Thermal and oxygen dynamics in four San Diego drinking water reservoirs

Raymond Lee
San Diego State University
Watershed Management Internship
May 2012 – May 2013
Water Resources Institute
U.S. Department of Agriculture

Advisor: Dr. Trent Biggs
San Diego State University

Submitted: May 29, 2013
# Table of Contents

ACKNOWLEDGEMENTS .................................................................................................................. 3

EXECUTIVE SUMMARY .............................................................................................................. 4

PROJECT OBJECTIVES ................................................................................................................ 5
  Introduction ...................................................................................................................................5
  Objectives ......................................................................................................................................6

PROJECT APPROACH .................................................................................................................. 8
  Data ...............................................................................................................................................8
  Land use .................................................................................................................................... 10
  Nutrient Modeling ....................................................................................................................... 10
  Analysis ....................................................................................................................................... 14
    Temperature and stability ......................................................................................................... 14
    Anoxia ....................................................................................................................................... 15
    Statistical Analysis ................................................................................................................ 16

PROJECT OUTCOMES ................................................................................................................ 16
  Lake volume and lake thermal dynamics ....................................................................................... 16
  Anoxic Factor ............................................................................................................................... 18

CONCLUSIONS ............................................................................................................................. 19

REFERENCES ................................................................................................................................ 20
ACKNOWLEDGEMENTS

I want to thank my advisor, Dr. Trent Biggs, for his continual guidance and support. I would also like to thank the City of San Diego, the San Dieguito Water District and the Santa Fe Irrigation District for providing the data that was used in this analysis. I want to acknowledge Lisa Thurn, who provided technical support while doing nutrient analyses on stormwater samples at the Ecology Analytical Lab at San Diego State University. This project was supported by Agriculture and Food Research Initiative Competitive Grant no. 2011-38422-31204 from the USDA National Institute of Food and Agriculture.
EXECUTIVE SUMMARY

Water quality in San Diego drinking water reservoirs is compromised by seasonal algal production, imposing health and financial burdens on the public. This study examines thermal, oxygen and nutrient dynamics in four drinking water reservoirs—Barrett, Hodges, Morena and Sutherland—to elucidate the physical processes leading to exceptionally severe anoxia observed at some of the sites (Lake Barrett and Lake Hodges). All of the reservoirs receive runoff from the watershed and no imported water from the Colorado River and Sacramento River. There is a range of agricultural activity (extremely high to none) and urban development/impervious surface over the catchment, which might contribute to eutrophication downstream. Though, data suggest that internal cycling of autochthonous phosphorus can be a significant source of nutrient loading. Also, lake water volume is a significant control on the length and strength of summer stratification and extent of anoxia.
PROJECT OBJECTIVES

Introduction

Water quality in drinking water reservoirs, including in San Diego, is adversely affected by vernal and summertime algal blooms. The blooms can cause hypoxia and health, odor and taste problems, in spite of water treatment (Falconer 1999; Izaguirre et al., 1999; Paerl et al., 2001). Copious algal production leads to high levels of total organic carbon (TOC) in the influent at water treatment facilities. High TOC results in an upper limit to the amount of chemical treatment that can be applied because disinfection is both expensive and introduces harmful disinfection byproduct (DBP) contaminants directly proportional to the presence of precursor organic matter (Camel and Bermond 1998; Richardson 2003; Singer 1994; Symons et al., 1975). The levels of DBPs must not surpass federal standards that were set to protect human health (U.S. EPA 2006).

High inputs of nitrogen (N) and phosphorus (P) are the primary catalysts for algal growth and TOC production. San Diego purchases a blend of imported water, which, historically, accounts on average for ~80% of San Diego’s annual water supply (CSDPUWW 2011). A majority of the imported water is sourced from the Colorado River (high in N and Total Dissolved Solids (TDS)) and the balance comes from the Sacramento River (high in P, TOC, and bromide). The remaining ~20% of drinking water comes from runoff in local watersheds, which can be a significant source of nutrient loading. Agricultural areas leach excess nutrients from fertilizer application (Li et al., 2011; Sheeder and Evans, 2004), which leads to increased algal production in water bodies downstream (Savage et al., 2010). Dairy farms, which produce cattle manure, have been shown to increase P concentrations in stormwater runoff (McFarland and Hauck, 1999). Urbanization introduces fertilizer from residential lawns (Kelling and Peterson, 1975)—which, in some American cities, are growing bigger in proportion to the lot size of new homes (Robbins and Birkenholtz, 2003)—and golf courses (King et al., 2007). Treated wastewater effluent enter streams (Bowen and Valiela, 2001; Kalscheur et al., 2012; McLaughlin et al., 2006).

Autochthonous nutrients, particularly P, which are generated internally, can be a more serious threat to water quality than allochthonous nutrients, as excess nutrient inputs accumulate in a water body over time. Internal cycling occurs when a legacy of nutrient loading is coupled with hypoxic conditions in the hypolimnion and sediment. Under anoxic conditions, ferric ions of iron reduce to the ferrous form, which leads to the dissolution of the P and iron (Fe) bond and causes sediments to release soluble reactive phosphorus (SRP) and Fe, as well as ammonia (NH₃) and manganese (Mn), to the hypolimnion (Beutel, et al., 2007; Correll 1998). Warmer water temperatures have also been shown to increase the release of P from sediment (Duan et al., 2012). The liberated P further enhances algal growth and decay in the euphotic zone upon entrainment (Effler et al., 1986), vertical eddy diffusion (Stauffer 1987) or mixis (Cooke et al., 1993). Algae eventually sink into the hypolimnion and sediment, feeding microbial bacteria that exert high oxygen demand. This positive feedback further degrades water quality and reinforces a cycle of nutrient supply, production of labile organic matter and DO depletion.
In addition to nutrient loading, the development of hypoxia in a water body also depends on the onset and duration of summer thermal stratification. Warm summer temperatures cause the formation of a density gradient in the vertical water column and the separation of warm surface waters from cold bottom waters. When a lake is stratified, aeration from the atmosphere or plant respiration cannot replenish the hypolimnion because oxygen cannot penetrate the dense upper boundary of the hypolimnion. Usually, this stratified state persists until autumn, when the temperature in the water column homogenizes and the water recirculates. The risk of anoxia depends on the concentration of dissolved oxygen (DO) in the hypolimnion at the onset of stratification, the duration of the stratified state and the oxygen demand by microbes in the water column and sediment.

The length and strength of summer thermal stratification are determined primarily by weather-related factors (Boehrer and Schultze, 2008), such as the penetration of solar radiation, sensible heat exchange with the atmosphere and wind mixing. A warming climate increases the potential for anoxia because it can move the onset of stratification earlier in the year, prolonging its duration (Foley et al., 2012; Minns et al., 2011; Robertson and Ragotzkie, 1990; Wang et al., 2012; Winder and Schindler, 2004); intensify vertical temperature gradients (Jankowski et al., 2006); and raise overall lake temperature (Arhonditsis et al., 2004), which is correlated with increasing microbial metabolism (Pettersson 1998). Wind can decrease anoxia by mechanically turning over the water to allow circulation or push oxygen across the thermocline and reduce Volumetric Hypolimnetic Oxygen Demand (VHOD; Foley et al., 2012), the volume of oxygen consumed by microbial bacteria in the hypolimnion and sediment.

Management of a reservoir can also affect anoxia, as a reservoir functions differently from a natural lake. Hydraulic conditions can increase eutrophication irrespective of external nutrient addition (Zohary and Ostrovsky, 2011). Shallow reservoirs generally have shorter water retention times and also experience lower P retention due to more intense mixing (Straskraba et al., 1995). Additionally, when the water level in a reservoir becomes extremely low, riparian vegetation may spread, causing biomass to accumulate (Coops et al., 2003). Subsequent die-off of the vegetation upon refilling the reservoir can release a shock of nutrient and carbon input to the lake and enhance anoxia (Geraldes and Boavida, 2003). High amplitude of seasonal and multi-annual water level fluctuation from drinking water drawdown during summer drought in a Mediterranean climate can prematurely destroy hydrodynamic stability and send nutrient-rich water from the hypolimnion to the epilimnion during the algal growing season, rather than later in the autumn (Naselli-Flores 2003; Nowlin et al., 2004). The choice of the location of the dam outlet for withdrawal of water can also impact stability. Drawing cold hypolimnetic water leaves behind a greater proportion of warm epilimnetic water, which also disturbs the stability of the thermocline (Ma et al., 2008) and transports bottom nutrients to be entrained in the epilimnion. Drawing water from the epilimnion can counteract the effects of climate warming, while drawing from the hypolimnion can exacerbate them (Moreno-Ostos et al., 2008).

**Objectives**

This project investigates the relationships among lake warming and summer thermal stratification, nutrient input from the watershed and oxygen depletion in the hypolimnion to
understand the possible effects of changing land use on water quality downstream. The focus of the project was expanded from inputs of nutrients to the study reservoirs from their contributing watersheds to include thermal and oxygen dynamics inside the reservoirs because of the significant contributions of nutrients that are likely occurring from internal cycling. Four drinking water reservoirs—Barrett, Hodges, Morena and Sutherland (Figure 1)—were selected for the project because they are similar insofar as they are in relatively close proximity to each other, located in a semi-arid region and receive only surface runoff and no imported water. They have different amounts of land use in their watersheds. Lakes Morena and Sutherland were added to the project and Miramar was removed from the original proposal due to the availability of data. During the project special attention was paid to Lake Hodges, where water quality was reported to be worse.

![Figure 1. Map of study reservoirs and the land uses in their contributing watersheds.](image)

Temperature profiles were created for each of the study reservoirs over a ~20 year period to determine the timing and intensity of the summer stratification. The original proposal called for the creation of dissolved oxygen profiles, which would be used to determine rates of oxygen depletion in the hypolimnia. However, due to the chronically anoxic conditions observed in some of the reservoirs, particularly Barrett and Hodges, oxygen profiles were used to calculate the
Anoxic Factor (Nurnberg 1995) instead. The Anoxic Factor is a metric that is better suited to make comparisons of oxic conditions across reservoirs that experience anoxia. Transparency (Secchi depth) and concentrations of chlorophyll a were additional measures of water quality used in the project.

Analyses of oxygen isotopes were originally proposed, but, due to prohibitive costs, they were not done. Instead, financial resources were devoted to analyzing water samples taken during several storms to start deriving estimates of nutrient inflow. Nutrients were modeled only at Lake Hodges because it is presumably the highest recipient of nutrients through runoff, due to the higher presence of agriculture and urbanization in its watershed. Land use maps of the watersheds were created for 1986, 1990, 1995, 2000, 2004, 2008, 2009 and 2050 (a projection of future land use) using data from the San Diego Association of Governments (SANDAG) Regional GIS Data Warehouse database (SANDAG 2012).

This project has helped prepare me for a career at the USDA because it required me to design a project and develop a budget. Over the course of executing the project, I have read the literature on summer stratification and the development of anoxia in lakes; gained more proficiency in analyzing large datasets using GIS applications, computer programming (R Statistics package) and doing statistical analyses; developed skills in the field, collecting stormflow samples at sporadic and opportune moments; made a model of nutrient inflow to Lake Hodges; and started developing a model of Lake Hodges with CE-QUAL-W2 (Cole and Wells, 2011), a 2-D dynamic lake model, which simulates thermal and oxygen dynamics in lakes. This project has helped me craft skills to do high-quality research on problems facing natural, agricultural and urban watersheds.

PROJECT APPROACH

Data

At Lakes Hodges and Sutherland, water quality parameters, including temperature, DO, pH, specific conductivity and TDS, were sampled by the City of San Diego throughout the water column in the lacustrine zone at approximately 1 m depth intervals, monthly, from January 1992 to December 2011. At Lakes Barrett and Morena, water quality sampling at approximately 1 m depth intervals occurred biweekly, and weekly in some cases, from June 1989 to February 1993, then monthly until August 2012. Sampling for chlorophyll a in the lake profile began in July 2003 at Barrett and in May 2004 at the other reservoirs. Where more than one sample was collected per month, data from the first sample event of each month was kept for the analysis in order to be consistent among the reservoirs. A comprehensive description of the data used in the project is in Table 1.
<table>
<thead>
<tr>
<th>Reservoir</th>
<th>Parameters</th>
<th>Sampling Location</th>
<th>Date Range</th>
<th>Temporal Resolution</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>Barrett</td>
<td>Temperature, DO, pH, specific conductivity, TDS</td>
<td>Lake profile</td>
<td>June 1989 – August 2012</td>
<td>Monthly</td>
<td>County of San Diego</td>
</tr>
<tr>
<td></td>
<td>Chlorophyll a</td>
<td>Lake profile</td>
<td>July 2003 – August 2012</td>
<td>Monthly</td>
<td>County of San Diego</td>
</tr>
<tr>
<td></td>
<td>TN, TP</td>
<td>Lake surface</td>
<td>April 2003 – July 2012</td>
<td>Monthly</td>
<td>County of San Diego</td>
</tr>
<tr>
<td></td>
<td>Maximum air temperature, minimum air temperature, total precipitation, mean windspeed</td>
<td>Alpine Station (32.83333, -116.783)</td>
<td>June 1989-August 2012</td>
<td>Daily</td>
<td>NCDC</td>
</tr>
<tr>
<td>Hodges</td>
<td>Temperature, DO, pH, specific conductivity, TDS</td>
<td>Lake profile</td>
<td>January 1992 – December 2011</td>
<td>Monthly</td>
<td>County of San Diego</td>
</tr>
<tr>
<td></td>
<td>Chlorophyll a</td>
<td>Lake profile</td>
<td>May 2004 – December 2011</td>
<td>Monthly</td>
<td>County of San Diego</td>
</tr>
<tr>
<td></td>
<td>NH₃, NO₂, NO₃, NO₂ plus NO₃, TN, and TP</td>
<td>Lake surface, water column (reservoir gage 75), lake bottom</td>
<td>March 2008 – April 2012</td>
<td>Weekly</td>
<td>County of San Diego</td>
</tr>
<tr>
<td></td>
<td>NH₃, NO₃, Reactive P, TP, Mn, DO, pH, Chlorophyll a, Temperature</td>
<td>Effluent</td>
<td>March 2008 – November 2011</td>
<td>Weekly</td>
<td>SFID</td>
</tr>
<tr>
<td></td>
<td>NH₃, NO₂, NO₃, NO₂ plus NO₃, TN, and TP</td>
<td>Various contributing streams (Figure 1)</td>
<td>January 2004 – March 2012</td>
<td>Monthly</td>
<td>County of San Diego</td>
</tr>
<tr>
<td></td>
<td>Maximum air temperature, minimum air temperature, total precipitation, wind speed, wind direction, solar radiation</td>
<td>Escondido/Escondido SPV Station (33.081111, -116.975833)</td>
<td>January 1992 – December 2011</td>
<td>Daily</td>
<td>CIMIS</td>
</tr>
<tr>
<td></td>
<td>Mean temperature, wind speed, wind direction, dewpoint</td>
<td>Montgomery Field Station (32.81583, -117.13944)</td>
<td>January 1992 – December 2011</td>
<td>Hourly</td>
<td>NCDC</td>
</tr>
<tr>
<td></td>
<td>Cloud cover</td>
<td>Carlsbad Station (33.133, -117.283)</td>
<td>December 1994 – December 2009</td>
<td>Hourly</td>
<td>BASINS</td>
</tr>
<tr>
<td>Inflow</td>
<td></td>
<td>Kit Carson Creek (33.08854, -117.064)</td>
<td>October 2012 – March 2013</td>
<td>5 minutes</td>
<td>Weston Solutions</td>
</tr>
<tr>
<td></td>
<td></td>
<td>San Dieguito River (33.06249, -117.031)</td>
<td>September 2010 – June 2011</td>
<td>15 minutes</td>
<td>Weston Solutions</td>
</tr>
<tr>
<td>Streamflow</td>
<td></td>
<td>Guejito Creek (33.115833, -116.952222)</td>
<td>October 2004 – March 2013</td>
<td>Daily</td>
<td>USGS</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Santa Maria Creek (33.052222, -116.944722)</td>
<td>January 1989 – March 2013</td>
<td>Daily</td>
<td>USGS</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Santa Ysabel Creek (33.106944, -116.865278)</td>
<td>January 1989 – March 2013</td>
<td>Daily</td>
<td>USGS</td>
</tr>
<tr>
<td>Morena</td>
<td>Temperature, DO, pH, specific conductivity, TDS</td>
<td>Lake profile</td>
<td>June 1989 – August 2012</td>
<td>Monthly</td>
<td>County of San Diego</td>
</tr>
<tr>
<td></td>
<td>Chlorophyll a</td>
<td>Lake profile</td>
<td>May 2004 – August 2012</td>
<td>Monthly</td>
<td>County of San Diego</td>
</tr>
<tr>
<td></td>
<td>TN, TP</td>
<td>Lake surface</td>
<td>April 2003 – July 2012</td>
<td>Monthly</td>
<td>County of San Diego</td>
</tr>
<tr>
<td></td>
<td>Maximum air temperature, minimum air temperature, total precipitation, mean windspeed</td>
<td>Campo Station (32.61667, -116.467)</td>
<td>June 1989 – August 2012</td>
<td>Daily</td>
<td>NCDC</td>
</tr>
<tr>
<td>Sutherland</td>
<td>Temperature, DO, pH, specific conductivity, TDS</td>
<td>Lake profile</td>
<td>January 1992 – December 2011</td>
<td>Monthly</td>
<td>County of San Diego</td>
</tr>
<tr>
<td></td>
<td>Chlorophyll a</td>
<td>Lake profile</td>
<td>May 2004 – December 2011</td>
<td>Monthly</td>
<td>County of San Diego</td>
</tr>
<tr>
<td></td>
<td>NO₃, NO₂, NO₃, NO₂ plus NO₃, TN, and TP</td>
<td>Lake surface, lake bottom</td>
<td>June 2003 – December 2004</td>
<td>Monthly</td>
<td>County of San Diego</td>
</tr>
<tr>
<td></td>
<td>Maximum air temperature, minimum air temperature, total precipitation, mean windspeed</td>
<td>Ramona Station (33.05, -116.867)</td>
<td>January 1992 – December 2011</td>
<td>Daily</td>
<td>NCDC</td>
</tr>
</tbody>
</table>
Land use

Lake Hodges (33.045, -117.128611) is located approximately 50 km north of the city of San Diego and 20 km inland from the coast at an altitude of 67 m a.s.l. The watershed area to lake volume ratio is high in Lake Hodges, compared to the other study reservoirs, which indicates that activities in the watershed may have a significant influence on the quality of surface runoff and water in the receiving lake. There is a considerable proportion of developed land in Hodges watershed, compared to the other study watersheds. Hodges watershed has large urban and agricultural areas (9.8 and 14.3% of the watershed area, respectively; Table 2). Agricultural land use is separated into two categories: extensive (cropland, orchards and vineyards) and intensive (poultry ranches and dairy farms). The area under agricultural land use is projected to decrease from 14.3 to 4.9% by 2050, while urban land use is projected to increase from 9.8 to 11.4%, a higher proportion of urban land use than in the other three study watersheds. Rural residential land use, which may include orchards or fields on the property, and the presence of managed parks (zoos, golf courses and cemeteries) are also disproportionately high, compared to the other watersheds. Currently, Barrett, Morena and Sutherland reservoirs are in sparsely urbanized watersheds (1.7, 2.5 and 0.6%, respectively). There is significantly less agriculture in Barrett and Morena watersheds (1.1 and 2.2%, respectively), but more in Sutherland (19.2%).

Table 2. Characteristics of the watersheds of the study reservoirs: watershed area; watershed area: lake volume; land use in watershed; impervious surface.

<table>
<thead>
<tr>
<th>Reservoir</th>
<th>Watershed area (km²)</th>
<th>Watershed area: lake volume (km²)</th>
<th>Land use in watershed (%)</th>
<th>Impervious surface (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Barrett</td>
<td>339</td>
<td>12.93</td>
<td>Agriculture (extensive): 1.07, Agriculture (intensive): 0</td>
<td>94.11 1.67 1.09 8.1</td>
</tr>
<tr>
<td>Hodges</td>
<td>641</td>
<td>26.53</td>
<td>Managed Park: 1.98, Rural Residential: 1.13, Undeveloped: 12.80</td>
<td>61.39 9.76 0.64 11.6</td>
</tr>
<tr>
<td>Morena</td>
<td>294</td>
<td>15.13</td>
<td>Rural Residential: 2.22, Undeveloped: 0</td>
<td>89.06 2.47 2.15 8.8</td>
</tr>
<tr>
<td>Sutherland</td>
<td>140</td>
<td>9.34</td>
<td>Undeveloped: 0, Urban: 0, Water: 19.21</td>
<td>76.39 0.61 1.55 7.7</td>
</tr>
</tbody>
</table>

Sutherland Reservoir (33.117856, -116.786642) is the smallest of the reservoirs, located about 72 km northeast of San Diego, upstream of Lake Hodges. Though Hodges actually drains the Sutherland watershed, water that is collected in Sutherland watershed is impounded in Sutherland Reservoir, from where it is transferred to San Vicente Reservoir, keeping Sutherland watershed hydrologically isolated from Hodges watershed. Barrett Reservoir (32.679254, -116.670492) holds the largest volume of water in the study and is located at the confluence of Cottonwood and Pine Valley Creeks, about 56 km east of San Diego. It receives transfers of water from Morena Reservoir (32.685688, -116.547287), located 24 km to the east, the highest and most remote of all the reservoirs in the San Diego reservoir system.

Nutrient Modeling

Grab sampling was done during four storm events throughout the wet season (December 2012—March 2013) at three inflow sites (Figure 2) to Lake Hodges, including Green Valley Creek, Kit Carson Creek and San Dieguito River. Samples (1L) were collected in duplicate throughout each
event to capture the rising limb, peak and falling limb. The samples were stored in bottles at 4°C and transported to the Ecology Analytical Lab at San Diego State University (SDSU), where they were filtered (0.45 um) and analyzed according to the Kjeldahl method for concentrations of NH$_4$, NO$_3$, total Kjeldahl Nitrogen (TKN) and TP (US EPA 1992).

Discharge was measured by Weston Solutions at three inflows to Lake Hodges, including San Dieguito River (every 15 minutes from September 2010 to June 2011), a tributary to Kit Carson Creek (every 5 minutes from October 2012 to March 2013) and Green Valley Creek (every minute from August 2010 to September 2010, then every 5 minutes until February 2013). The data was then aggregated by the author into daily mean flow. A variety of technologies, including ultrasonic sensors, bubblers and submerged pressure transducers, measured stream stage and sent that information to an American Sigma flow meter, which calculated flow rates according to a preprogrammed discharge equation. These equations were made initially by field crews that measured stream profiles and discharge with a hand held flow meter.

Regression relationships were established between the streamflows at Green Valley Creek and USGS stream gages, for which exist an extensive dataset, in the watershed, upstream of Lake Hodges. The streamflow model derived for the Green Valley Creek subwatershed was applied to the subwatersheds of Felicita and Kit Carson Creeks as a ratio based on their watershed areas. This is justified because of the similarities in land use, geographical proximity and lack of data to
produce independent flow models. Inflow data is available for Kit Carson Creek but, due to the insufficient number of data points, an independent model could not be generated. Positive streamflow in the San Dieguito River is highly irregular because of irrigation pumping in the San Pasqual Valley, which reduces average annual runoff by 8,000 to 10,000 acre ft per year (AFY; CSDPUWW 2011), so it will not be modeled, but calculated as the difference between the monthly lake volume record and the sum of the modeled inflows.

Linear and logarithmic regressions were calculated with the y-intercept constrained to 0 for observed flow in 1) Guejito Creek; 2) Santa Maria Creek; 3) Santa Ysabel Creek; and 4) the sum of flow in Santa Maria and Santa Ysabel Creeks against observed flow in Green Valley Creek. The strongest relationship was linear, between Guejito Creek, the streamflow gaging station nearest to Hodges, and Green Valley Creek (adj. R² = 0.8; p-value < 0.001; RMSE = 3.34; Figure 3 and Figure 4). However, there is no discharge data available before October 1, 2004 at Guejito Creek, so an alternative model will be used for that time. The next best correlation was also linear, between the sum of the flows from Santa Maria Creek and Santa Ysabel Creek and Green Valley Creek (adj. R² = 0.72; p-value < 0.001; RMSE = 3.93; Figure 3 and Figure 4). The two regressions produced the following equations:

\[
\hat{Q}_{GVC1} = 0.249424 \times Q_G 
\]  

(1)

where \(\hat{Q}_{GVC1}\) is the estimated daily average flow (cfs) in Green Valley Creek, including and after October 1, 2004, and \(Q_G\) is the observed daily average flow (cfs) in Guejito Creek;

\[
\hat{Q}_{GVC2} = 0.047306 \times Q_{SM+SY} 
\]  

(2)

where \(\hat{Q}_{GVC2}\) is the estimated daily average flow (cfs) in Green Valley Creek, before October 1, 2004, and \(Q_{SM+SY}\) is the sum of the observed daily average flow (cfs) in Santa Maria and Santa Ysabel Creeks;

\[
\hat{Q}_{KCC} = 0.9282 \times Q_{GVC} 
\]  

(3)

where \(\hat{Q}_{KCC}\) is the estimated daily average flow (cfs) in Kit Carson Creek and \(Q_{GVC}\) is the modeled/observed daily average flow (cfs) in Green Valley Creek;

\[
\hat{Q}_{FEL} = 0.3199 \times Q_{GVC} 
\]  

(4)

where \(\hat{Q}_{FEL}\) is the estimated daily average flow (cfs) in Felicita Creek and \(Q_{GVC}\) is the modeled/observed daily average flow (cfs) in Green Valley Creek.
Figure 3. Regressions of predicted and observed flows at Green Valley Creek. The dashed line represents a 1:1 relationship.

Figure 4. A timeseries showing the performance of predicted flows against observed flows.
Annual nutrient concentration rating curves will be developed using the flow models and observed nutrient concentrations when samples were taken. Nutrient concentrations will be back-calculated for the periods for which there are flow data, but not nutrient data. The relationship between flow rates and nutrient concentration, a power function, will be fitted on a log-log plot:

$$C_t = aQ_t^b$$  \hspace{1cm} (5)

where $C_t$ is predicted nutrient concentration (mg L$^{-1}$) at time $t$, $Q_t$ is streamflow (cfs) at time $t$ and $a$ and $b$ are regression coefficients.

Annual load will be calculated as:

$$\text{Annual TP Load} = \sum_{t=1}^{n} Q_t \times C_t \times 2.4466$$  \hspace{1cm} (6)

where $Q_t$ is streamflow (cfs) at time $t$, $C_t$ is predicted nutrient concentration (mg L$^{-1}$) at time $t$ and 2.4466 is the conversion factor from cfs and mg L$^{-1}$ to kg day$^{-1}$. Instantaneous load is derived by multiplying $Q_t$, $C_t$ and 2.4466; daily load is the summation of instantaneous loads over the day; annual load is the summation of daily loads over 365 days.

**Analysis**

**Temperature and stability**

Volume-weighted water temperature was calculated for the epilimnion, hypolimnion and the whole lake for each observation of water temperature using:

$$T_e \text{ or } T_h \text{ or } T_w = \frac{1}{V_w} \sum_{z=1}^{n} t_z V_z$$  \hspace{1cm} (7)

where $T_e$ is the temperature of the epilimnion, $T_h$ is the temperature of the hypolimnion, $T_w$ is the temperature of the whole lake, $V_w$ is the volume of the layer of interest (epilimnion, hypolimnion or whole lake), $t_z$ is the temperature of a depth interval in that layer, $V_z$ is the volume of the depth interval and $n$ is the number of depth intervals. The summation was taken over all depths at 1 m intervals from the initial depth of the layer of interest to the bottom of that layer.

The strength of thermal stratification in the reservoir is expressed in terms of Schmidt Stability Index, $S$, J m$^{-2}$ (Hutchinson 1957), which indicates the minimum amount of work that must be done by wind to return a lake to an isothermal state. $S$ was calculated from differences in water density in the water column using equation (8) in Idso (1973):

$$S = \frac{g}{A_0} \int_{z_0}^{z_m} (z - z^*)(p_z - p^*)A_z \, dz$$  \hspace{1cm} (8)
where \( g \) is acceleration due to gravity, \( A_0 \) is surface area of the lake, \( A_z \) is lake area at depth \( z \), \( p_z \) is water density as calculated from the temperature at depth \( z \), \( p^* \) is the lake’s mean water density, and \( z^* \) is the depth where the mean water density occurs. The summation was done for all depths \( (z) \) at an interval \( (dz) \) of approximately 1 m from the surface \( (z_0) \) to the maximum depth \( (z_m) \).

Water density was calculated as a function of temperature using the Thiessen-Scheel-Diesselhorst Equation (Maidment 1993):

\[
p = 1000 \left[ 1 - \frac{t + 288.94}{508929.2(t + 68.12963)} (t - 3.9863)^2 \right] \tag{9}
\]

where \( p \) is water density \( (\text{kg m}^{-3}) \) and \( t \) is temperature \( (^\circ\text{C}) \).

The date of stratification onset was considered as the day of the year when Schmidt stability in the water column reached a minimum threshold of 100 J m\(^{-2}\) and the date of fall turnover was the day of the year when Schmidt stability fell below 100 J m\(^{-2}\). This threshold was selected because it generally correlates with the development of a thermocline, defined as a minimum temperature gradient of 1° C in the water column (Wetzel 2001). The estimated days of the start and end of stratification did not usually coincide with an observation of lake water temperature, so piecewise linear interpolation between observation dates and general rounding rules were used to determine them.

**Anoxia**

Hypolimnetic oxygen is expressed as the Anoxic Factor, the number of days in a year for which an area equivalent to the surface area of a lake is anoxic. Anoxic water is defined as having an oxygen concentration of 1.5 mg L\(^{-1}\) or less (Nurnberg 1995). AF was calculated for each year with observations \( n \) as:

\[
AF = \sum_{i=1}^{n} \frac{t_ia_i}{A_{0i}} \tag{10}
\]

where the duration of anoxia \( (t_i \text{ in days}) \) is multiplied by the shallowest anoxic plane \( (a_i \text{ in m}^2) \) and divided by the lake surface area \( (A_{0i}, \text{m}^2) \) for period \( i \).

AF was chosen as a measure of anoxia because it accounts for the high fluctuations in water volume that occurs in the study reservoirs. Comparison of AF is a more suitable approach than using oxygen depletion curves to evaluate anoxia in the hypolimnia because oxygen depletion curves determine only the speed of developing anoxia but provide no further information about the anoxia after that (Nurnberg 2004). The hypolimnion in Lake Hodges was anoxic for most of the study period, which makes determination of the rate of oxygen depletion and interannual comparisons difficult.
**Statistical Analysis**

Monotonic trends over time in variables related to water quality, including transparency and AF, were determined using simple linear regression. The Shapiro-Wilk test was used to check data for normality and the Mann-Kendall test was applied to non-parametric data. Trends were tested for statistical significance at $\alpha = 0.05$. The predictive capacity of independent variables, such as mean annual lake volume, mean annual stability and air temperature, on dependent variables related to water quality were also tested using linear regression.

**PROJECT OUTCOMES**

**Lake volume and lake thermal dynamics**

Lake thermal behavior is generally regular and similar in all four study reservoirs over the study period, with few monotonic trends observed in lake thermal properties. There were no trends of increasing or decreasing volume-weighted average temperatures in either the epilimnion, hypolimnion or the entire lake at any of the study sites, with the exception of a cooling trend (adj. $R^2 = 0.21$; p-value = 0.04) in the epilimnion of Lake Hodges. This may be explained by a corresponding cooling trend (adj. $R^2 = 0.37$; p-value = 0.003) in the mean annual night-time air temperature (minimum air temperature) at Lake Hodges—which also occurred at Lakes Barrett and Sutherland—over the study period. The temperature of the epilimnion in Hodges is more strongly correlated with minimum than maximum air temperature. There were no trends in the mean annual Schmidt stability or max annual stability. The duration of stratification decreased only at Morena (Mann-Kendall p-value < 0.001), where the start of the stratification season shifted later into the year by an average of 5.9 days year$^{-1}$ (adj. $R^2 = 0.8$; p-value < 0.001). It should be noted that Morena stratified in only 13 years over a study period of 23 years. There was a shallowing of the depth of the thermocline at Lake Hodges (Mann-Kendall p-value = 0.04).

Lake volume is highly coupled with lake thermal properties (Figure 5) at all of the study reservoirs because it is a major control on the depth of solar penetration and heat storage. One of the primary reasons that monotonic changes in thermal properties were not detected at the reservoirs was that large fluctuations in lake volume occurred throughout the study period and the temperatures in the lakes were constantly responding. Mean annual lake volume was a powerful explanatory variable, as it was negatively correlated with the volume-weighted average temperature of the lake at Hodges (Mann-Kendall p-value = 0.04) and Morena (adj. $R^2 = 0.22$; p-value = 0.01) and negatively correlated with the volume-weighted average temperature of the epilimnion at Barrett (adj. $R^2 = 0.26$; p-value = 0.01) and Sutherland (adj. $R^2 = 0.16$; p-value = 0.05). Mean annual lake volume was also positively related to the mean annual Schmidt stability (all Mann-Kendall p-values < 0.001) and the duration of summer stratification at all of the lakes (all Mann-Kendall p-values < 0.001). The mean annual depth of the thermocline at all of the lakes, except Morena, was positively correlated with mean annual lake volume (Barrett, adj. $R^2 = 0.22$, p-value = 0.01; Hodges, Mann-Kendall p-value = 0.015; Sutherland, adj. $R^2 = 0.4$, p-
value = 0.002). A residual analysis shows that the decrease in the duration of stratification at Morena can be accounted for by holding mean annual lake volume constant.

Figure 5. Timeseries of lake volume and stability. The threshold for stratification is a Schmidt stability of 100 Jm$^{-2}$.
Anoxic Factor

A monotonic increase in the Anoxic Factor was detected in two reservoirs, Barrett and Hodges (Figure 6). The average rate of the increase at Barrett (adj. $R^2 = 0.3$; $p$-value = 0.005) is 3.03 days year$^{-1}$. A linear regression trendline at Hodges was not significant because the data is non-parametric (Shapiro-Wilk $p$-value = 0.001). This is due to the severely low water volumes observed in 2002, 2003 and 2004, when the water column remained isothermal and inhibited stratification and the development of anoxia. The Mann-Kendall test confirmed the significance ($p$-value = 0.03) of the increasing trend in the Anoxic Factor over the study period. The Anoxic Factor was similarly influenced by low lake volumes at Morena and Sutherland.

![Timeseries of the Anoxic Factor](image)

Figure 6. Timeseries of the Anoxic Factor. Data without significant trends are in grey.

There were a few trends that could account for the increase in anoxia at Barrett and Hodges, though they have very different land uses in their watersheds. At Barrett, there was a negative correlation between anoxia and volume-weighted mean lake temperature (adj. $R^2 = 0.21$; $p$-value = 0.02) and transparency (adj. $R^2 = 0.4$; $p$-value < 0.001). Transparency data in Barrett is non-parametric (Shapiro-Wilk $p$-value = 0.002), but the Mann-Kendall test ($p$-value = 0.003) indicates that transparency is decreasing over time. The increasing turbidity could be due to the lake being more productive. This is evidence that the development of anoxia at Barrett can be particularly sensitive to nutrient loading and algal production. The increasing trend in the Anoxic Factor disappears when transparency is held constant in a residual analysis.

At Hodges, there was no significant relationship between anoxia and transparency or lake temperatures, which are indicators of biological activity. Nutrient addition may not impact
oxygen levels as much as other factors. There was a negative correlation between anoxia and the depth of the thermocline (adj. $R^2 = 0.2; p$-value $= 0.04$)—a shallower thermocline increases the risk of anoxia. The increasing trend in the Anoxic Factor disappears when the depth of the thermocline is held constant. The thermocline, controlled by lake volume, at Lake Hodges is shallowing by 7.1 cm year$^{-1}$ (adj. $R^2 = 0.29; p$-value $= 0.01$) over the study period. There is a concomitant rise of the upper boundary of the hypolimnion (adj. $R^2 = 0.17; p$-value $= 0.057$) and anoxic depth (used in the Anoxic Factor equation; adj. $R^2 = 0.13; p$-value $= 0.062$), though $p$-values were above 0.05. This leads to a larger proportion of the lake to be anoxic. The anoxic depth is negatively correlated with the anoxic factor (Mann Kendall $p$-value $= 0.006$). When holding mean annual lake volume constant, the mean annual depth of the thermocline still declines by an average of 5.3 cm year$^{-1}$ (adj. $R^2 = 0.22; p$-value $= 0.03$). It is unclear what is causing the depth of the thermocline to decrease. It could be related to the decline in night-time temperatures and the corresponding decline in the temperature of the epilimnion.

CONCLUSIONS

This project investigates the oxic conditions, a measure of water quality, in four drinking water reservoirs that receive runoff from their local watersheds. Anoxia was observed to increase at Lakes Barrett and Hodges. The depth of the thermocline and lake turbidity were significantly correlated with anoxic conditions and are the subject of future research. There were few monotonic trends in thermal properties, such as volume-weighted average temperatures, the mean annual Schmidt stability and the duration of the summer stratification season. Lake volume was confirmed to be a significant control on thermal dynamics in all of the reservoirs. The original emphasis was placed on nutrient input to the lakes and a nutrient model was developed, though it was not used in this project. It will be used in future research in conjunction with a lake model of Lake Hodges, developed using CE-QUAL-W2, to simulate the effects of changing land use and nutrient export in its watershed. Additionally, changes in land use over time in the study watersheds will be correlated with changes in water quality in the study reservoirs.

This project has given me the opportunity to research biogeochemical processes in watersheds of varying degrees of development and receiving lakes downstream. It has led to my proficiency in computer programming and statistical analyses, and has made possible field trips to study sites and the collection of water samples during storm events. This experiential learning has assured me that I have chosen a field, watershed science, in which I want to have a career and has given me a foundation for my prospective career as a research scientist with the USDA.
REFERENCES


Falconer, I. R. (1999). An overview of problems caused by toxic blue-green algae (cyanobacteria) in drinking and recreational water. *Environmental Toxicology, 14*(1), 5-12. doi: 10.1002/(sici)1522-7278(199902)14:1<5::aid-tox3>3.3.co;2-s


