anthropogenic causes) and therefore are subject to an abundance of external nutrient loading. Oxygen concentrations are variable, potentially ranging from supersaturation in surface waters to depression and anoxia in bottom waters. Eutrophic lakes are further characterized by algal blooms, particularly blue-green algae, and have very low water transparency.

Oligotrophic lakes occupy the opposite end of the spectrum from eutrophic lakes and are generally defined as unproductive lakes. They tend to reside in small, rocky watersheds with infertile soils. Nutrient levels are low as is primary productivity. These conditions result in transparent waters with deep light penetration. Oxygen levels do not vary significantly from saturation in the epilimnion or the hypolimnion.

Mesotrophic lakes are intermediate between oligotrophic and eutrophic lakes, exhibiting features that are in transition from oligotrophic to eutrophic.

Eutrophication is the process of an aquatic system becoming increasingly productive. Cultural eutrophication occurs when human activities cause a rapid increase in the nutrient and/or sediment loading, resulting in increased primary productivity and algal growth. Examples of causes of cultural eutrophication include urban storm runoff, municipal sewage, and intensive agriculture. Eutrophication can sometimes be reversed, although restoration programs are expensive and slow. Trophic status and measures to assess level and rate of eutrophication include, but are not limited to, nutrient concentrations, light penetration measures, algae biomass estimates, and presence/absence of anoxia. In Table 2 the ranges for each trophic state for data are readily determined from typical field monitoring programs. Winter nutrient levels are used to identify the available nutrients when primary production is largely absent.

Table 2 Trophic status parameters

Parameter	Units	Trophic Status		
		Oligotrophic	Mesotrophic	Eutrophic
Nitrate – winter average	mg-N/L	<0.05	0.05-0.11	>0.11
Orthophosphate – winter average	mg-P/L	<0.02	0.02-0.042	>0.042
Total P – winter average	mg/L	<0.02	0.02-0.042	>0.042
Secchi Depth (growth season minimum)	feet	>26	8-26	<8
Chlorophyll *a* (annual average)	µg/L	<1	1-3	>3
Chlorophyll *a* (peak)	µg/L	<2	2-9	>9
Anoxia Presence	Yes/No	Typically not present	Typical in stratified systems	Almost always in stratified systems–
Anoxia Duration	Days per year	Typically not present	–Variable, months	–Variable, months

7. Lake Shastina

The limnology of Lake Shastina is directly related to inflow and outflow quantity, reservoir storage, and processes within the reservoir. Because Lake Shastina is relatively small, fills infrequently, and is subject to considerable drawdown, inflow quantity and quality have a direct bearing on reservoir conditions. Thus prior to presenting the reservoir limnology a description of inflows to the lake are presented. This section concludes with a brief description of Lake Shastina outflow characteristics.

The nutrient values reported in the following sections reflect a compilation of several small data sets that were provided by the NCRWQCB. These data range in time from the early 1970s to 2004. The data included in this time frame has been collected by various agencies including the Department of Water Resources (DWR) and the State Water Resources Control Board (SWRCB). Because the data has been collected by different agencies, and various laboratories have performed the chemical analyses, there is a significant amount of variation in how data is reported. It therefore has been assumed that all nutrient analyses have been reported in their elemental forms (e.g. Nitrate-Nitrogen and Phosphate-Phosphorus). An examination of the data under this assumption does not produce any gross deviations, so it therefore appears to be a reasonable assumption.

8. Reservoir Inflows

Inflows into Lake Shastina consist of several sources that contribute to the physical and biological makeup of the reservoir. The Shasta River originates in the Scott Mountains, where rainfall and snowmelt provide the bulk of the runoff. These waters commingle with Beaughton and Boles creeks, which both originate as springs on the western flank of Mt. Shasta. During winter and spring months, waters from the Parks Creek watershed, which also originates in the Scott Mountains, are diverted into the Shasta River upstream of Edgewood. These sources combine to form the inflow into Lake Shastina. Garrick

Creek (which in some references is referred to as Carrick Creek) is a spring creek on the western flank of Mt. Shasta, which flows directly into Lake Shastina. The chemical and physical composition of these varied inflows are detailed below.

Springs in the Shasta Valley are typically nutrient rich, probably as a result of the volcanic formations from which they emanate. The levels of nitrogen and phosphorus species found in several springs in the Shasta Valley are outlined in
Table 3. Although there are only a few dates and samples, spring water quality parameters are often more uniform than creek and stream waters, as the springs are not generally subject to land use patterns and other anthropogenic impacts. The values presented indicate that the springs in the Shasta Valley, at the source, are typically nutrient rich and that the Shasta River has a high baseline of nutrients that can support a productive environment.

Table 3 Water quality conditions of springs in the Shasta Valley (NCRWQCB 2004)

Location	Date	pH	Specific Conductance	NH4-N	NO2+NO3 as N	Ortho-P as P	Total P
			umhos/cm		mg/L		
Bassey Spring at Martin Ranch	08/21/2003	6.8	327	0.088	0.29	-	-
	09/17/2003	7.1	303	0.071	0.24	0.066	0.46
Evans Spring at Martin Ranch	08/21/2003	7.1	101	ND*	0.21	-	ND*
Jims Spring at Martin Ranch	08/21/2003	7.3	91	ND*	0.26	-	0.11
Spring at Hidden Valley Ranch	06/18/2003	7.3	290	ND*	0.22	0.12	0.088
	07/23/2003	-	-	ND*	0.18	0.098	0.20
Big Springs Spring	09/16/2003	7.0	381	ND*	0.14	0.16	0.22

* ND=Non detect

When natural waters are sampled during the winter months, during which algae and plant growth does not occur, a background level of nutrients can be ascertained. Summer or growth season samples may show fewer nutrients due to algal uptake and use and if not taken in consideration with other parameters, such as winter nutrient levels, may give a false impression of the aquatic system productivity.

Boles Creek and Beaughton Creek have been developed for water resources use (as per Watermaster records) and have elevated levels of nutrients although it is unclear from past sampling records if this is due to natural conditions versus anthropogenic sources. A comparison of nutrient levels directly from the spring and above the confluence with the Shasta River may help to shed more light on this subject.

Boles Creek data is limited to two sampling dates in 1971-1972, a series of sampling in 1997and 1998, and in 2003, but what is available emphasizes that Boles Creek also experiences elevated levels of nitrogen and phosphorus. Figure 8 and Figure 9 are graphs depicting growth season (mid April through October) and non-growth season (November through mid April) nutrient levels. These graphs are a combined look at all of the data for the period of record and are shown by Julian Day. Ammonia-nitrogen was “Not Detected” (ND) in all but one sample, on August 25, 1971 a reading of 0.11 NH_3-N mg/L was observed at Boles Creek. As mentioned earlier in this document in regard to land use, Boles Creek runs past the city of Weed wastewater evaporation-percolation ponds (pers. comm. M. St. John).

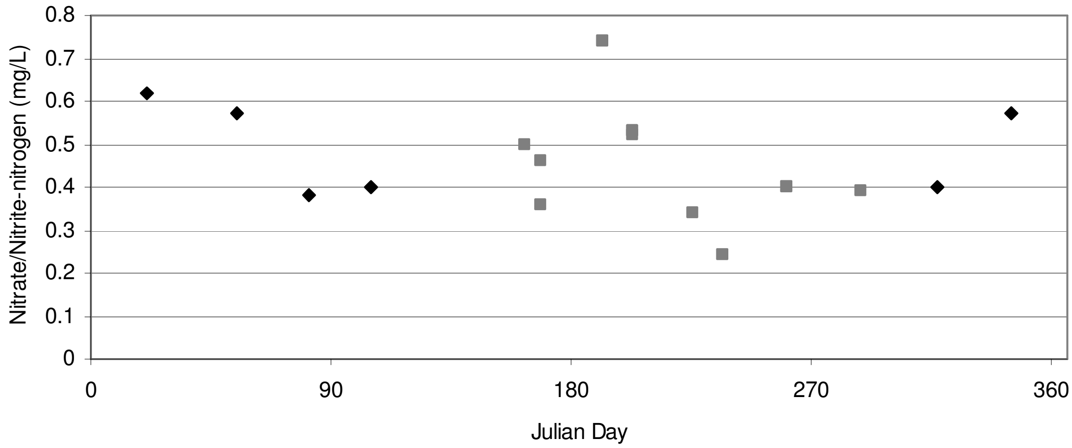

Figure 8. Boles Creek nitrate/nitrite-nitrogen data for non-growth season (black diamonds) and growth season (gray squares) . Data from DWR, NCRWCQB, and SWRCB between 1971 and 2003.

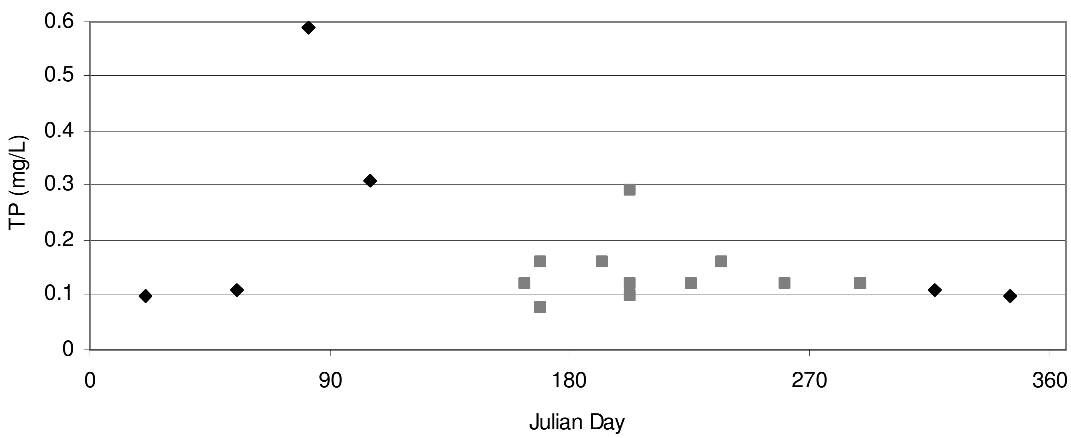

Figure 9. Boles Creek total phosphorus levels for non-growth season (black diamonds) and growth season (gray squares). Data from DWR, NCRWCQB, and SWRCB between 1971 and 2003.

Winter levels of phosphorus and nitrogen for Beaughton Creek near Edgewood are presented below in Table 4 and Table 5. The Beaughton Creek numbers indicate that the creek is potentially productive and probably contributes nutrients to the Shasta River and, and subsequently to Lake Shastina, particularly during late fall through early spring.

The Shasta River at Edgewood had some of the lowest nutrient readings upstream of Lake Shastina (Table 4 and Table 5). This is probably due to waters of lower nutrient content originating in the Shasta River and Parks Creek headwaters, providing some level of dilution. During other periods of the year, particularly during late spring and summer, nutrient values in the Shasta River at Edgewood are probably lower due algal growth and sequestering of nutrients in algal cells (biomass).

Garrick Creek, a spring creek, flows directly into Lake Shastina. Garrick Creek also shows some of the highest levels of nutrients entering the system. The Garrick Creek nutrient data came primarily from data taken in November 1996 and April 1998 (DWR data). It should be noted that Garrick Creek dried up at the second sampling station (above the reservoir) during the summer period of June 1997 thru September 1997 (n=2 for all summer readings at Garrick Creek above the reservoir station).

Table 4. Non-growth season (November thru mid-April) parameter averages of inflows into Lake Shastina

Inflow Location		pH	Alkalinity (mg/L)	Specific conductance (umhos/cm)	Nitrate/Nitrite as N (mg/L)	Ammonia as N (mg/L)	Total Phosphorus as P (mg/L)	Ortho-phosphate as P (mg/L)	n*
Boles Creek	Near Weed	8.0	73	166	0.45	ND	0.22	0.07	6
Beaughton Creek	Near Weed	7.6	54	110	0.09	ND	0.18	0.14	8
	Near Edgewood	7.6	56	112	0.35	ND	0.09	0.08	8
Shasta River	At Edgewood	8.1	101	200	0.12	ND	0.09	0.05	16
Garrick Creek	Near Weed	8.1	138	301	0.42	ND	0.17	0.12	15
	Above Reservoir	8.2	215	473	0.22	ND	0.23	0.14	13

Beaughton Creek was sampled by the NCRWQCB in 2003 and 2004 at the source (i.e., above potential inputs from adjacent land uses)

*= n varies due to the variety of data sources and grab samples available for each site during each time period. Data from DWR, NCRWCQB, and SWRCB between 1971 and 2003.

Table 5 Growth season (mid-April thru October) averages of inflows into Lake Shastina.

Inflow Location		pH	Alkalinity (mg/L)	Specific conductance (umhos/cm)	Nitrate/Nitrite as N (mg/L)	Ammonia as N (mg/L)	Total Phosphorus as P (mg/L)	Ortho-phosphate as P (mg/L)	n*
Boles Creek	Near Weed	8.1	79	171	0.40	ND	0.15	0.10	12
Beaughton Creek	Near Weed	7.8	54	103	0.06	ND	0.23	0.14	8
	Near Edgewood	7.4	60	110	0.06	ND	0.18	0.12	8
Shasta River	At Edgewood	8.2	181	223	<0.05	ND	0.13	<0.05	16
Garrick Creek	Near Weed	8.2	306	139	0.19	0.07	0.31	0.24	13
	Above Reservoir	8.1	252	539	0.04	ND	0.18	0.10	2

Beaughton Creek was sampled by the NCRWQCB in 2003 and 2004 at the source (i.e., above potential inputs from adjacent land uses)

*= n varies due to the variety of data sources and grab samples available for each site during each time period Data from DWR, NCRWCQB, and SWRCB between 1971 and 2003.

Only four sampling dates on Parks Creek above the diversion were presented in the data sets. These samples were taken between June 19, 2003 and October 21, 2003 above Stewart Springs on Parks Creek (NCRWQCB 2004a). The average pH was at 8.4 (n=4); the average specific conductivity was 200 umhos/cm (n=4); the average nitrate/nitrite was 0.05 mg-N/L (n=3); and the average total phosphorus reading was 0.07 mg-P/L (n=3). It is recommended that Parks Creek above the diversion to the Shasta River be monitored for water quality parameters more frequently.

The NCRWQCB also sampled in the Shasta River south of Interstate 5 during the summer of 2003. These few grab samples (3 samples at two sites) revealed low nutrient levels, the majority of which were non-detect with two samples that had nitrogen species levels just above the detection limit (NCRWQCB, 2004).

8.1. Limnology of Lake Shastina

Analysis of Lake Shastina is more complex than analysis of the river due to the longitudinal, transverse, and vertical variability; increased residence time; operations; and overall volume. Characterizing this variability is further challenging because relatively limited sampling has been completed within the reservoir and most of the work has been done in the vicinity of the dam. Dong et al (1974) sampled at three locations, and in September 2003, the NCRWQCB sampled at two different stations in the reservoir. Data from these results indicate that longitudinal and vertical variability are present in the reservoir.

8.1.1. Temperature

Lake Shastina exhibits seasonal thermal stratification patterns. Surface waters begin warming in March and by June stratification has set in and the waters no longer mix. The months of June, July, and August are strongly stratified. Around September, cooler air temperatures and shorter solar days lead to a deepening and breaking down of the thermocline. Isothermal conditions generally occur in late fall and persist through the winter months. By November the lake has typically fully mixed and is thermally homogenous once again.

Typical summer thermal stratification conditions for Lake Shastina can be seen in Figure 10. Maximum water temperature of 20.4°C (68.7°F) occurs at the surface while 13 meters deep the water temperature is 11.4°C (52.5°F): a difference of 9°C (16.2°F). Large temperature differences between the surface and bottom waters are typical for Lake Shastina during summer stratification periods so long as there is sufficient storage.

The thermal profile of Lake Shastina on October 17, 2001 is illustrated in Figure 11. The change in temperature from top to bottom is 0.2°C (0.4°F) (14.6°C (58.3°F) at the surface to 14.4°C (57.9°F) at a depth of 8 meters) and there is no appreciable thermocline. However, there is a sufficient density difference in near-bottom waters to limit mixing in the near-bottom waters; this is apparent in the dissolved oxygen profile at the 7 to 8 meter depth (Figure 11). By November 2001 the water temperature in Lake Shastina appears to be uniform with depth (Figure 12). These fall profiles have less depth due to reservoir draw down.

In addition to persistent seasonal stratification, separating the epilimnion from the hypolimnion, there are short duration conditions that lead to weak, intermittent stratification in the epilimnion. An example of such conditions, sometimes called a secondary thermocline, may be seen in Figure 10 between the depths of 2 and 4 meters. The presence of this secondary thermocline is validated by the observed dissolved oxygen profile that illustrates depressed dissolved oxygen at 4 meters (Figure 10).

Surface water temperatures of Shasta River inflows to Lake Shastina, Lake Shastina near the dam, and the Shasta River below Lake Shastina are illustrated in Figure 13. A comparison of the data available over a similar time period indicates that surface water temperatures at above Lake Shastina and Lake Shastina surface water temperatures are roughly similar. Lake Shastina near the dam exhibits slightly warmer surface water temperatures in the spring of 1998. The site below Dwinnell dam is generally cooler than upstream locations in summer. This is probably due to release of cooler, deeper water. The below Dwinnell dam site is similar to upstream locations in late fall through mid-spring. The traces begin to diverge in the spring with the onset of stratification. In the fall, the discontinuity in the water temperature trace below Dwinnell dam during the October through November period most likely represents turnover. The data record is too sparse to draw more detailed conclusions concerning variability in water temperature among the three locations. Available temperature profiles are included in Appendix A and temperature profiles recreated from the 1971 limnological study (Dong et al 1974) are found in Appendix B.

8.1.2. Dissolved Oxygen

Lake Shastina exhibits typical dissolved oxygen characteristics of a eutrophic reservoir. Under summer thermal stratification the epilimnion is supersaturated with dissolved oxygen while the hypolimnion exhibits undersaturated conditions (Figure 10). Anoxia or near anoxic environments in the hypolimnion occurred between June and September of 2001.

The dissolved oxygen profile changes substantially after appreciable fall mixing (Figure 11). It is important to observe how dissolved oxygen levels are between 40 and 55 percent of saturation after whole lake mixing. These depressed dissolved oxygen levels, which range between 3.8 mg/l to 5.1 mg/l, are the result of mixing a relatively large anoxic hypolimnetic volume with a relatively small epilimnion volume, creating a reservoir that is overall undersaturated with regard to dissolved oxygen. This can have serious effects on biotic species that are sensitive to dissolved oxygen levels. Similar conditions were reported during the USGS limnological study in September 1971 and 1972 (USGS 1974).

Under winter conditions, Lake Shastina is generally well mixed and primary production is low, resulting in little or no variability of dissolved oxygen throughout the measured depths (e.g., Figure 12), and values are near saturation. The findings of the NCRWQCB are consistent with the results of the 1974 USGS limnological study of Lake Shastina (Dong et al 1974). Available dissolved oxygen profiles are included in the Appendix.

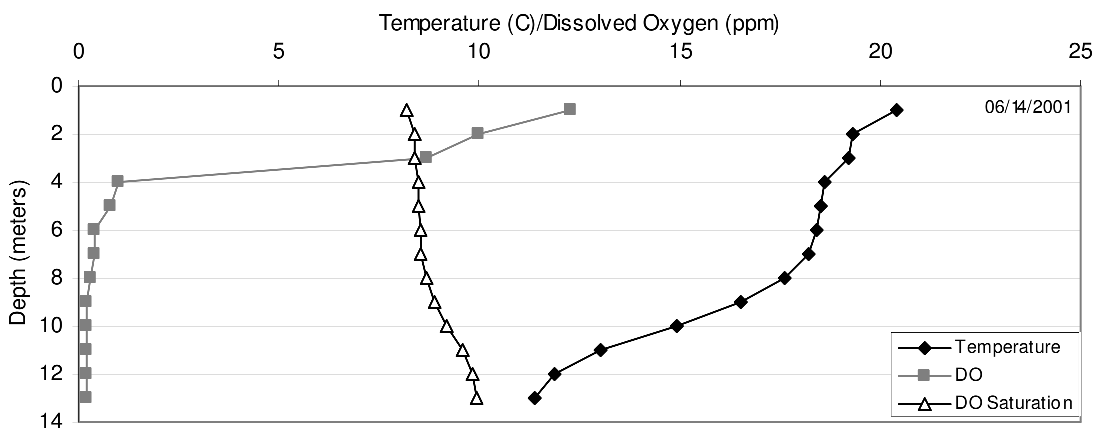

Figure 10 Lake Shastina temperature and dissolved oxygen profiles for June 11, 2001.

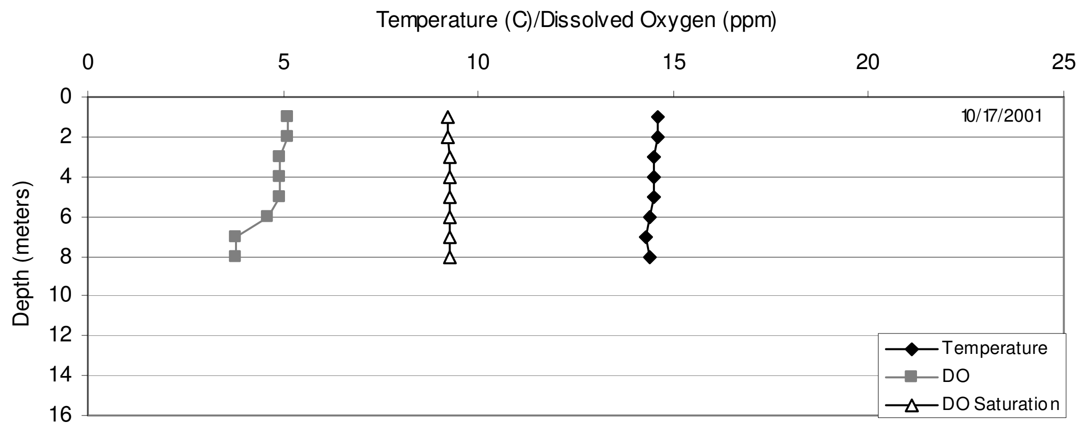

Figure 11 Lake Shastina temperature and dissolved oxygen profiles for October 17, 2001.

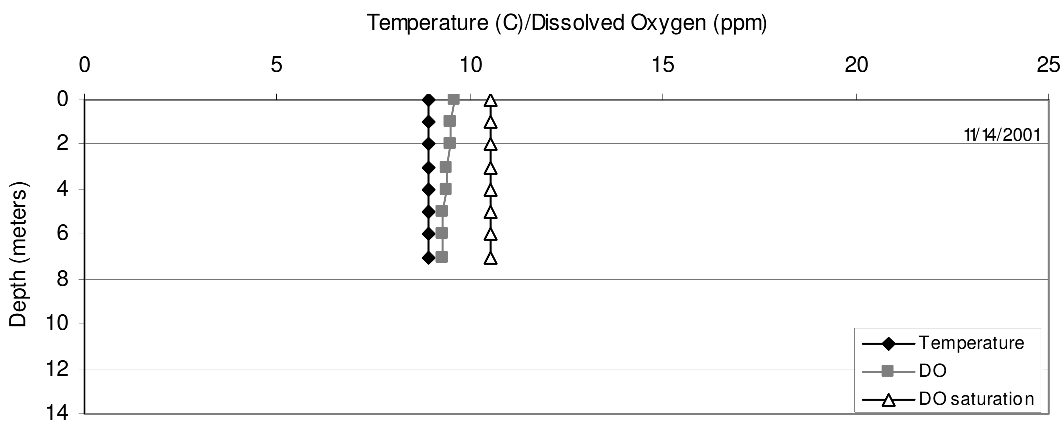

Figure 12 Lake Shastina temperature and dissolved oxygen profiles for November 14, 2001.

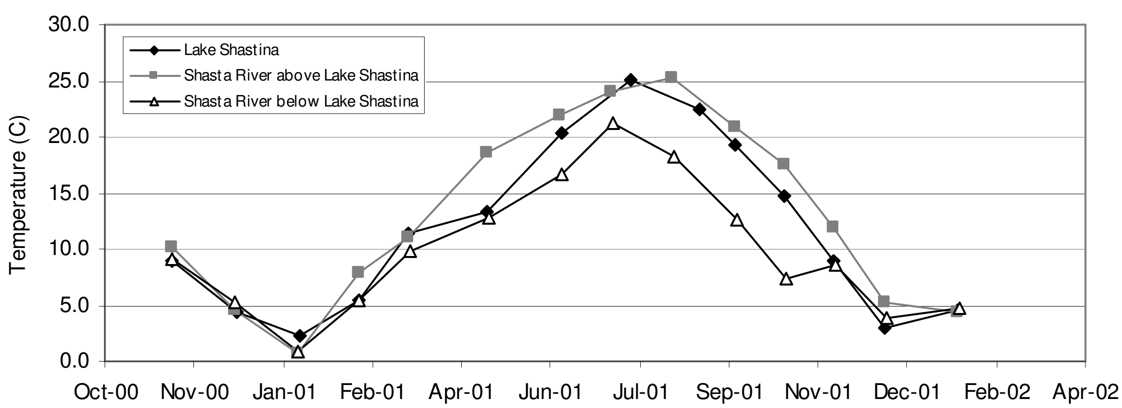

Figure 13 A comparison of surface water temperatures in the Shasta River above Lake Shastina the surface water temperature of Lake Shastina near the dam, and in the Shasta River below Lake Shastina.

8.1.3. pH, Alkalinity and Electrical Conductivity

Alkalinity, a measurement of the lake's buffering capacity against changes in pH, was measured near the dam in Lake Shastina between November 2000 and April 2002 (n=66) during lake profiling. The maximum recorded value was 201 mg/L and the minimum recorded value was 126 mg/L with an average of 154 mg/L. The summer average is 150 mg/l and the winter average is 160 mg/l. These levels of alkalinity fall within the transition zone (100-200 mg/L) and can result in increased pH levels during periods of high photosynthesis. However, such conditions are limited to late spring through fall when primary production is the greatest. Winter and summer alkalinity averages are shown in Table 6.

The pH of Lake Shastina during two periods of the year can be seen in Figure 14: summer, a photosynthetically productive time wherein the pH typically decreases with depth; and winter, once the lake has been mixed the pH is homogenous along the vertical profile. The pH decreases with depth during the summer because Lake Shastina is a moderately buffered system, as illustrated by the average alkalinity levels. During the growth season, algal production in near surface waters leads to increases in pH, a reflecting the impacts of algal utilization of bicarbonate to obtain CO_2 in a weakly buffered system.

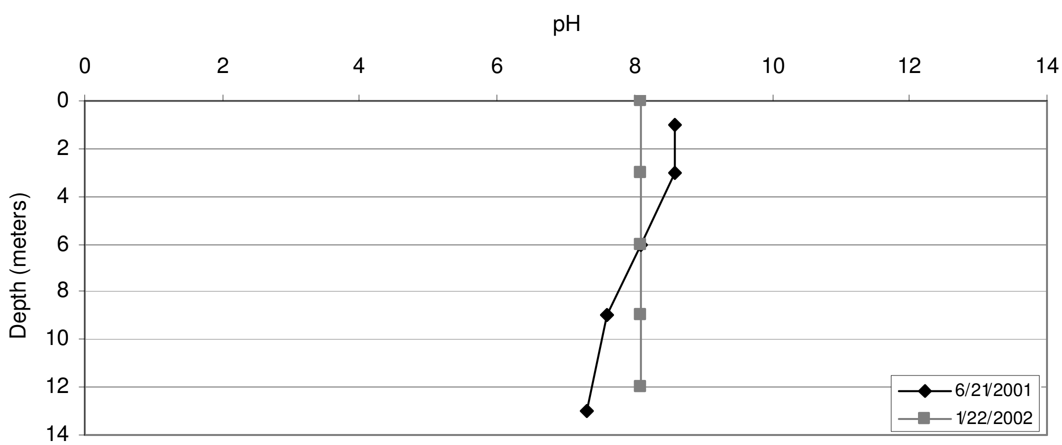

Figure 14 pH profile for Lake Shastina in June 2001 and January 2002 (DWR).

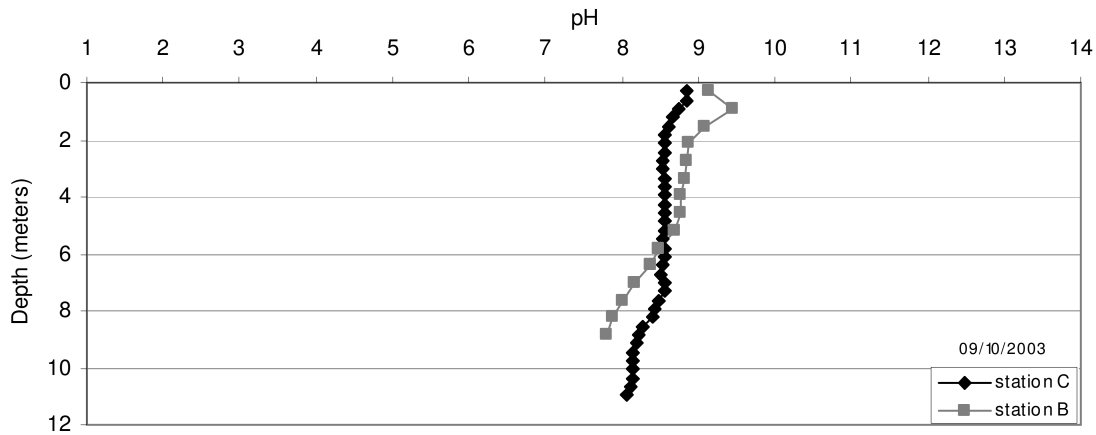

Figure 15 pH profile for Lake Shastina on September 10, 2003 (NCRWQCB 2004)

Data from two locations in Lake Shastina, near the dam (station C) and mid-reservoir (station B), are presented in Figure 15. The data illustrate the potential local impact that primary production has on pH – elevating near surface values to above 9 pH units at station B. Station C was sampled at 11:30 AM PST and station B was sampled at 1:45 PM PST (pers. com. M. St. John). Time of day, cloud cover, wind conditions, and other factors may provide insight with regard to the spatial variation between sites B and C.

8.1.4. Nutrients

Lake Shastina exhibits characteristics typical of eutrophic systems: elevated nutrient concentrations and significant seasonal primary production (as well as hypolimnetic oxygen concentrations well below saturation). Nutrient levels in the reservoir reflect inflow concentrations, as well as internal nutrient processes. It should be noted that the sampling dates often occurred when reservoir storage was low, particularly during the 1977 period. The nutrient data may reflect low storage levels in the reservoir and not provide a complete picture of Lake Shastina dynamics under the full range of storage conditions. Additional sampling over an extended period of time would provide additional insight into water quality response under variable storage conditions as well as reservoir dynamics during wet, dry, and normal years.

Table 6 The winter and summer averages of chemical and biological parameters in Lake Shastina (NCRWQCB 2004, DWR, and SWRCB).

Location		pH	Alkalinity (mg/L)	Specific Cond. (umhos/cm)	Nitrate/ Nitrite as N (mg/L)	Ammonia as N (mg/L)	Total Phos. as P (mg/L)	Ortho-phosphate as P (mg/L)	n
Lake Shastina (winter averages)	Near the dam (surface)	8.2	150	320	<0.05	ND	0.12	<0.05	13
Lake Shastina (summer averages)	Near the dam (surface)	8.3	160	300	<0.05	ND	0.12	<0.05	9

Vertical profiles of nutrients were completed by Dong et al (1974) and more recent NCRWQCB 2003 sampling in Lake Shastina. Some vertical variation in nutrient concentrations was observed in the September 2003 profiling (Table 7). There was a marked increase in ammonia and orthophosphate in near bottom waters at the NCRWQCB sampling location near the dam (station C) in 2003. Dong et al (1974) found well-defined vertical variation of ammonia, orthophosphate, total phosphorus, and total nitrogen at station L3, near the dam (approximately equivalent to station C). Higher concentrations of these nutrients were found in the anoxic hypolimnion, indicating

- nitrification inhibition of ammonia to nitrate under low dissolved oxygen conditions
- potential ammonia and phosphorous release from the sediments in an anoxic environment
- decay of organic matter to inorganic nutrients (NH_4^+ and PO_4^{3-}), but no appreciable update of nutrients by primary production due to light limitation

Table 7 Vertical profiles for nutrient species in Lake Shastina (n=1) (NCRWQCB 2004)

Station Location	Date	Time	Depth (m)	Nitrate/Nitrite as N (mg/L)	Ammonia as N (mg/L)	Total Phosphorus	Ortho-phosphate as P (mg/L)
B	9/10/2003	15:15	S*	ND	0.09	ND	ND
B	9/10/2003	16:00	4.0	ND	ND	0.07	ND
B	9/10/2003	16:15	8.0	ND	ND	0.16	0.08
B	9/11/2003	09:40	S*	ND	ND	ND	ND
B	9/11/2003	09:30	4.0	ND	ND	ND	ND
B	9/11/2003	09:15	8.0	ND	ND	ND	ND
C	9/10/2003	17:05	S*	ND	ND	0.59	ND
C	9/10/2003	16:55	6.0	ND	ND	ND	ND
C	9/10/2003	16:45	13.5	ND	1.60	0.18	ND
C	9/11/2003	10:10	S*	ND	ND	ND	0.17
C	9/11/2003	09:55	6.0	ND	ND	ND	ND
C	9/11/2003	09:50	13.0	ND	2.20	0.06	0.37

*S=Surface sample

Dong et al (1974) found "distinct vertical stratification of ammonium, orthophosphate, and total phosphorus, and total nitrogen (organic plus inorganic) at station L-3 [near the dam] during the summer thermal stratification period." Significantly higher concentrations of these nutrient species were reported near the bottom of the reservoir, in the hypolimnion, during this study. It is noted in the report that the greater total nitrogen concentrations in the hypolimnion were probably due to increased ammonium concentrations; the organic nitrogen concentrations were similar along the vertical profile (Dong et al, 1974). The Dong et al (1974) nutrient data is presented in the report as graphical data and could not be accurately replicated for inclusion in this report; please refer to the USGS limnological study for further information.

Although ammonia is generally non-detect, there are instances when it is present in surface waters. Ammonia in unionized form (NH_3) is highly toxic to fish at lethal concentrations and at sub-lethal concentrations may reduce growth, damage gills and other organs, and be a predisposing factor in bacterial gill disease (Colt et al 1979). Salmonids are particularly susceptible to toxic effects (Wade et al 1998). Magaud et al (1997) noted that variable concentrations of unionized ammonia may be more toxic than constant concentrations, the phenomenon being linked to the ability of fish to acclimate or adapt to new conditions. The amount of unionized ammonia can be computed in freshwaters knowing the total ammonia, pH, and temperature. The EPA (1999) calculated the acute criterion level of unionized ammonia for salmonids based on pH and temperature (Table 8). In Table 8 each column represents a particular temperature or range of temperatures in degrees Celsius, while rows represent pH levels. The values within the table identify critical criterion for salmonids based on the concentration of ammonia (in mg-N/L) present in the system. For example, at 18^{o}C and a pH of 8.0 the amount of ammonia that would lead to chronic stress in salmonids is 1.940 mg-N/L. At the same pH of 8.0, but at 22^{o}C, 1.50 mg-N/L will lead to chronic stress. Notice the inverse relationship where an increase in pH and/or temperature results in an environment where a smaller amount of ammonia will cause stress to salmonids than in an environment with cooler temperatures and/or lesser pH levels. The EPA (1999) information is only calculated out to a pH of 9.0; because Lake Shastina often exceeds a pH of 9.0 the amount of ammonia needed to cause stress at pH readings greater than 9.0 are extrapolated.

Table 8 The temperature and pH dependent critical criterion matrix for ammonia in units of mg-N/L. Adapted from EPA (1999).

		Temperature, °C				
		0-14.0	18.0	22.0	26.0	30.0
pH	7.6	**3.980**	**3.180**	**2.450**	**1.900**	**1.470**
	7.8	**3.180**	**2.540**	**1.960**	**1.520**	**1.170**
	8.0	**2.430**	**1.940**	**1.500**	**1.160**	**0.897**
	8.2	**1.790**	**1.430**	**1.110**	**0.855**	**0.661**
	8.4	**1.290**	**1.030**	**0.796**	**0.615**	**0.475**
	8.6	**0.920**	**0.735**	**0.568**	**0.439**	**0.339**
	8.8	**0.661**	**0.528**	**0.408**	**0.315**	**0.244**
	9.0	**0.486**	**0.389**	**0.300**	**0.232**	**0.179**
	9.2	**0.411**	**0.367**	**0.278**	**0.210**	**0.157**

8.1.5. Other constituents

The presence of hydrogen sulfide was observed in the hypolimnion of Lake Shastina on

August 24, 1971, a time when there were anoxic conditions (Dong et al 1974). Further, field notes from the September 19, 2001 sampling effort indicate that there was a sulfur smell when filling the bottom most bottles for the site in Lake Shastina near the dam. The dissolved oxygen levels on this day were 0.3 mg/L near the bottom, indicating anoxic conditions. Without consistent records it is difficult to understand the frequency with which hydrogen sulfide is observed in Lake Shastina.

8.1.6. Algae

The algal assemblage in Lake Shastina is typical of eutrophic waters. Data were collected by the NCRWQCB on July 30, 2004 at stations B and C. Please refer to NCRWQCB (2004) for details on sample collection and sites. Information regarding the algal species collected is provided in Table 9, including the abbreviations or "codes" that are used in Figure 17 and Figure 18 as well as the common name of the group to which the algal species belongs (e.g. greens, blue-greens, or diatoms).

Table 9 Algal species in Lake Shastina (NCRWQCB et al 2004).

Species Code	Species Name	Common Name of Group
ABFA	*Anabaena flos-aquae*	Blue-green
AFPR	*Amphora perpusilla*	Diatom
CHXX	*Chlamydomonas spp.*	Green
COPC	*Cocconeis placentula*	Diatom
COPD	*Cocconeis pediculus*	Diatom
CXER	*Cryptomonas erosa*	Cryptomonads
EPSX	*Epithemia sorex*	Diatom
GFAN	*Gomphonema angustatum*	Diatom
NZFR	*Nitzschia frustulum*	Diatom
RDMN	*Rhodomonus minuta*	Cryptomonads
RHCU	*Rhoicosphenia curvata*	Diatom
SCQD	*Scenedesmus quadricuada*	Green
SNRD	*Synedra radians*	Diatom
STAM	*Stephanodiscus astrea minutula*	Diatom
TEMN	*Tetraedron minimum*	Green

Phytoplankton often exhibit vertical variation, with diminishing numbers with depth. This is most likely due to light and nutrient availability. Total algal density (number of cells per milliliter) decreases with depth in Lake Shastina, as do chlorophyll a and phaeophytin concentrations (Table 10). There are modest differences between the two stations at the surface and 2 meter depths in terms of total density, but chlorophyll a data diverge more markedly. The deepest readings for each station were several meters apart and cannot be directly compared, but both profiles terminate approximately one foot above the bottom and results are consistent with what would be expected for deeper sampling depths.

Recall from earlier in this report that one of the limiting factors for algal growth is the depths of the photic zone or how far light will travel through the water, allowing algae to photosynthesize. In Figure 16 the measured Secchi depth is depicted along with the approximated photic zone (the depth where photosynthetically available radiation is approximately 1 percent of the value at the water surface). Although the ration of photic zone depth to Secchi depths varies, a rough rule of thumb assumes the photic zone is approximately 3 times the Secchi depth (Horne and Goldman, 1994). The transparency remains similar throughout the winter and early spring months (November 2000-April 2001) before it begins to decrease in clarity during late spring and summer, most likely due to algal growth. Secchi depth increases again in September, only to decrease through December, possibly due to increased sediment turbidity from mixing.

Table 10 Total density, chlorophyll a, and phaeophytin with depth of algae in Lake Shastina (NCRWQCB et al, 2004).

Station	Depth (meters)	Total Density (#cells /mL)	Chlorophyll a (µg/l)	Phaeophytin (µg/l)
C (near dam)	0 (surface)	3458	46.7	8.3
C (near dam)	2	2553	40.9	2.0
C (near dam)	12	9	35.0	1.9
B (near inflow)	0 (surface)	2988	26.5	21.8
B (near inflow)	2	2472	5.5	0.9
B (near inflow)	7	619	8.5	1.8

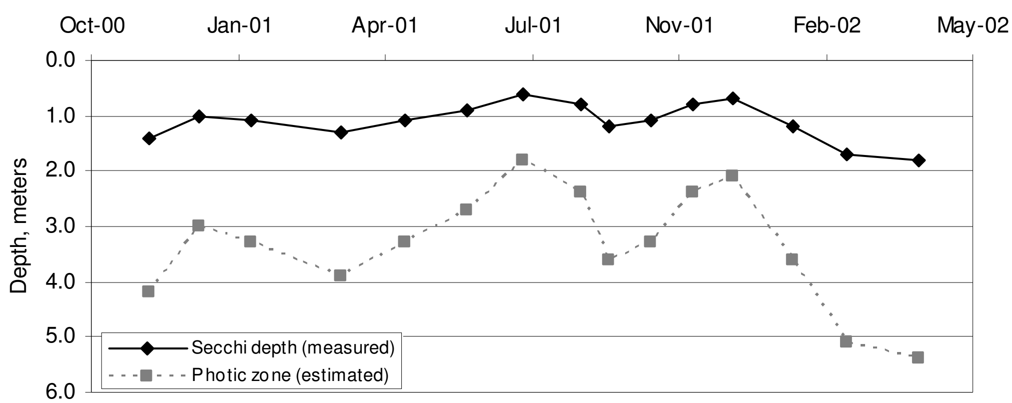

Figure 16 Secchi depth and photic zone depths for Lake Shastina near the dam (October 2000-April 2002; DWR).

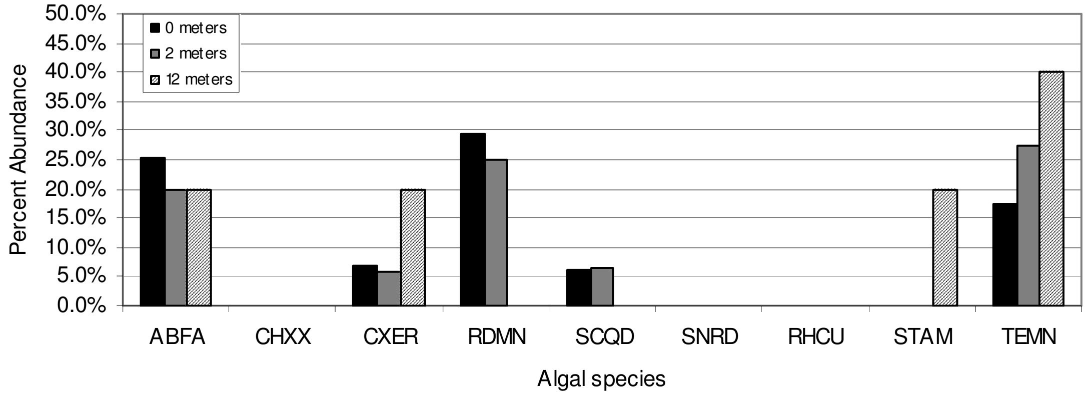

Figure 17 Common algal species by percent abundance at station C (NCRWQCB et al, 2004)

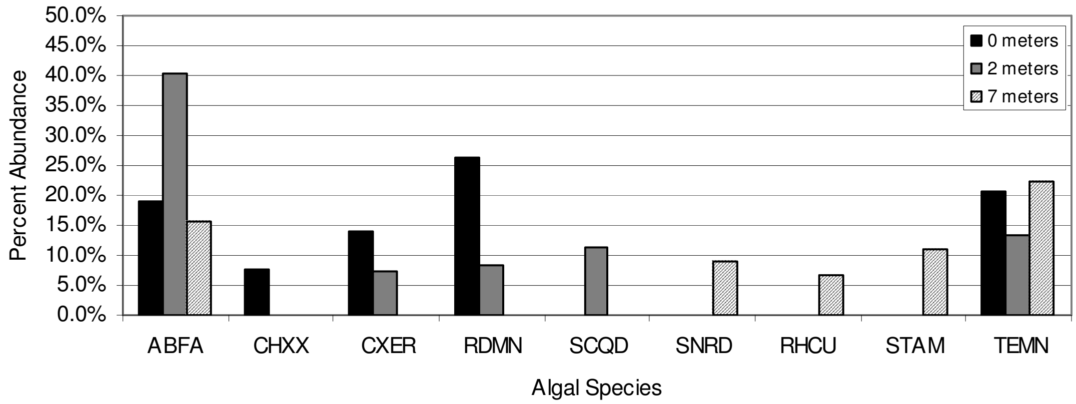

Figure 18 Common algal species by percent abundance at Station B (NCRWQCB et al, 2004)

At both station locations greens, diatoms, blue-green algae were present in the phytoplankton assemblage, which is consistent with the algal types present in the USGS 1974 report during the month of August 1972. Since algae, especially blue-greens, can be nuisance species it is important to regularly monitor their presence throughout the growth season.

One species of concern is Anabaena, which produce hepatoxin, a naturally produced chemical capable of causing liver damage. Hepatoxins can be produced by species and strains of *Anabaena, Cylindrospermopsis, Microcystis, Nodularia, Nostoc* and *Oscillatora.* Although short-term effects of these toxins may include visual disturbances, gastroenteritis, nausea, vomiting and muscle weakness, hepatoxins represent great concern to human health because of the risk of long-term exposure, to comparatively low concentrations of the toxins in drinking water supplies. Acute (short- term) exposure to high doses of hepatoxins causes death from liver hemorrhage or from liver failure. Chronic (long-term) exposure to low doses may promote the growth of liver, kidney and other tumors (Water Doctorate, 2001; Queensland Health, 2001).

Exposure pathways to cyanobacteria and their toxins include direct contact to exposed, sensitive parts of the body (e.g., ears, eyes, mouth, and throat); ingestion of water; consumption of fish or shellfish from contaminated waters; inhalation of water and aerosols. Swimming (ranging from wading to jumping and diving) and water sports (water skiing personal watercraft) are two activities providing exposure for all identified pathways. High exposure activities include direct contact activities (e.g., swimming, diving, water skiing, paddling, sail boarding, personal watercraft (jet ski)). Medium exposure activities include indirect contact or proximity through on-water activities (e.g., canoeing, sailing, and rowing). Low exposure activities include generally non-or minimal contact activities (pleasure cruising, fishing, non-contact shoreline recreation (picnicking, walking)) (Queensland Health, 2001; Stone, ____).

Health risks identified by the World Health Organization (Chorus and Bartram, 1999) for managing bathing waters that may contain cyanobacteria cells are:

- Low risk: <20,000 cells/ml
- Moderate risk: 100,000 cells/ml
- High risk: Cyanobacterial scum formation in contract recreation areas

Although waters of Lake Shastina generally exhibited low counts of Anabaena – maximum values on the order of 1000 cell/ml during summer months – these samples were collected in the main body of the reservoir where algae blooms are distributed roughly throughout the upper layer. However, wind can accumulate blooms in shoreline scums and cell densities can readily be increased by 1000 times or more (Brookes, et al, 2005).

It is recommended that more vigilant monitoring be completed at Shastina to determine if there are any health risks associated with toxic substances originating from cyanobacteria. Oregon currently manages blue-green algae conditions in surface waters through testing, advisories, and even lake closures. More information can be found at the State of Oregon Department of Human Services website[1].

8.1.7. Trophic Status

In accordance with the trophic status parameters and values set forth earlier in this report, Lake Shastina's trophic status is analyzed in Table 11. Overall the lake exhibits eutrophic characteristics. Although winter nutrient levels are unusually low, the amount of data is very limited. Inflow concentrations, as well as productivity of the lake, indicate that there are sufficient nutrients to produce an appreciable standing crop of phytoplankton. The adverse effects of water quality are well documented. DFG (1964a, 1964b, 1965a, 1965b, 1969, 1975, 1979, and 2001) report on various water quality conditions and/or fish kills at Lake Shastina. Identified problems that potentially lead to fish die-off included elevated temperatures, low dissolved oxygen levels or anoxia, algae blooms, elevated ammonia, and elevated pH. DFG (1975) noted that although summer fish kills occurred nearly every year during the 1960's; summer fish kills became less frequent in

[1] http://www.dhs.state.or.us/publichealth/esc/docs/maadvisories.cfm

the 1970's due to improved waste treatment and water quality practices in the drainage. This record, extending back over 40 years, indicates that Lake Shastina has had and continues to experience water quality impairment as a result of eutrophication.

Table 11 Lake Shastina's trophic status

Parameter	Lake Shastina		Trophic Status		
	Value	Status	Oligotrophic	Mesotrophic	Eutrophic
Nitrate – winter average (mg-N/L)	<0.05	O,M	<0.05	0.05-0.11	>0.11
Orthophosphate – winter average (mg-P/L)	<0.05	M	<0.02	0.02-0.042	>0.042
Total P – winter average (mg/L)	0.12	E	<0.02	0.02-0.042	>0.042
Secchi Depth – growth season minimum (feet)	1.97	E	>26	8-26	<8
Chlorophyll *a* – annual average (µg/L)*	n/a	-	<1	1-3	>3
Chlorophyll *a* – annual peak (µg/L)*	>40	E	<2	2-9	>9
Anoxia presence	Yes	M,E	Typically not present	Typical in stratified systems	Almost always in stratified systems–
Anoxia duration	~3 months	M,E	Typically not present	–Variable, months	–Variable, months
Anoxic extent	Can be extensive	E	–If present, typically small	–Variable, with modest amount of hypolimnion affected	–Variable, with significant amount of hypolimnion affected

*=Chlorophyll *a* was available on one date at various depths in the reservoir

9. Reservoir Outflows

Outflows from Lake Shastina are either regulated via the dam and canal for the MWCD water rights holders or are unregulated spill releases into the Shasta River. Measurements immediately downstream of the dam in the Shasta River near Riverside Drive indicate water quality is similar to the quality of water in Lake Shastina (Table 12). Comparing summer values suggests that pH is greater in reservoir waters, while specific conductance is lower. Because a portion of the water in the Shasta River below Dwinnell Dam is groundwater accretion, this variability is not unexpected. The most noticeable difference is the winter-to-summer variation in nitrate/nitrite as nitrogen values below the Dam. This may be due to low primary production in winter months and thus lack of uptake by algae. The MWCD canal shows similar nitrogen and phosphorus relationships as compared to the reservoir. There is a paucity of available data and additional long term monitoring is recommended to more fully characterize the interface between the reservoir and the downstream river reaches.

Table 12 Reservoir and downstream physical and biological parameters

Location		pH	Alkalinity (mg/L)	Spec. Cond. (umhos/cm)	Nitrate as N (mg/L)	Ammonia as N (mg/L)	Total Phos. as P (mg/L)	Ortho-phosphate as P (mg/L)
Lake Shastina (winter averages)	Near the dam (surface)	8.2	150	320	<0.05	ND	0.12	<0.05
Lake Shastina (summer averages)	Near the dam (surface)	8.3	160	300	<0.05	ND	0.12	<0.05
Shasta River (winter averages)	Riverside Drive (n=3)	8.0	157	320	0.57	ND	0.12	<0.05
Shasta River (summer averages)	Riverside Drive (n=18)	8.0	156	320	0.06	ND	0.10	<0.05
MWCD Canal (summer averages)	Below reservoir (n=7)	8.7	-	253	<0.05	ND	0.10	<0.05

Examining the benthic algae assemblage below Dwinnell Dam indicates that eutrophic diatoms dominate the river. The percent abundance of algal species at two sites immediately downstream of Dwinnell dam is shown in Figure 19. This data was collected within the same approximate sampling period as the Lake Shastina data presented above. There is only one algal species that is present in both the reservoir and the river, the diatom *Rhoicosphenia curvata.* This species is typically epiphytic (benthic) and found in alkaline flowing waters (Patrick and Reimer, 1966). Its presence in Lake Shastina is anomalous, but it could be washed in since it was detected at station B. There is insufficient data (both in the reservoir, canal and downstream river reaches) to assess the impacts of reservoir releases on algal species in downstream river reaches – both the fate of reservoir phytoplankton species in river reaches and the water quality implications on existing downstream benthic communities.

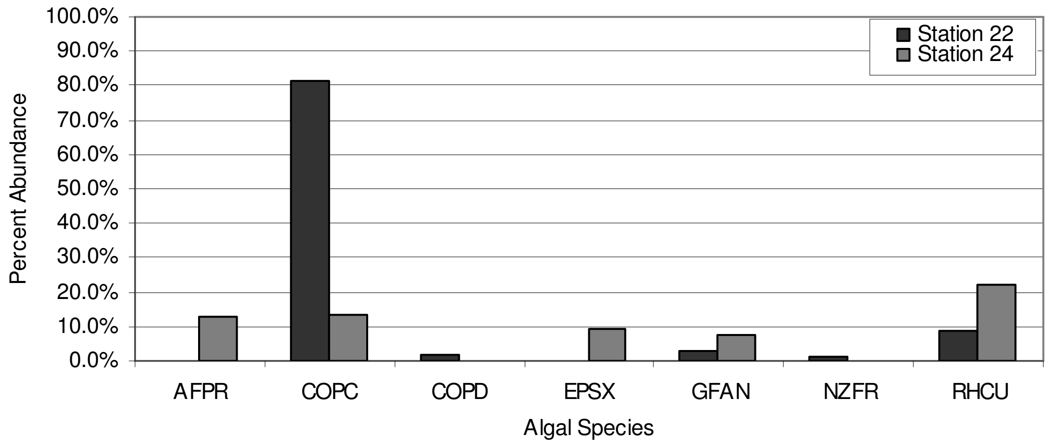

Figure 19 Percent abundance of benthic algal species in the river downstream of Dwinnell Dam at sampling stations 22 and 24 (NCRWQCB et al 2004)

9.1.1. Summary

The Shasta River, including the inflows of Boles, Beaughton, and Parks creeks, flows into Lake Shastina along with Garrick Creek. Lake Shastina is a typical eutrophic reservoir with a summer thermal stratification pattern, and anoxia in the hypolimnion.

Winter averages of nutrients suggest that nutrient inflow from the Shasta River is significant. To some degree these nutrients are naturally present in the system, as indicated by nutrient levels at the spring creeks flowing into the upper Shasta River. However, land use activities indicate that anthropogenic sources are present as well; however, it is unclear how much of the contribution in natural and how much is anthropogenic. It appears that the waters from Parks Creek and the headwaters of the Shasta River in the Scott mountains do not have the same elevated nutrient content that as the springs originating on volcanic Mount Shasta. Thus, the diversion from Parks Creek into the upper Shasta River potentially plays a role beyond simply volume of storage, but also in Lake Shastina limnology. The in-reservoir samples show lower levels of nutrients than the inflows, but the data are extremely limited.

Further, the reservoir experiences typical alkalinity readings between 100 and 200 mg/L, which translates to a moderately buffered system. The outcome is that during the growth season the pH exceeds 9.0 due to photosynthetic activity associated with primary production. Although ammonia is generally not present, there are instances when detectable levels of ammonia occur coincident with elevated pH values – leading to concern about ammonia toxicity. Generally, both nitrogen and phosphorus are at or below detection limits in surface samples, which implies that they are being actively used by phytoplankton.

The Shasta River downstream of Lake Shastina exhibits roughly similar water quality characteristics as the lake, but limited data do not allow detailed investigations of the interface between the lake and the river.

10. Lake Mixing: The Wedderburn and Richardson Numbers

In addition to assessing the trophic status of Lake Shastina, an assessment of the thermal stability and associated consequences of potential destratification under low storage were examined. Lake Shastina, the primary water supply reservoir for MWCD, experiences annual drawdown of approximately 10,000 to 30,000 acre-feet or more. Two assessments based on mixing theory were completed: application of the Richardson number, and calculation of the Wedderburn number

10.1.1. Richardson Number

The stability of seasonal stratification can be estimated using the Richardson number, R_i^*, which is defined as

$$R_i^* = \alpha g \Delta T h / u^{*2} \tag{3}$$

where α is the thermal expansivity of water (a function of water temperature), and ΔT is the water temperature difference across the thermocline. Values for the thermal expansion of water, α, and water density, ρ_w, vary as a function of mixed layer water temperature, T_w, and are presented in Table 13.

Table 13 Physical properties of water

T_w $(°C)$	ρ_w (kg/m^3)	α $(1/°C)$ $(\times 10^{-4})$
0 (32°F)	999.868 (62.420 lbm/ft^3)	-0.68
5 (41°F)	999.992 (62.427 lbm/ft^3)	0.16
10 (50°F)	999.726 (62.411 lbm/ft^3)	0.88
15 (59°F)	999.125 (62.373 lbm/ft^3)	1.51
20 (68°F)	998.228 (62.317 lbm/ft^3)	2.07
25 (77°F)	997.069 (62.245 lbm/ft^3)	2.57
30 (86°F)	995.671 (62.158 lbm/ft^3)	3.03

As presented in Fischer et al. (1979), the Richardson number can be calculated and used to classify lakes according to four distinct deepening regimes, which explain lake specific mixing process for different wind conditions (See Fischer et al. (1979) for additional details as well as Figure 20):

Regime A, where $R_i^* > (L/2h)^2$, represents a strongly stratified condition wherein sunlight is the dominant force in the absence of strong winds; internal waves form and decay over long periods of times; very slow deepening of the epilimnion is observed; and the thermocline is well defined and sharp.

Regime B, where $L/2h < R_i^* < (L/2h)^2$, stratification is still present; internal waves are still the prominent feature with slight billowing at the thermocline; billowing at the thermocline is the result of the mixing which is energized by the shear production; and slow deepening of the epilimnion interface is observed.

Regime C, where $1 < R_i^* < L/2h$, stratification breaks down; large billows with severe thermocline tilt are observed; mixing is mostly energized by shear production (wind). The final state is a longitudinal temperature gradient that will be mixed horizontally.

Regime D, where $R_i^* < 1$, deepening is rapid; there is no stratification, and severe vertical mixing leading to a weak longitudinal temperature gradient is observed.

The four regimes are illustrated in Figure 20, below.

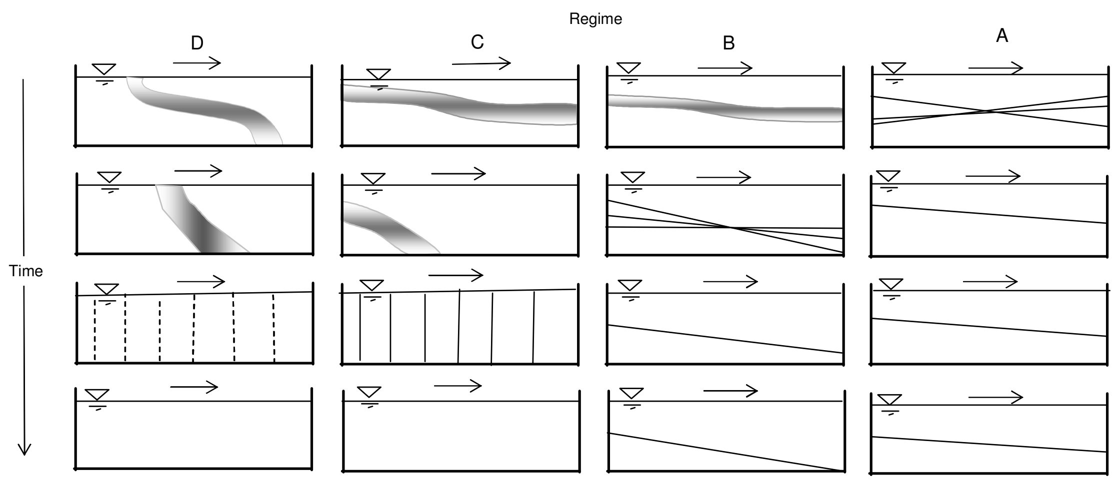

Figure 20 Illustration of the four mixing regimes. Directional arrow above each sketch indicates wind direction. Shaded areas represent mixing along the thermocline interface. Adapted from Fischer (1979).

10.1.2. Wedderburn Number

The Wedderburn number represents short-term mixing patterns (e.g., daily) in the epilimnion of a lake. In a reservoir with highly variable amounts of water stored at a time, such as Lake Shastina, this number may be useful in determining the ability of the lake to stratify at low lake levels. The Wedderburn number is a ratio of stratifying forces to wind mixing forces. Values less than 1.0 indicate that wind mixing conditions dominate and the epilimnion is prone to unstable conditions leading to no thermal stratification. Values larger than 1.0 indicate stability within the epilimnion and stratification within this layer may occur. A highly stable structure has the vertical density stratification strength that typical winds could not overcome. On the other hand, a lake in an unstable condition can be readily mixed by typical winds.

The Wedderburn number is defined as

$$W = \frac{g'h^2}{u_*^2 L} = \frac{\text{depth} - \text{based buoyancy}}{\text{length - based wind mixing}} \quad (1)$$

where g' is the reduced acceleration of gravity ($g\Delta\rho/\rho$, where g is acceleration of gravity, ρ is water density and $\Delta\rho$ is the water density difference across the thermocline), h is the depth of the mixed water layer, u_* (defined in Eq.2) is the characteristic shear velocity based on wind speed, and L is the fetch length (basin length in direction of the wind). The shear velocity, u_*, is computed as

$$u_* = \sqrt{\left(C_D \frac{\rho_A}{\rho_W} U^2\right)} \quad (2)$$

where C_D is an empirically-derived coefficient (1.0×10^{-3} for wind speeds up to 5 m/s (16.4 ft/s), 1.5×10^{-3} for wind speed of 15 m/s (49.2 ft/s). Linear interpolation was applied to estimate C_D values for wind speeds between 5 m/s and 15 m/s), ρ_A is the air density, ρ_w is the water density (dependent on mixed layer water temperature), and U is the wind speed above the water surface.

11. Analysis

11.1. Existing Conditions

Existing conditions for Lake Shastina stratification were calculated using the temperature profile data for 2001. Wind speed data from the Weed airport (accessed from CDEC) for the period of 1995 through 2004 were utilized. To select a representative wind speed, the 50 percent exceedance value was used. In addition, the maximum wind speed (99 percent exceedance) was also used in the analysis to determine potential for destratification as a worst case condition. Figure 21 through Figure 24 represent the exceedance curves for the Weed Airport wind data for individual months June, July, August and September. As illustrated in these figures, the wind conditions are quite similar during the summer months. Thus the 50 percent exceedance wind speed applied in the analysis was assumed to be was approximately 2.2 meters per second (4.9 miles per hour) for all months, and the maximum wind speed (99th percent exceedance) was assumed to be approximately 13 meters per second (29.1 miles per hour) for all months. Fetch length in this instance was assumed to be the length of the longest axis of the lake (6,120 meters). This fetch length is a conservative measurement because the wind may not always travel along the length of the lake's axis.

11.1.1. Richardson Number

Results of the Richardson number analyses under the 2001 conditions are shown in Table 14. Calculation of the Richardson number to determine potential mixing regimes indicates that the only month that the stratification regime does not change from a "B" to a "C" category with an increase in wind speed is June. Both increased storage and the difference in temperature between surface and bottom waters serve to maintain thermal stability.

In addition to a 50% and 99% exceedance value for each month, the critical wind speed at which the reservoir became unstable and switched from a B to a C regime was determined. The month of June was very stable in the reservoir and winds beyond the maximum recorded (13.0 m/s, 29 mph) would be needed to change from a stable B regime to an unstable C regime. However, July, August, and September all have the potential to transform from a B to a C regime. In July, the wind speed needed to destratify the lake would be 12.5 m/s (28 mph), which has a 99% exceedance value. Under the lower storage conditions of August and September the wind speed would only need to be 3.5 and 3.0 m/s (7.8 mph and 6.7 mph) respectively. This equates to a 73% exceedance value for both months or approximately once in every 4 years the lake has the potential to turn over in August or September.

Summer destratification is potentially detrimental to reservoir water quality conditions, possibly leading DO concentrations considerably less than saturation throughout the water column. Nutrients can also be introduced to the photic zone during summer destratification and lead to undesirable algae blooms.

Table 14 Model results for Richardson numbers under existing conditions in Lake Shastina.

Date	Wind Speed (m/s)	Storage (acre-feet)	Depth* (feet)	Richardson # R_i*	L/2h	$(L/2h)^2$	Regime
6/14/01	2.2	11,300	42.65	23,826	382	146,292	B
	13.0			487	382	146,292	B
	2.2			22,546	510	260,100	B
7/23/01	12.5	4,960	36.10	510	510	260,100	B/C
	13.0			461	510	260,100	C
	2.2			2,982	1,530	2,340,900	B
8/31/01	3.5	2,000	26.25	1,530	1,530	2,340,900	B/C
	13.0			61	1,530	2,340,900	C
	2.2			1,406	1,020	1,040,400	B
9/19/01	3.0	2,020	22.97	1,020	1,020	1,040,400	B/C
	13.0			29	1,020	1,040,400	C

* Temperatures reported until the noted depth; not necessarily through to the bottom of the reservoir.

11.1.2. Wedderburn Number

Results of the Wedderburn number analyses under the 2001 conditions are shown in Table 15. Recall, the Wedderburn number defines the stability within the epilimnion and determines whether or not a secondary thermocline may form, as one did in Lake Shastina in June 2001. For the analyzed conditions the Wedderburn number is well below 1.0 (unstable) in all but two instances: June and July at a wind speed of 2.2 meters per second. Although the August and September profiles did not reach the bottom of the reservoir (each fell about 5 feet short of the bottom), it was assumed to not significantly impact the results of the Wedderburn and Richardson analysis.

As discussed earlier in this report, a secondary thermocline is temporary, lasting from hours to possibly weeks. The presence of a secondary thermocline may lead to undesirable conditions including depressed DO levels below the secondary thermocline (see Figure 10). The result is that when mixing occurs within the epilimnion, the low dissolved oxygen water in the lower portion of the epilimnion will mix vertically with surface waters and the overall oxygen concentration throughout the full depth of the epilimnion may be well below saturation – a condition potentially adverse to fish populations and other aquatic life. Nutrients can also be mixed vertically under such circumstances, promoting algae growth.

Table 15 Model results for the Wedderburn number.

Date	Wind Speed (m/s)	Storage (acre-feet)	Depth (feet)	Thickness of Epilimnion (feet)	Wedderburn #
6/14/01	2.4	11,300	42.65	26.25	7.56
	6.3				1.00
	14.0				0.15
7/23/01	2.4	4,960	36.10	19.70	9.51
	7.0				1.00
	14.0				0.19
8/31/01	2.4	2,000	26.25	6.50	0.000
	N/A*				1.00
	14.0				0.000
9/19/01	2.4	2,020	22.97	9.85	0.000
	N/A*				1.00
	14.0				0.000

N/A* = A Wedderburn number of 1.0 is not possible under these reservoir conditions.

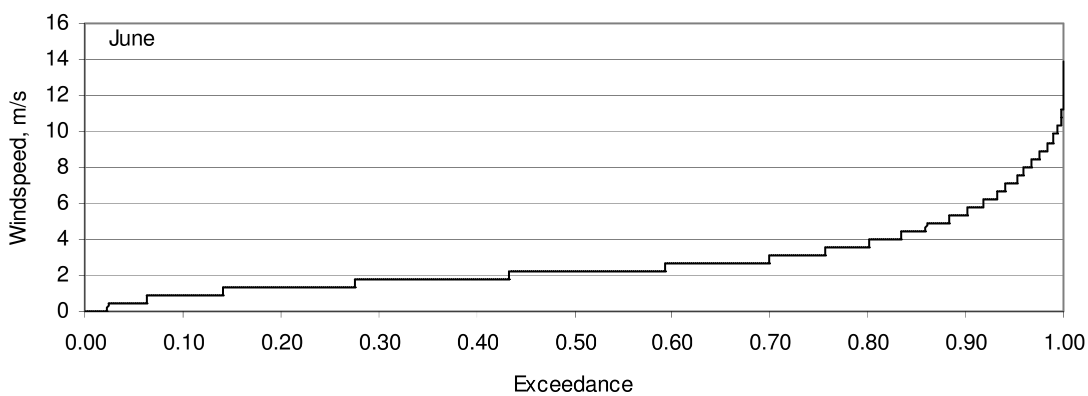

Figure 21 Wind speed exceedance graph for June, based on wind speed data from Weed Airport from 1995-2004 (CDEC).

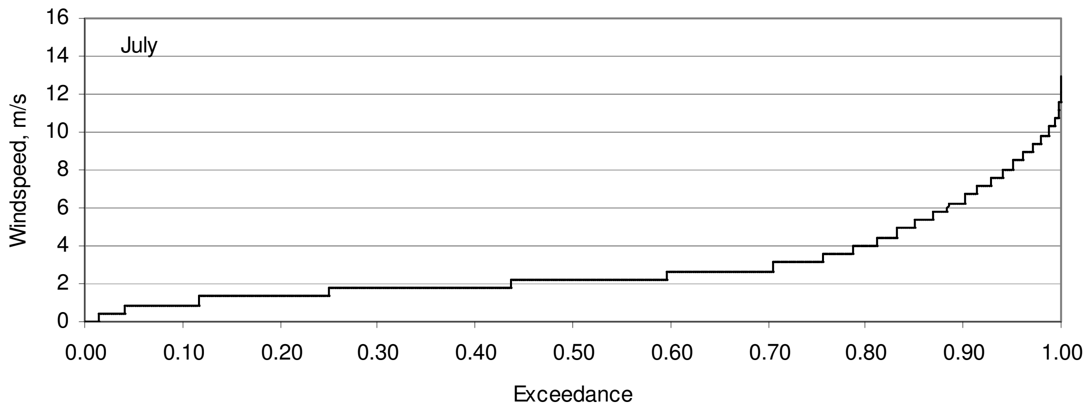

Figure 22 Wind speed exceedance graph for July, based on wind speed data from Weed Airport from 1999-2004 (CDEC).

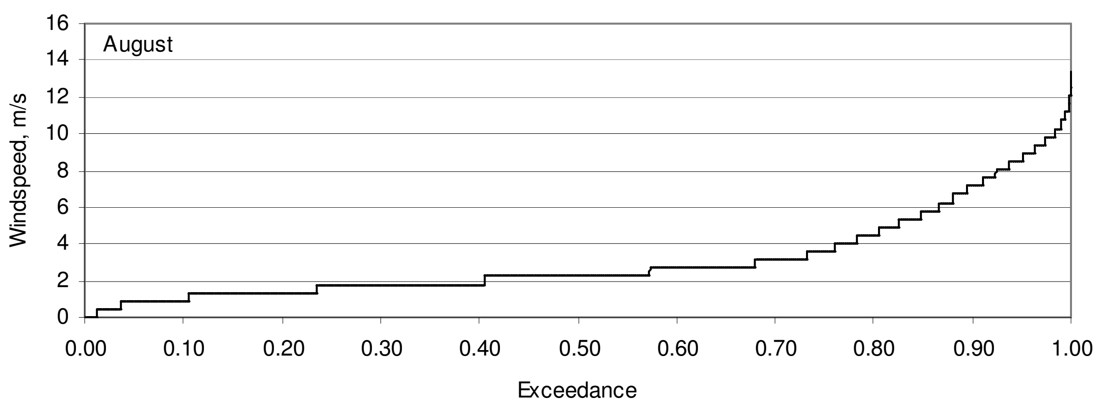

Figure 23 Wind speed exceedance graph for August, based on wind speed data from Weed Airport from 1995-2004 (CDEC).

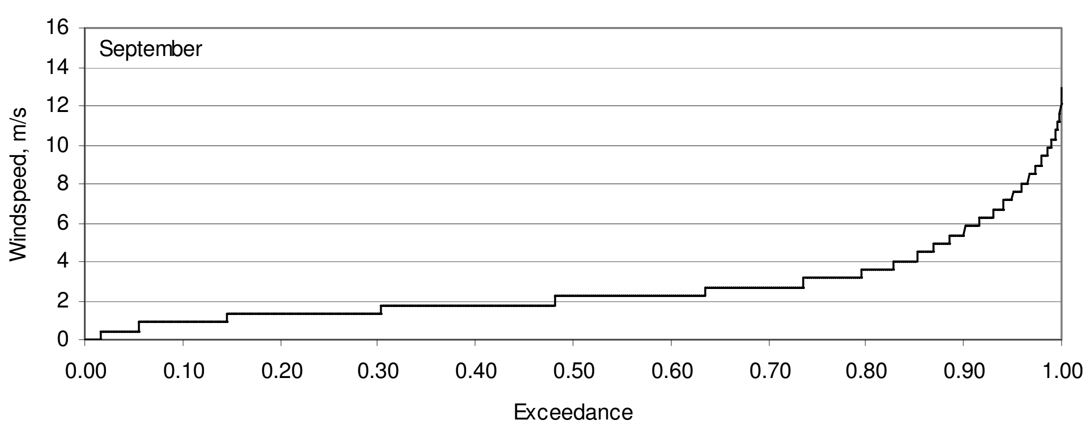

Figure 24 Wind speed exceedance graph for September, based on wind speed data from Weed Airport from 1995-2004 (CDEC).

11.2. Decreased Reservoir Storage

To assess the conditions under a decreased lake level two scenarios were devised: a decrease in 2 meters and a decrease in 4 meters of depth. The elevations for the original thermal profiles were approximated based on the CDEC average storage for the individual month and using linear interpolation to obtain the elevation based upon the Montague Water Conservation District storage elevation data. Subsequently a 2 meter (approximately 6.5 feet) decrease was applied to each thermal profile. Any elevation that was calculated to be less than 2,735 feet, which was previously determined to be the bottom of the reservoir at with storage equal to zero, was not used in the analysis. This leads to a loss of cold water at depth in the shallower profiles – a reasonable assumption. The following figures (Figure 25, Figure 26, Figure 27, and Figure 28) show the original thermal profile and the 2 meter shift for June, July, August and September 2001. It was found that a 4 meter decrease was too drastic in the scenarios and resulted in storage levels well below "dead" storage of 400 acre-feet.

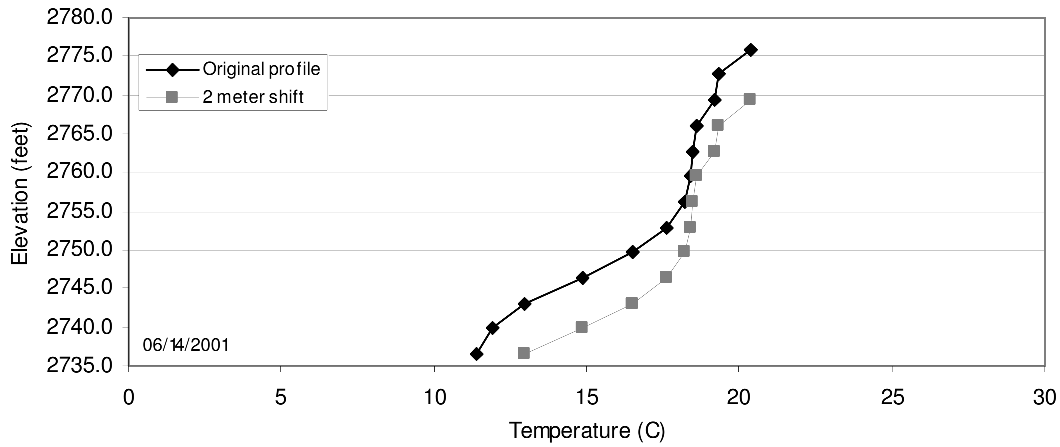

Figure 25 June 14, 2001 original and shifted thermal profiles for Lake Shastina.

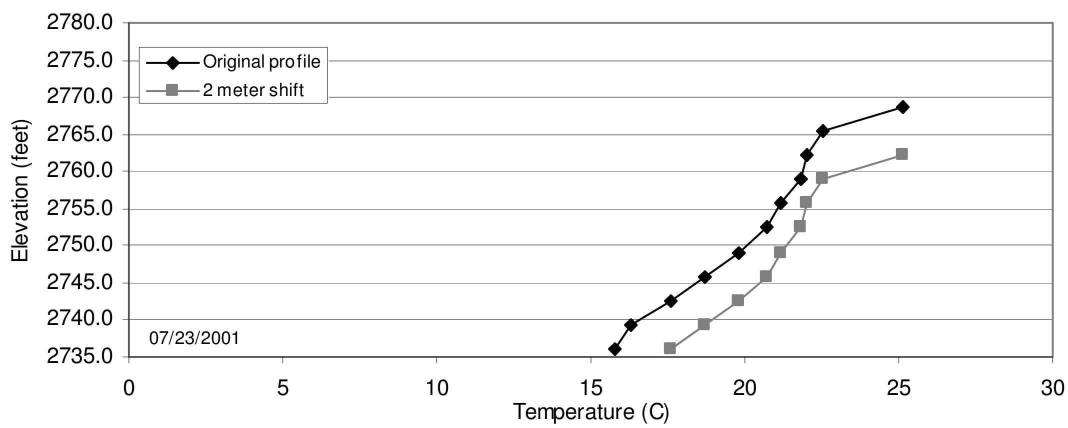

Figure 26 July 23, 2001 original and shifted thermal files for Lake Shastina.

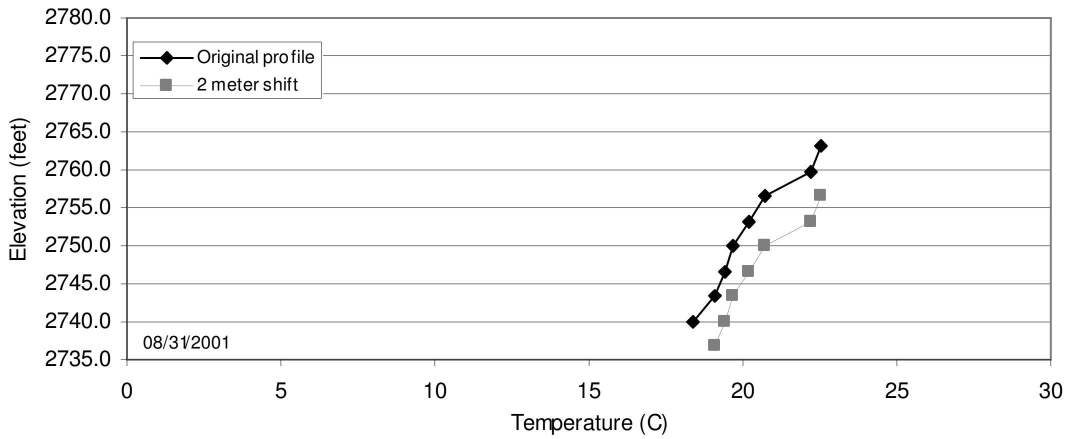

Figure 27 August 31, 2001 original and shifted thermal profiles for Lake Shastina.

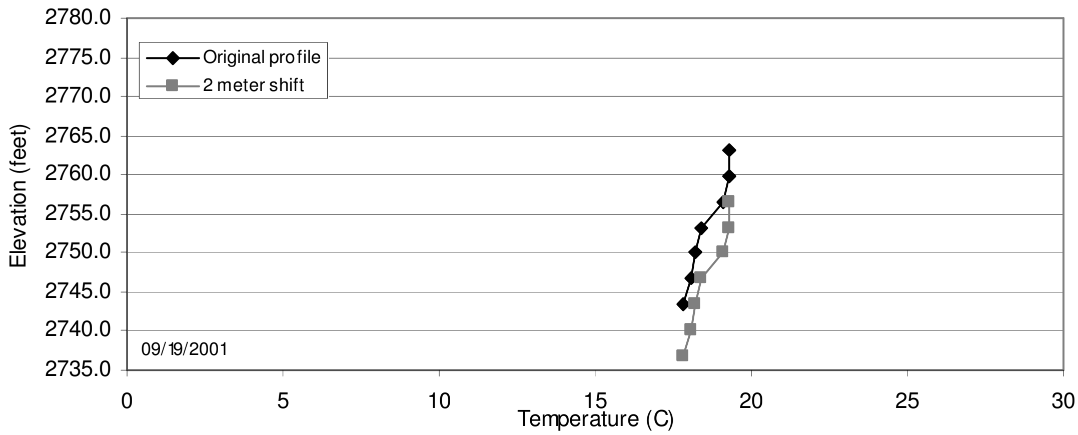

Figure 28 September 19, 2001 original and shifted thermal profiles.

The scenarios were analyzed in a similar manner to the existing conditions. Reported in Table 16, below, are the results of the scenarios in each month. In addition to reporting the 50 percent exceedance wind speed, as in the existing conditions, the approximate wind speed at which the system changed from a “B” regime to a “C” regime is reported. Modified August and September profiles resulted in storage that was below the estimated “dead” storage of 400 acre-feet.

Table 16 Wedderburn and Richardson analysis for 2 meter reduction in lake level

Date	Scenario	Orig. S (ac-ft)	New S (ac-ft)	ΔS (ac-ft)	Wind speed (m/s)	Wed. #	Richardson #			
							Ri*	L/2h	$(L/2h)^2$	Regime
6/14/01	Scenario 1 2 m/6.5 ft	11,300	5,347	5,953	2.2	8.999	19,590	382	146,306	B
					13.5	0.168	365			B/C
7/23/01	Scenario 1 2 m/6.5 ft	4,960	1,757	3,201	2.2	9.585	18,182	510	260,100	B
					11.5	0.265	503			B/C
8/31/01	Scenario 1 2 m/6.5 ft	2,000	396*	1,604	2.2	<0.000	2,473	1530	2,340,900	B
					3.0	<0.000	1,330			B/C
9/19/01	Scenario 1 2 m/6.5 ft	2,020	397*	1,623	2.2	<0.000	1,406	1,020	1,040,400	B
					3.0	<0.000	756			B/C

*=below level of "dead" storage (400 acre-feet)

Analysis of Richardson number indicates that lower storage results in a shift from the "B" to "C" regime under wind conditions that can occur in the Lake Shastina area. August and September were especially prone to destratification; when modeled, both months were destratified with winds that were 3 meters per second or less. Although June and July took considerably more energy and stronger winds, the wind speeds needed to turn the lake from "B" to "C" are not atypical of the area.

A considerable limitation of this analysis was the lack of available date at greater reservoir storage. Since only one year of data for thermal profiles are available for Lake Shastina it is not possible to make an accurate approximation of water temperature in deep water during a normal year. Nonetheless, the analysis does suggest that at lower storage, e.g., storage less than approximately 10,000 acre-feet, there is potential for summer destratification due to wind mixing. Summer destratification is potentially detrimental to reservoir water quality conditions, potentially leading DO concentrations considerably less than saturation throughout the water column. Nutrients can also be introduced to the photic zone during summer destratification and lead to undesirable algae blooms.

12. Reservoir Management Strategies

This section contains identifies reservoir water quality management strategies that might be considered for management of water quality conditions in Lake Shastina with regard to eutrophication, anoxia, and associated processes. The focus of this section is in-reservoir management strategies; however, watershed scale measures are also an appropriate approach to managing water quality in reservoirs such as Shastina.

12.1. Watershed Management

A critical element in any reservoir management strategy is to assess the potential impacts of watershed processes, conditions, and activities. Watershed practices to protect and sustain land and water resources are often called best management practices (BMP's). Within the watershed BMP's can be implemented to address both point and non-point sources of pollution. To successfully manage watersheds for management of water quality in lakes requires effective planning, wherein

- the purpose and objectives are defined (including active parties, scope, limitations, etc.)
- a description of the watershed is completed (including natural features, land and water resources development, water quality, known point and non-point sources)
- problems are identified (sources of impairment, how impairment is manifest in the watershed and water bodies, status of species of interest, etc.)
- planned activities are outlined (including goals, specific management objectives, timelines, adaptive management, etc.) (Holdren et al, 2001).

Best management practices (BMP's) for urban and rural land use, as well as water resources development and use, can provide important flexibility in water quality management. Point source management may include assessment of

- wastewater treatment facilities and septic systems
- stormwater management
- regulatory permitting of municipal and industrial water use
- hazardous material and contaminants programs (emergency response)
- agricultural return flow management.

Non-point source management may include assessment of

- agricultural practices (animal waste management, conservation tillage, crop rotation, fertilizer management, integrated pest management, range and pasture management)
- urban activities (flood storage, street maintenance and cleaning activities, salting and graveling of roads during winter)
- forestry practices (ground cover management, pesticide/herbicide management, riparian zone management, road and trail management)
- construction activities (erosion control measures, seasonal scheduling of activities)
- general land use activities (zoning and ordinances) (Holdren et al, 2001).

Watershed management can reduce nutrients and pollutants entering the reservoir but in many reservoirs, Shastina included, the natural background inflow (as well as potential internal nutrient cycling) of nutrients is sufficient to cause nuisance algae blooms. Therefore, several in-lake water quality management strategies are presented.

12.2. Reservoir Water Quality Management Strategies – Identification and Screening

Several water quality management actions are available for lakes and reservoirs. These in-lake practices may be used individually, but probably have the greatest utility when used in concert. This section contains an overview of reservoir water quality management strategies and briefly describes how these were screened for effectiveness and feasibility.

Table 17 provides a list of all the water quality management strategies evaluated in this chapter and identifies those that are considered effective and feasible for Lake Shastina. The background on these options can be found in most textbooks on lake management.

Table 17 Identified water quality control strategies for Lake Shastina (denoted with an "X")

Hypolimnetic Aeration	Alum Treatment
Hypolimnetic Oxygenation	Copper Treatment
Continuous Destratification	Hypolimnetic Flushing
Extended Winter Mixing	Hypolimnetic Release
Fall Destratification	Selective Withdrawal
Intermittent Destratification	Dilution
Hypolimnetic Mixing/Circulation	Blending
Biomanipulation	Drawdown
Wetlands/Riparian Buffer	Dredging
Algae Harvesting	Reservoir Operation Restriction

12.2.1. Addition Of Air Or Oxygen To The Hypolimnion

In Lake Shastina dissolved oxygen (DO) tends to become depleted in hypolimnetic waters. The hypolimnion is the cool, dense bottom water layer in a reservoir that lies below the warmer, lighter epilimnion on the surface. These two layers form in spring (stratification) and break down in fall or early winter (destratification). The water from the two layers does not generally intermix in summer so oxygen cannot pass from surface to the bottom sediments. This results in hypolimnetic anoxia (oxygen depletion). When this occurs, nutrients such as ammonia and phosphate are released from the sediments in a process termed internal loading. Over the course of the summer and fall these sediment-derived nutrients find their way into the surface waters where they promote excessive algal growth. In addition, anoxia in the hypolimnion can cause release of iron and manganese which can produce taste and odor. A reduction in internal loading can be achieved by adding oxygen to the sediments. In this way phosphate, ammonia, iron and manganese are all suppressed and production of organic taste and odor compounds such as mercaptans or other odors from anoxic sediments is also suppressed. Oxygen can be added in many ways ranging from stirring oxygenated surface water down to the bottom to the engineered addition of compressed air or pure oxygen. Aeration and oxygenation of the hypolimnion are described in this section.

Hypolimnetic Aeration with Compressed Air (Confined and Unconfined Air Bubble Plumes)

Hypolimnetic aeration supplies air containing 20 percent oxygen to the hypolimnion. Compressed air can be introduced into the hypolimnion in one of several ways. In most systems, one or more large compressors are installed near the reservoir and compressed air is piped to several deep locations in the lake. The most common variations are:

- Full Airlift Aerator. Compressed air is injected into the base of a vertical pipe that lifts both air bubbles and some entrained bottom water to the surface. During the lifting process some oxygen from the air dissolves in the water. The water emerges into a floating box where the majority of the added air escapes to the atmosphere. The partially aerated water then sinks back to the hypolimnion via another pipe. In some designs, the pipes are concentric. Iron is sometimes added to the floating box to enhance phosphorous removal. This simple system was originally popular but has been replaced in most places by partial lift systems that do not have large boxes floating on the water surface. Ice, wind, and vandalism are problems for full-lift aerators. In addition, the system requires that an aerator and its associated air pipes are placed at several sites within the reservoir.

- Partial Airlift Aerator. The partial airlift aerator is similar to the full airlift aerator except that the cylinder is capped below the water surface and gas is released via a pipe to the surface. Many of these systems are now in operation. As with full lift systems, partial lift aerators also have the drawback that several need to be placed throughout the reservoir. The associated piping costs can be considerable. Limnos-style "Rocket Ship" systems can be designed for this role. However, partial lift aerators have not been used for suppression of internal loading in reservoirs and it is not clear that they can actually work in this way. The remaining problem is that the plume released near the sediments may have a slightly higher density than the bottom water. Thus it will float above the bottom and not fully affect the sediments. The released plume is lighter since it still contains a small air bubbles and may be warmed at it rises through the full height of the hypolimnion. Since very large amounts of water are moved in lift aerators, their potential for sediments stirring must be accounted for. The greatest problem in operation most aerators has been under-sizing probably due to not accounting for the oxygen demand of the sediments. The greatest logistic problems presented by lift aerators is the large number needed for a normal reservoir and the amount of bottom piping required.
- Unconfined Bubble Plume. Fine bubble diffusers have been demonstrated to be effective in lakes with a hypolimnion depth of more than 100 feet. A rising bubble requires about 100 feet to dissolve completely. Destratification can result from the rising bubble plume in shallower hypolimnia if the bubbles are too large. The main problem with unconfined bubblers of all types is that the mixed cell caused by the bubbles is only about 14 times (diameter) the height of the water column. Thus in shallow lakes a large number of bubblers are needed. Since air will flow out of the shallowest orifice the pipes must be placed on a level gradient, a problem in lakes where the bottom usually slopes toward the dam.

Hypolimnetic aeration is an older technology and in most cases oxygenation would be used. If for some reason it is infeasible to supply oxygen to a reservoir site or to generate oxygen onsite, aeration is an alternative.

Hypolimnetic Oxygenation with pure oxygen (Bubble plumes and bubble-free systems)
Pure oxygen is generally delivered using one of two primary approaches: a bubble system or a bubble-free system. These systems are illustrated in Figure 29.

- Bubble systems. The bubble systems consist of pipes with small holes laid throughout the reservoir. Gaseous oxygen is passed to the underwater pipes and fine oxygen bubbles rise, losing oxygen to the water as they do so. If the hypolimnion is greater than 100 feet deep the bubbles will essentially dissolve completely. Typically the efficiency of bubble plume oxygenation is about 85 percent. The method is good for oxygenating the bulk hypolimnion but does not fully oxygenate the sediments. The non-bubble systems consist of some kind of pressuring device into which the deep water is pumped to compress the oxygen for an efficiency of almost 100 percent. Two popular types of bubble oxygenators are the Swiss-type unconfined fine bubble diffuser Figure 29a and the TVA-Mobley-type unconfined and diffuse bubble curtain Figure 29b. The Swiss-type bubble plume sends oxygen from the bottom at a few sites using discrete diffusers. The TVA-Mobley-type uses long arrays of garden soaker hoses that emit fine bubbles over the entire length of the hose. Several very large TVA-Mobley systems, some using 100 tons of oxygen a day are in use in the South East US.

- Bubble-free systems. The non-bubble oxygenators send oxygen into a pressurized container where the gas mixes with water pumped from the reservoir. The pressurized container can be situated at the bottom of the lake, taking advantage of the natural pressure of deep water. A downflow contact oxygenator system (DCOS) one version of which is a Speece cone is currently in use at the East Bay Municipal Utility District's Camanche Reservoir Figure 29c. The oxygenated water is then dispersed over the sediments via a short manifold or sometimes with a few direct pipes. In some systems a U-tube is used where a hole is drilled about 100 feet into the ground at the side of the reservoir or river. The hole is flooded and a U-tube inserted. Water is pumped into the U-tube and mixed with oxygen at the bottom where pressure is greatest, as in a DCOS. The oxygenated water flows out and is moved where needed via manifolds. Other pressurized tanks on the shore or with fast moving venturi systems (as in the Canning River in Western Australia) can be used to add oxygen without bubbles.

Oxygen is usually provided as liquid oxygen and stored lake-side but can easily by generated on site by a pressure swing compressor and molecular sieve.

Aeration is probably not feasible in Lake Shastina due to the required depth to either (a) install a full lift or partial lift system, or (b) provide sufficient depth to transfer oxygen from the air bubble to the water column. As with aeration systems, the fact that Lake Shastina experiences extreme drawdown on a fairly frequent basis, it is probably not practical to implement an oxygenation system. However, if a minimum pool was assigned to the reservoir to ensure sufficient water depth to maintain a hypolimnion it may be beneficial. Typically bubble plumes require about 10 meters of hypolimnion depth to provide sufficient oxygen transfer from the oxygen bubble to the water. Bubble free systems do not have this requirement. Due to the leakage from the reservoir it may not be feasible to maintain a minimum pool elevation. These facilities may present boating hazards, be unsightly, and require additional maintenance due to ultraviolet light exposure (TVA-Mobley type facilities). Given the expense of oxygenation, careful consideration as to the potential limitations and benefits should be explored prior recommending such management actions at Lake Shastina.

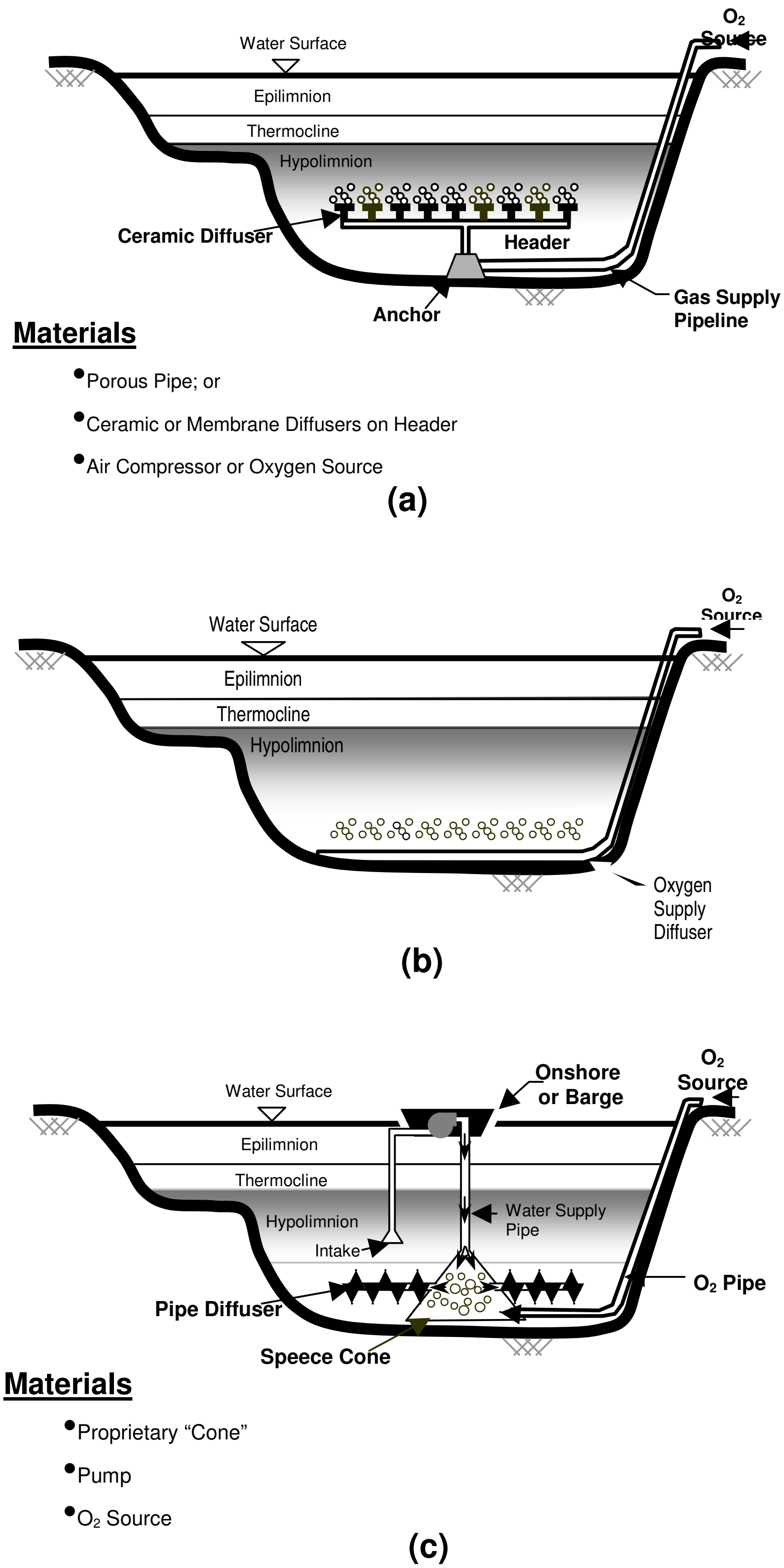

Figure 29 Schematics of Three Types of Oxygenation Systems: (a) Unconfined Fine Bubble Diffuser, (b) Unconfined Bubble Soaker Hose, and (c) Downflow Contact Oxygenator System (Speece Cone)

12.2.2. Continuous Destratification

In theory, continuous reservoir destratification provides two separate pathways to control algae. First, destratification may directly control algal growth by mixing algae out of the euphotic zone (surface zone with sufficient light for algal growth) where they die. Second, continuous destratification can be used as an indirect method of algae management by controlling bioavailable nutrients. In this case, destratification introduces oxygen to the sediments and reduces internal nutrient loading.
Both direct and indirect algae control methods have had mixed success. The most modern destratification experiments to control blue-green algae are being carried out in Australia. Destratification works in theory by preventing blue-green algae from floating near the surface and by providing a competitive advantage to diatoms. A dispute exists about the effectiveness of these systems since even a small temperature differential from top to bottom (as low as 0.2 °C) can prevent blue-greens from being mixed down. A concern with destratification is that the colonial blue-green algae such as *Microcystis* or *Aphanizomenon* would probably just manufacture more gas vacuole buoyancy and resist the additional vertical mixing. Very vigorous destratification experiments carried out by Dr. Horne and his students in Clear Lake in the 1970's showed little effect on large colonial blue-greens. However, the smaller colonies of the taste and odor forming small-celled *Anabaena* might be mixed and reduced by destratification if they do not form very large colonies.

For indirect management of algae through control of nutrients, results have similarly been inconclusive. Destratification can be as effective as hypolimnetic aeration or oxygenation in reducing internal nutrient loading, because all approaches deliver oxygen to the sediments. However, the problem lies in the fact that nutrient rich hypolimnion water and shallow sediments are mixed up into the photic zone during destratification where they are available to algae. But, as noted above, even small, vertical temperature differences can prevent blue-green algae from being mixed down and out of the photic zone. Many destratification experiments have been discontinued since algal growth was increased, not decreased. In principle, starting destratification early in the season, prior to thermal stratification should eliminate the problem but little evidence exists that this has been tried.

Destratification is accomplished with unconfined plumes of air provided by compressors and distributed with a network of pipes and diffusers that float above the reservoir bottom. In smaller reservoirs, propellers have been used to mix reservoir waters and breakdown or impede thermal stratification.

Given the mixed success, continuous destratification may not provide a robust management practice to nutrient control in Lake Shastina.

12.2.3. Extended Winter Mixing

California reservoirs tend to have a period of isothermal conditions (top-to-bottom reservoir mixing) that is only about 2 to 3 months long; although it is longer for Lake Shastina. This relatively short period of isothermal conditions favors growth of blue-green algae to form gas vacuoles that enable them to float to the surface during the lengthy period of stratification. Diatoms, without this ability to float, sink below the thermocline, where insufficient light for growth is present. In addition, the short period of mixing may not allow adequate time for surface oxygen to penetrate down into the sediments and fully satisfy the oxygen demand built up over the long period of stratification. If the summer oxygen debt is not fully satisfied during isothermal conditions, the hypolimnion will gradually become increasingly anoxic with resulting greater influx of nutrients from the sediments.

The extended winter mixing alternative involves extending the duration of isothermal conditions using compressed air or oxygen as in the continuous destratification alternative described above. The compressors could be activated in fall (early November) prior to the time that isothermal conditions would otherwise occur and run until December when normal isothermal conditions occur (See fall destratification, below). The compressors could be activated again in late winter (late February or March) and run until April when thermal stratification would be allowed to occur. The spring mixing operations should be timed to cease depending on the temperature of the water.

Extended winter mixing may provide some short-term benefit to Lake Shastina by delaying the onset of stratification and delaying the replacement of diatoms with blue-green species. There is a chance that the reservoir would completely lose the cold water pool and be prone to early turnover. Additional data would be required to complete an assessment of the benefits or limitations of this strategy.

12.2.4. Fall Destratification

The fall destratification alternative is similar to the extended winter stratification management alternative except that holomixis would be initiated in fall only. The objective is to introduce oxygen to the sediments and reduce the period of the anoxic period and thus the period of internal loading.

Fall destratification would probably not provide additional benefits at Lake Shastina because the lake turns over fairly early (generally by the autumnal equinox). However, in years when carryover storage is greater, this approach may provide some benefit.

12.2.5. Intermittent Destratification

Intermittent destratification can be used to create different conditions throughout the spring through fall period. One approach is to use intermittent destratification to create alternating oxic and anoxic conditions in the hypolimnion, which would favor denitrification. Intermittent destratification has been shown to be more effective at controlling blue-green algae blooms relative to continuous mixing in a 5-year lake test. Another application of intermittent destratification is to destratify impoundments that stratify only rarely. In these instances destratification is intended to prevent or breakdown stratification such that anoxic conditions that could lead to internal nutrient loading do not occur. Intermittent destratification is likely to raise reservoir release temperatures during summer periods if completed in large reservoirs prone to strong thermal stratification. This may adversely impact fish species below some dams that require colder temperature regimes (e.g., steelhead).

Given the mixed success, intermittent destratification may not provide a robust management practice to nutrient control in Lake Shastina.

12.2.6. Hypolimnetic circulation

Hypolimnetic circulation is a method to mix hypolimnetic waters to distribute oxygen without breaking down thermal stratification. This strategy is designed to introduce oxygen to the bottom sediments using available dissolved oxygen in the hypolimnion. Such circulation is not applicable to all reservoirs, and is typically limited to those with large hypolimnia that are predominately oxygen rich.

Typically there is insufficient oxygen available in the hypolimnion of Lake Shastina to support such an approach.

12.2.7. Biomanipulation

The biomanipulation alternative involves two components: summer harvesting of small fish that would feed on zooplankton and creation of refuges for zooplankton to hide from fish predation during the daylight hours. Fish are sight predators that must see zooplankton prior to eating them.

Algae are subject to grazing by zooplankton, which occurs at night in the epilimnion. The amount of algae that can be grazed by zooplankton depends on zooplankton abundance and characteristics of algae such as size and taste. Zooplankton are subject to predation by small- and medium-sized fish and in many lakes too many small fish are present. In nature, biomanipulation occurs naturally if adequate numbers of large fish are present to eat the medium and small fish. In many waters large fish are either over-fished or otherwise reduced in numbers. Studies have shown that algae can be reduced by restoring large fish stocks, even in lakes as large as Lake Michigan. The most effective work has avoided the indirect effect of large predatory fish by replacing them with human harvesting. In Europe, extremely good benefits have resulted from two years of summer fish harvest. Lakes that had not been safe for swimming for 30 years and had been subject to blue-green algae blooms became clear.

To be successful, biomanipulation also needs to provide a refuge for the main algae-grazing zooplankton, *Daphnia*, from fish predation during the day. *Daphnia* can avoid predation if it can swim down into the dark part of the lake, in this case the hypolimnion. At some times of the year no oxygen is present in the hypolimnion so no refuge is produced. The effectiveness of biomanipulation is dependent on the assumption that *Daphnia* will feed on very small blue-green algal colonies, even though this type of algae is rejected when the colonies are larger.

Biomanipulation has had mixed success, but may be a viable option for Lake Shastina. Additional study would be required to assess algae, zooplankton, and fish communities present in the reservoir.

12.2.8. Wetlands/Riparian Buffer

Wetlands could be designed and constructed to remove nitrogen and phosphorous from watershed runoff. Wetlands or mixed riparian buffer strips about 100 feet wide all along the length of the streams are becoming more popular in many areas, particularly in agricultural land (row crops and livestock). The surface runoff is intercepted, flows decreased and particles held in the wetlands and among the bushes. Subsurface runoff is intercepted by tree roots and evidence exists of enhanced denitrification with a riparian zone. Both nitrogen and phosphorous can be removed by such buffer strips and turbidity is reduced. Buffer strips work well in summer but this is not relevant to the local watershed where streams are running at low levels by May. Buffer strips also work best for winter pollution where concentrations of undesirable nutrients and sediments are high. Larger wetlands or a series of small wetlands that are actually set into the streams to intercept the entire flow (or flow in most storms) can be very effective in removing particles (including TP) but require a residence time of hours to days to be effective. However, wetlands are a very popular wildlife addition or restoration to watersheds. Thus wetlands on stream or with flood flows routed to them could give a water quality benefit if combined with wetlands built for other purposes.

Due to annual variability in water surface elevations, the fact that most inflows enter the reservoirs in winter, and limited appropriate land near inflowing tributaries, wetlands were not deemed an effective strategy for Lake Shastina.

12.2.9. Algae Harvesting

Large colonies of blue-green algae such *Aphanizomenon* can be harvested by suction from the surface followed by elaborate back-washed micro-screens. Blue-green algae are harvested commercially in Mexico, the Imperial Valley in southern California, and Upper Klamath Lake in Oregon. The product is sold to health stores as a complex high-protein supplement (up to 60% protein). However, the algae blooms need to be thick and the price of the product high to make removal economical.

Lake Shastina experiences notable algal blooms, but the species present may not be desirable or sufficiently extensive (or persistent/reliable) economically support harvest, i.e., few thick blooms occur and the vast majority of the blue-green algae that are present are widely dispersed through the surface waters.

12.2.10. Alum Treatment

Aluminum sulfate has been used in dozens of lakes in the United States and Europe to remove excess phosphorous and thus reduce algae blooms. Alum has been added to reservoirs to strip (precipitate) phosphorous from the water column or and influent stream and to chemically isolate phosphorous in sediments. Alum can be added to influent creeks and/or dispersed in the reservoir using a boat. Alum is added as a slurry to the epilimnion. Unlike the natural form of ferric phosphate, aluminum phosphate is not redissolved if the sediments become anoxic. Typical dose rates are 12 mg/L (as Al) which is expensive in a large lake and alters the lake chemistry in soft waters.

Alum additions have not always been successful, despite the simple chemistry involved. Where failure to reduce algal biomass by reducing phosphate has occurred with alum addition, it has often been because new sources of phosphorous arrive each year. In many reservoirs, new phosphorous arrives every year in the inflowing streams. In a wet year a large excess of new phosphorous may arrive that could supply the algae in a reservoir for many years. This new phosphorus is not held by the alum already in the sediments from previous years of treatment. In such cases, alum addition would probably have to be made every 2-3 years.

Given a cost of about $500,000 to $800,000 per treatment this alternative is not considered to be economical for Lake Shastina. In addition, potential environmental problems exist with such frequent additions including a thick layer of alum on the sediments and the lowering of the pH that is part of the chemical reaction during alum application. At a pH below 6.1, aluminum (natural levels plus the added amount) concentrations would be toxic to trout. Normally, this is a problem only in soft waters and can be eliminated by using other, but more costly aluminum formulations and by adding a buffer compound. This would increase the cost identified above.

12.2.11. Copper Treatment

Copper is toxic to algae and has long been used in reservoirs to prevent algae blooms. Use of copper in the reservoirs may increase the amount of copper in the drinking water supply but the amount of increase is a function of application rate and frequency and is thus uncertain. Because of a recent court decision that discharges of pollutants from the use of aquatic pesticides to waters of the United States require coverage under an NPDES permit, the State Water Resources Control Board (SWRCB) has developed a General Permit on an emergency basis in order to provide coverage for broad categories of aquatic pesticide use in California.

Copper is toxic to fish but only at concentrations in excess of those normally used for algae control. In addition, early work at Clear Lake showed that some blue-green algae can be controlled at doses many times lower than typically used (Horne and Goldman, 1974). Recent studies in Lake Mathews in southern California have indicated that targeted copper applications can greatly reduce copper use for taste and odor control for benthic blue-green algae. Targeted copper application at lower doses may be more acceptable to the public and the fish and game authorities.

Copper treatments can carry a risk of toxicity as well as adverse public response. However, such treatment may be considered under emergency conditions or if public perception changes.

12.2.12. Hypolimnetic Flushing

Hypolimnetic flushing is the discharge of hypolimnetic water to remove the nutrients accumulated during internal loading from the anoxic sediments over the summer. The technique was used successfully in Calero Reservoir (approx. 10,000 af., approx. 14 m deep) in Santa Clara Valley Water District (SCVWD) in 1991 (Seligman & Horne, unpublished). Taste and odor blooms that had caused the problem in the previous year were determined to have been due to phosphate buildup in the hypolimnion mixing with the epilimnion at overturn. Release of hypolimnion water in autumn decreased phosphate concentrations from 120 to 55 µg/L during the fall turnover. Since this procedure potentially consumes water, SCVWD decided against flushing as a routine management strategy. A pipeline was built around the reservoir for use when conditions favored taste and odor blooms. Flushing alone is not considered to be an adequate nutrient control strategy because most of the hypolimnion would need to be discharged to lower nutrients sufficiently to avoid nutrient conditions that favor undesirable algae in the reservoirs. Furthermore, this alternative reduces nutrients in the fall only.

Hypolimnetic flushing at Lake Shastina is not feasible due lack of existing facilities, but also would have limited efficacy due to the annual drawdown, small hypolimnetic volume, anoxic nature of bottom waters, and potential destratification of the lake during summer periods.

12.2.13. Hypolimnetic Release

Hypolimnetic release is intended to release nutrient rich water to downstream reaches through systematic long-term operations. Hypolimnetic release option may be constrained by reservoir outlet facilities and operations. Problems with hypolimnetic release include possible low dissolved oxygen and high nutrient water being released to the downstream water bodies. Adding oxygen may be necessary if the hypolimnetic release water is low in dissolved oxygen and downstream waters are oxygen-sensitive.

Hypolimnetic release is done at many reservoirs because the outlet works access the lower levels of the reservoirs. However, these actions are not generally carried out as active water quality management strategies, but rather as passive measures that are a result of the physical structure of the outlet works. In certain reservoirs, this operation has been identified as important to maintaining water quality.

Hypolimnetic release at Lake Shastina is not feasible due lack of existing facilities, but also would have limited efficacy due to the annual drawdown, small hypolimnetic volume, anoxic nature of bottom waters, and potential destratification of the lake during summer periods.

12.2.14. Selective Withdrawal

The ability to withdraw water from different depths (versus simply one depth) allows operational flexibility in that the best water quality can be selectively withdrawn, avoiding waters with impaired water quality. This may not benefit lake water quality conditions, but if waters are intended for downstream use, this strategy can provide increased flexibility.

Selective withdrawal at Lake Shastina is not feasible due lack of existing facilities, but also has limited efficacy due to the annual drawdown, small hypolimnetic volume, anoxic nature of bottom waters, and potential destratification of the lake during summer periods.

12.2.15. Dilution

Higher quality water can be added to a reservoir to improve the water quality in the reservoir. Dilution may not be feasible for certain reservoirs.

Dilution at Lake Shastina occurs during the winter season through direct diversions from Parks Creek into the Shasta River above the reservoir. Additional monitoring would be needed to quantify any benefit from such practices. Due to the seasonal nature of the diversion, benefits may be minor.

12.2.16. Blending

Blending water from different sources (e.g., reservoirs) can improve supply water quality, thus keeping treatment costs low. This strategy requires examining system-wide operations in an attempt to control timing of delivery from various sources to meet downstream water quality goals at various times of year.

Blending is not an option at Lake Shastina due to limited external sources of high quality water.

12.2.17. Dredging

Removing sediment from a lake can lower the in-lake nutrient concentration and algae production by preventing the release of nutrients from the sediments. Dredging can be completed when the lake is drained, or when the lake is full.

Dredging is generally expensive, requires an appropriate disposal area, and can lead to toxic waste disposal issues if contaminants are present in the sediments. Watershed measures may play a critical role in the decision to dredge a lake or reservoir. Dredging is an option at Lake Shastina.

12.2.18. Drawdown

Lowering the water level exposes sediment to the atmosphere and provides a means of oxidation and compaction. These processes can reduce the oxygen demand and phosphorous release rate in these lake bottom sediments.

Lake Shastina probably benefits, to some degree, from, the large annual drawdown in storage, or at least the fairly frequent (every several years) drawdown to low storage. However, results are mixed in terms of algae control, possibly due to the difficulty in dewatering the sediments sufficiently, drawdown has little effect on algal resting cysts, and the effects of drawdown are easily overwhelmed by high external loads of phosphorous (Holdren et al 2001). Drawdown at Lake Shastina may provide benefits, but the feasibility of this is limited due to conflicts with adjacent landowners (Shastina Properties), the possibility of undesirable water quality during low summer storage, and the fact that the lake is actively used for storing water during the rainy season may limit its efficacy.

12.2.19. Reservoir Operation Restriction

Maintaining reservoir storage or inflow and outflow within a fixed range can be an effective tool in water quality management. However, such restrictions may not be feasible for all reservoirs.

Minimum storage, changes in withdrawal timing and volume, as well as controlling inflow volumes may be effective management strategies to maintain or improve water quality both in the reservoir and downstream of the dam in the Shasta River. However, such options require consideration of existing water rights, public participation and acceptance, and necessary facilities to make operations feasible. Such considerations would require significant changes to the current operations and may or may not be feasible.

12.2.20. Conclusion

Management strategies, relative effectiveness, and cost to maintain and improve water quality conditions at Lake Shastina are outlined in Table 18. Watershed management activities can vary dramatically depending on their location, magnitude, scope, and intended purpose. Expensive watershed management practices may not yield attendant high returns in terms of Lake Shastina water quality improvement. Careful consideration to the economics and identified benefit should be given to BMP's and other activities aimed at improving water quality. Further, it is common that a single BMP or activity is insufficient, but a suite of BMP's or activities may provide a level of synergism and provide considerable benefits. Regardless of impact, well designed and implemented watershed management activities generally raise awareness of water quality issues in a basin.

With regard to in-lake strategies there are several critical factors that limit the efficacy of the identified management strategies for Lake Shastina:

- outlet facilities at Lake Shastina provide limited flexibility
- storage volumes are relatively small, leading to a relatively small hypolimnion
- reservoir drawdown is considerable in many years, limiting reservoir depth and resulting in early turnover (sometimes prior to the first day of fall)
- nutrient inputs to the system are sufficient to produce considerable algal blooms
- reservoir leakage is significant, leading to challenges if a minimum pool was to be identified
- modification of operations would require re-assessment of water rights and possibly re-allocation of waters
- there is insufficient data to completely assess many of the identified strategies.

Although there are several challenges to in-lake activities at Lake Shastina, there may be a combination of watershed and in-lake management strategies that will provide a desired level of benefit.

Table 18 Management strategies, relative effectiveness, and relative cost to maintain/improve water quality in Lake Shastina

Management Strategy	Relative Effectiveness	Cost
Watershed Management	Low to High	Low to High
Hypolimnetic Aeration	Low	Medium
Hypolimnetic Oxygenation	Medium to High	High
Continuous Destratification	Low	Low-medium
Extended Winter Mixing	Low	Low-medium
Fall Destratification	Low	Low-medium
Intermittent Destratification	Low	Low-medium
Hypolimnetic Mixing/Circulation	Not feasible	N/a
Biomanipulation	unknown	unknown
Wetlands/Riparian Buffer	Not feasible	N/a
Algae Harvesting	Not feasible	N/a
Alum Treatment	low	High
Copper Treatment	Low to High	Medium
Hypolimnetic Flushing	Not feasible	N/a
Hypolimnetic Release	Not feasible	N/a
Selective Withdrawal	Not feasible	N/a
Dilution	Unknown	Unknown
Blending	Not feasible	N/a
Dredging	Low to Medium (?)	High
Drawdown	Low to Medium (?)	Low
Reservoir Operation Restriction	Low to High	High
Watershed BMP's	Low to High	Low-medium

Relative Effectiveness
- Low – minimal overall effect
- Medium – some effect, possibly clear benefits in some years, with modest benefits in other years
- High – robust action that has readily measurable benefits in most years

Relative Cost (approximate)
- Low: < $100,000
- Medium: <$500,000
- High: >$500,000

13. Summary and Conclusions

A limnological assessment of Lake Shastina was completed based on previous studies and available data. Lake Shastina can be characterized as a eutrophic reservoir, with sufficient nutrients to produce an appreciable standing crop of algae during summer periods, as well as experiencing diminished light penetration and seasonal anoxia. Further, the lake is often drawn down considerably during the irrigation season, leading to potential for destratification during the growth season. Such conditions can lead to the introduction of nutrients into the photic zone during periods when phytoplankton populations can readily utilize these nutrients, leading to nuisance algae blooms.

In summary, Lake Shastina is a complex system that is characterized by the following conditions:

-Inflows include the Shasta River and Garrick Creek. The Shasta River inflows are supplemented by a diversion from Parks Creek into the Shasta River near Edgewood.

-Outflows from Lake Shastina consist of regulated flows through the Montague Water Conservation District canal and unregulated spills. Additionally, a great loss of storage in the reservoir is due to seepage or leaking into the groundwater.

-Meteorological conditions in the Shasta Valley include warm, sunny summers and cold, wet winters. These warm and sunny summers provide for a long growth season for phytoplankton.

- Due to the warm summers, thermal stratification occurs in the reservoir, resulting in a vertically stratified water body. Storage decreases, due to withdrawals in order to meet water rights downstream, can weaken the thermal regime and may result in early lake turnover and mixing.

-Nutrient levels are high, and are considered eutrophic or very productive, in Lake Shastina as well as the reservoir's inflows and outflows. This results in ideal conditions for phytoplankton growth as nutrients are not limiting factors. These high levels of nitrogen and phosphorus are especially beneficial to nuisance blue-green algae.

-Dissolved oxygen is often extremely low in the hypolimnion during the summer months of thermal stratification. When lake mixing occurs in late summer the whole lake exhibits low oxygen levels, resulting in negative impacts on biota and potential fish kills.

Activities to assist in long term management of Lake Shastina Water Quality include watershed management activities as well as in-lake strategies. Several activities and strategies are presented. More detailed exploration and assessment of these approaches is recommended.

13.1. Recommendations

Through the process of reviewing existing information and analyzing available data several recommendations for ongoing water quality management and assessment of Lake Shastina have come to light. These are outlined below.

(1) Limited field data are available for conditions when reservoir storage is greater than approximately 10,000 acre-feet. Further, a systematic assessment of the reservoir has not been completed, i.e., consisting of careful characterization of inflows, outflows, and operations of the reservoir; variations in vertical and longitudinal water temperatures; assessing water quality including dissolved oxygen, nutrients, and algae; and local meteorological conditions. It is highly recommended that a new bathymetry of Lake Shastina be performed in order to clearly characterize the surface area and volume and

dispel any previous inconsistencies. A general plan of action should be produced for reservoir management.

(2) Because reservoirs are dynamic environments that change over time, an adaptive management plan is highly recommended. In an adaptive management program, as new information from reservoir monitoring is gathered the data is evaluated and the results incorporated into the management plan as necessary.

(3) A comprehensive, long-term water budget for Lake Shastina is necessary to fully understand the dynamics of the reservoir. Included in this water budget should be inflows, specifically, Beaughton, Boles, and the Shasta River, Parks Creek's contributions, and Garrick Creek. Outflows to be included are seepage into the groundwater table, evaporation, and scheduled and unscheduled releases. Because Lake Shastina is an actively managed reservoir with variable year round storage that affects the water quality, water budgeting is also important for management purposes. Part of this process would involve keeping up accurate elevation and storage records, especially on days when samples are collected and profiles taken. Additionally, a meteorological station and evaporation pan, used to measure local meteorological conditions as well as evaporation, would prove useful in identifying patterns in those parameters and relating them to conditions at the lake. Further, characterizing the amount of leakage and under what conditions, if any, leakage is most likely to occur or to occur in greater or lesser amounts is also required.

(4) It is recommended that algal types in Lake Shastina and in the Shasta River be closely monitored. Blooms of any algal type, especially blue-greens, may be visually unappealing and have an offensive odor, damaging Lake Shastina's aesthetics. Additionally, some blue-greens, like *Anabaena* and *Microcystis*, can produce and release compounds that are toxic to other biological organisms. Algae sampling should include stations that are representative of the lake: near the inflows in the shallower part of the lake, mid-lake, and near the dam. Samples should be analyzed for biovolume (cells/mL) as well as individual algae identification. As Lake Shastina exhibits eutrophic conditions that are conducive to the blooms of toxic algae, such as some *Anabaena* species, monitoring for toxins and appropriate adaptive management strategies (e.g., recreational closure, warning signs) should be utilized.

If releasing appreciable volumes of water from Lake Shastina into the Shasta River is considered, a complete limnological analysis of the system should be done. One of the main components of such an analysis is to promote long-term study of the lake, rather than a single year study or a compilation of discontinuous years, because great variations in the lake dynamics may be seen from year to year, making reservoir management difficult and prone to error. Options for completing a limnological analysis are outlined below.

(5) Ideally, temperature profiles should be collected from top to bottom in at least two locations: near the dam and near the Shasta River and Garrick Creek inflows (at a minimum one location near the dam). The addition of a third location mid-lake would be

ideal. Temperature readings should be taken every meter until the bottom of the reservoir is reached. Data collection should be completed once a month (minimum during the summer, bi-monthly in winter). If time and money permits two readings should be taken during the months of June, July, August, and September, the readings spaced approximately two weeks apart. This sampling schedule will allow improved characterization of the changing thermal stratification during the summer and early fall season. At a minimum, temperature profiles should be taken quarterly: in October, January, April and July. Quarterly temperature profiles over the course of many years would be useful in assessing long-term trends. Characterization of inflow and outflow conditions are likewise recommended.

(6) Dissolved oxygen measurements should be collected in the same manner, and at the same time, as the temperature profiles above. Since most temperature and/or dissolved oxygen probes also collect pH and electro-conductivity (EC) it is advisable to collect this information, and any other data the probes may provide, at this time as well.

(7) Nutrient and selected physical parameters should be collected at the same locations as the temperature and dissolved oxygen. Grab samples should be taken every three to five meters, beginning with one meter below the surface and continuing until the bottom of the reservoir. These samples should be analyzed for total Kjeldahl nitrogen, NH_4-N, NO_3-N, and total phosphorous, PO_4-P, and total organic carbon, as well as total dissolved solids and alkalinity (at a minimum – other parameters can be added as deemed necessary). At a minimum, nutrient analysis should be performed quarterly, although more frequent, consistent observations are always ideal, especially during the growth season. Characterization of inflow and outflow conditions are likewise recommended.

(8) Minerals and metals only need to be analyzed twice a year in January and July, unless a particular mineral or metal becomes of concern, in which case more frequent sampling is advised at a rate to be determined at that time. This is a low priority item.

(9) Meteorological data is important. If possible, a met station should be set up near the lake to record air temperature, wind speed, wind direction, atmospheric pressure, and precipitation. A temporary met station during the summer months, when lake drawdown and turnover are occurring, is also an option.

(10) Ideally, analysis should also be completed on the inflows and outflows of Lake Shastina. Physical, chemical, and biological parameters should be sampled in Garrick Creek, Shasta River above the Parks Creek diversion, Shasta River below the Parks Creek diversion, Parks Creek above the diversion, Beaughton Creek and Boles Creek. Additionally, urban runoff from Weed and the Shastina community should be closely monitored. The Shasta River downstream of the dam as well as the MWCD canal should be examined as well.

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Personal Communications

William Bennett, California Department of Water Resources
Matt St. John, North Coast Regional Water Quality Control Board
Dave Webb, Shasta Valley Coordinated Resources Management Program

Appendix A

Figure 30 through Figure 43 are temperature and dissolved oxygen profiles from Lake Shastina near the dam (DWR).

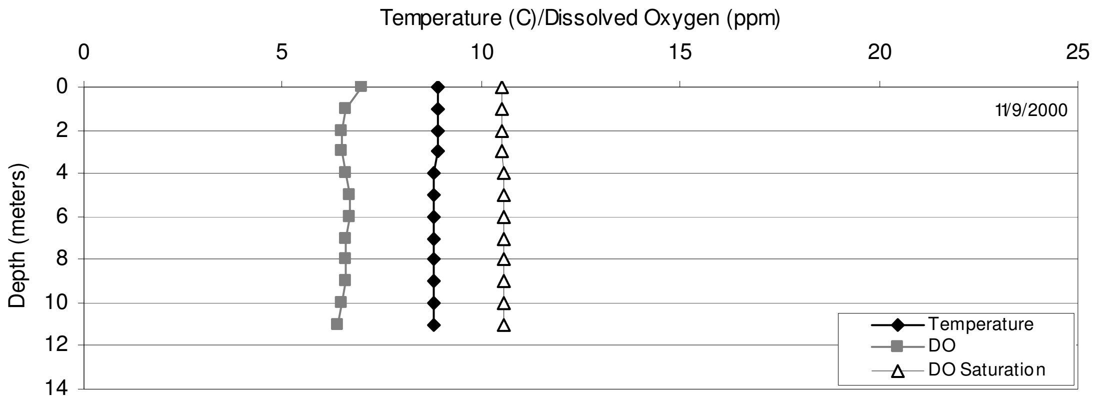


Figure 30

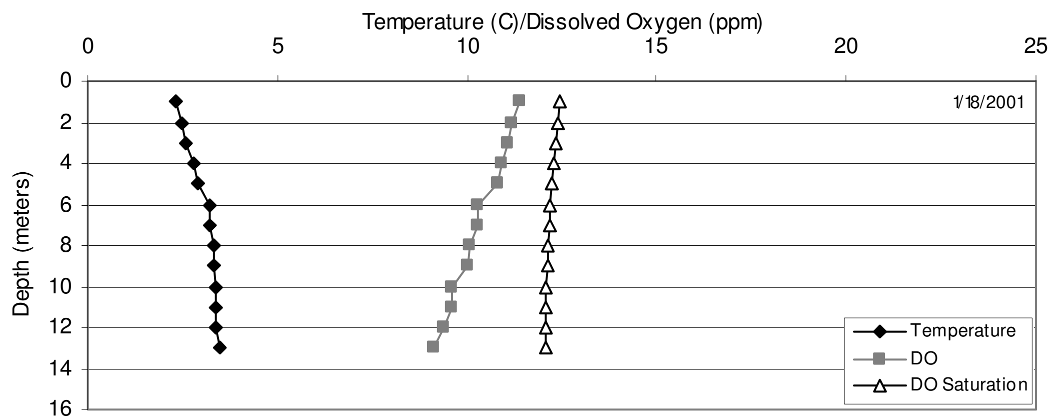


Figure 31

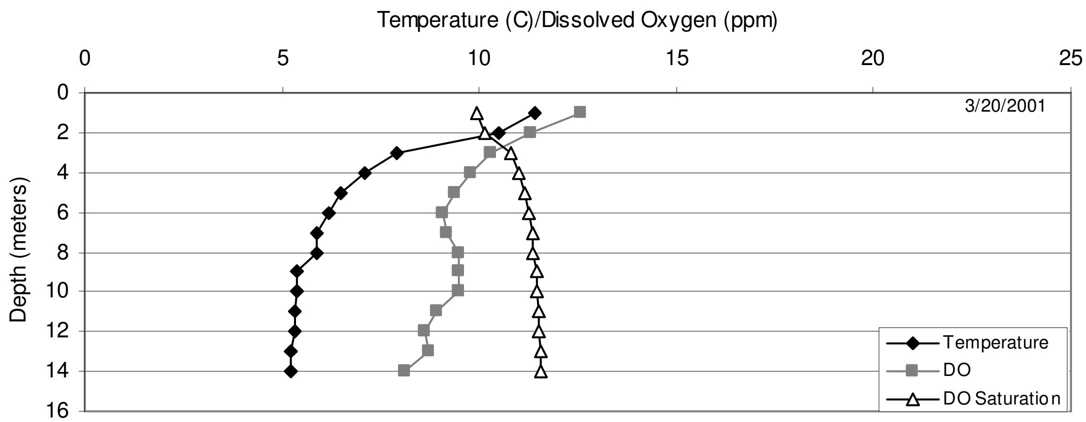


Figure 32

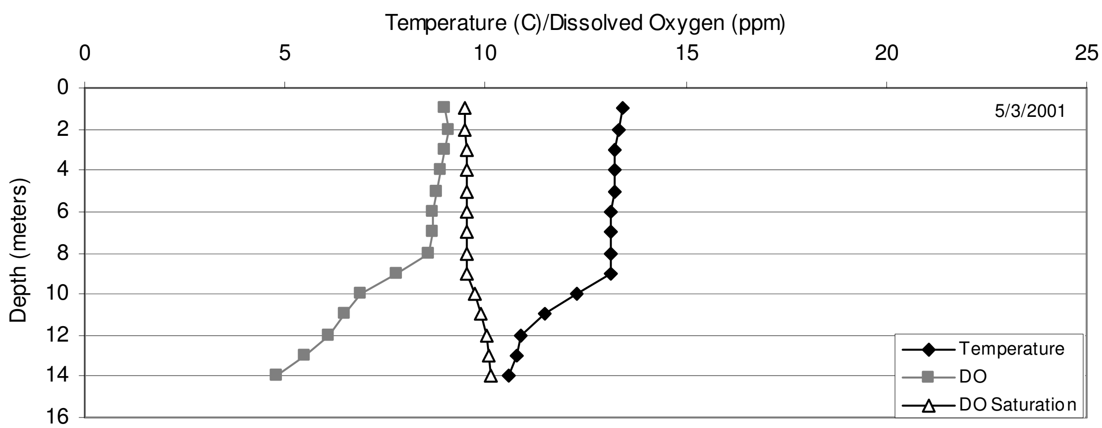


Figure 33

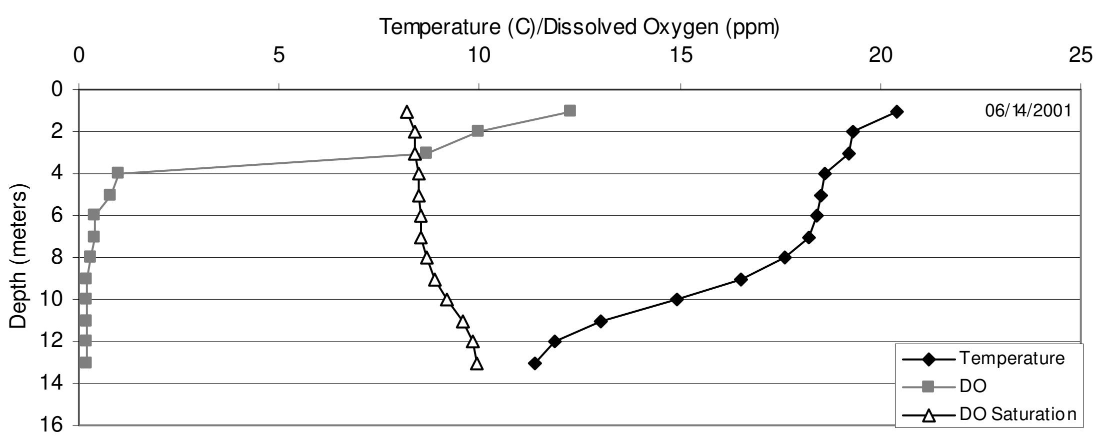


Figure 34

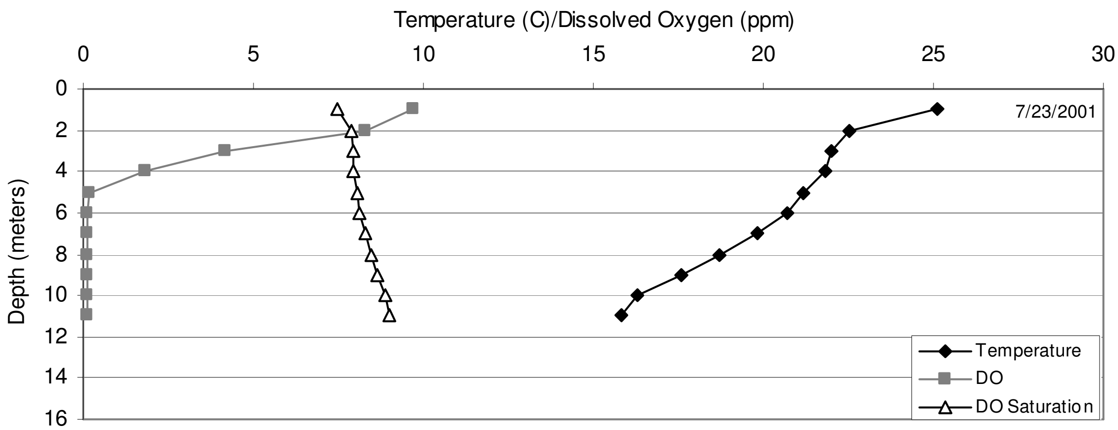


Figure 35

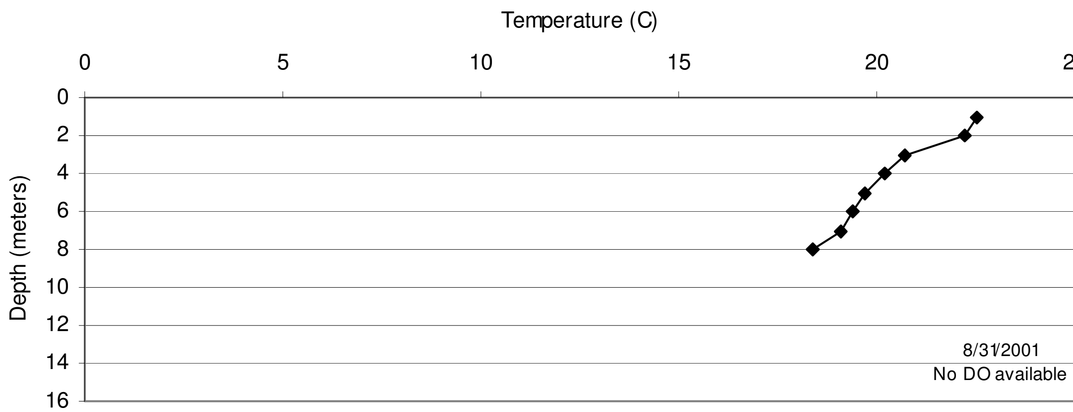


Figure 36

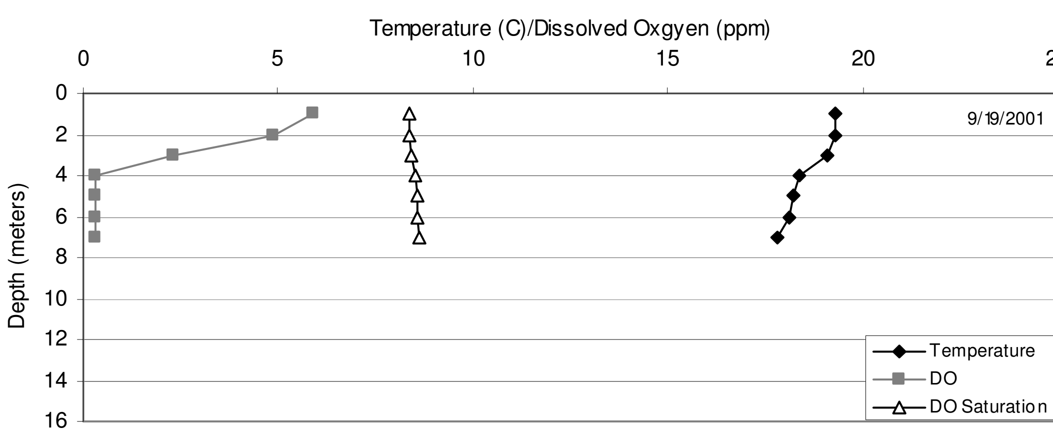


Figure 37

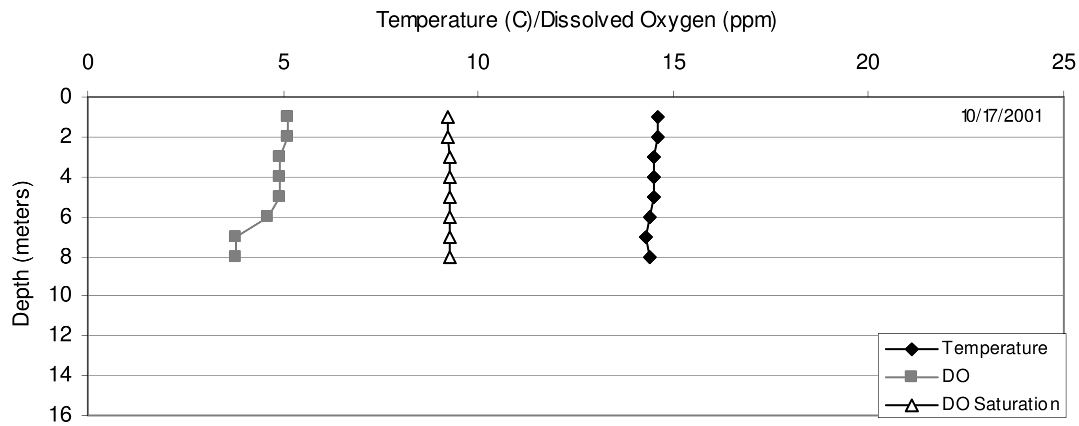


Figure 38

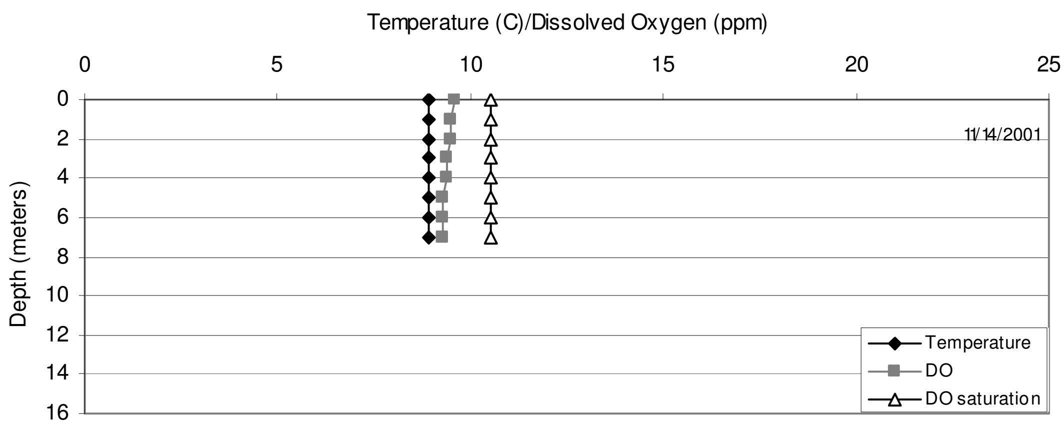
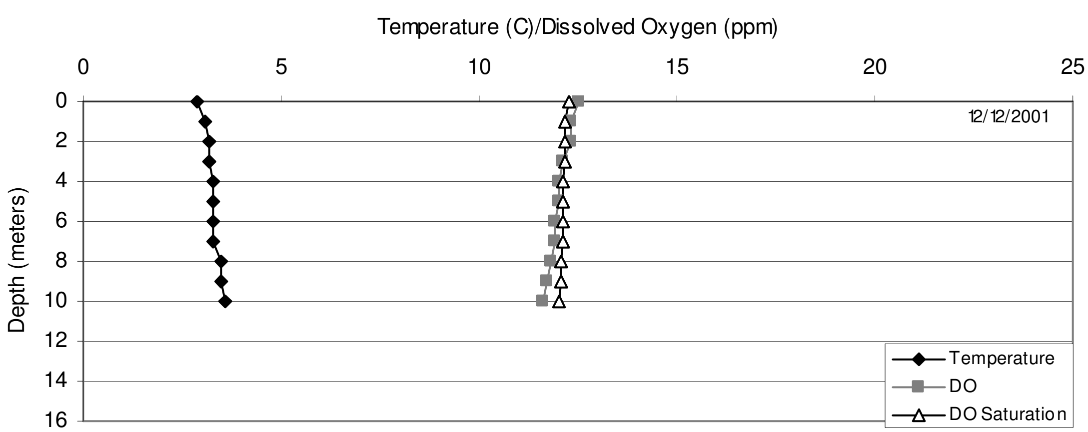


Figure 39



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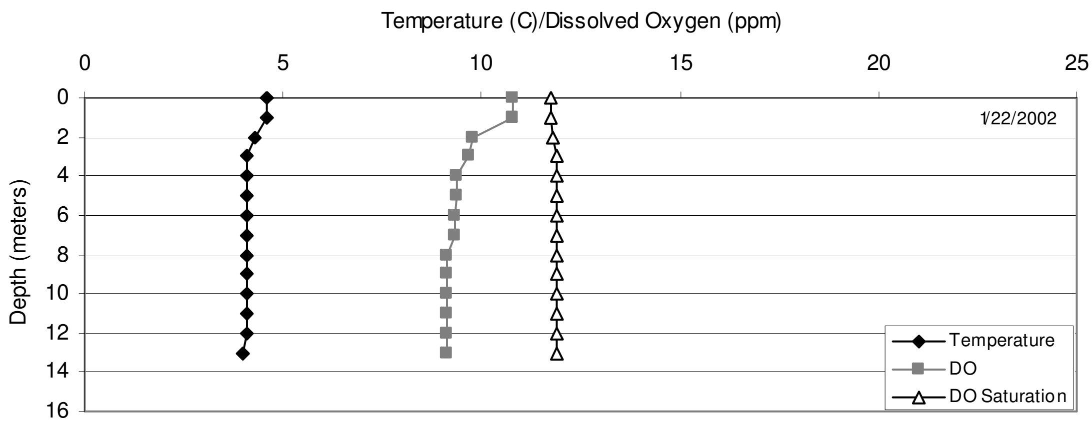

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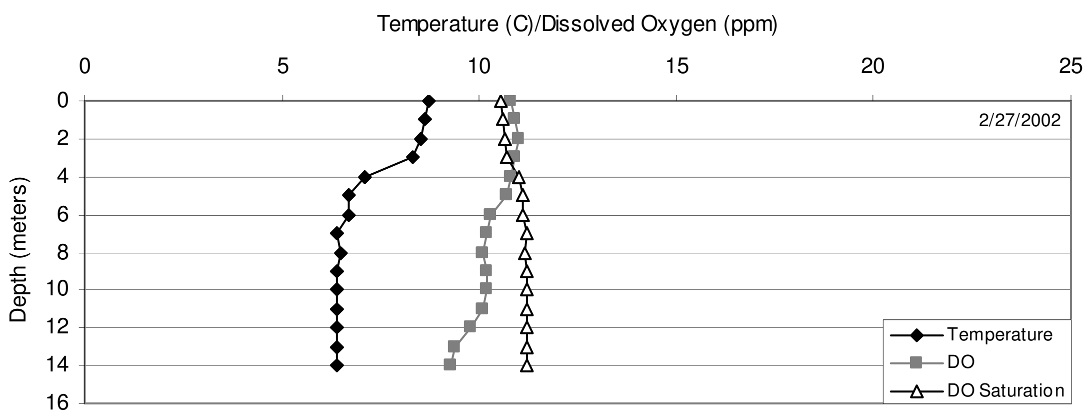

Figure 42

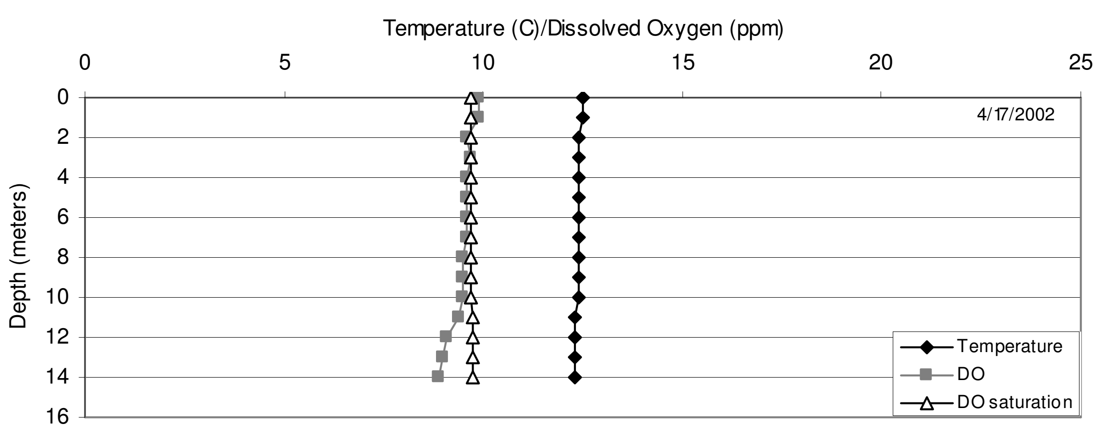

Figure 43

Appendix B

The following thermal profiles were recreated based on the graphs presented in Dong et al (1974).