NUTRIENT CONTRIBUTION OF THE SHALLOW UNCONFINED AQUIFER TO PINEVIEW RESERVOIR

by

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ABSTRACT

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Major Professor: Dr. Darwin L. Sorensen
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Pineview Reservoir, near Utah's populous Wasatch Front, could play an important role in modulating water supply as water demands and water uses change in response to increasing population densities. The reservoir is currently mesotrophic but threatens to become eutrophic. Ground water in the shallow water table aquifer that surrounds the reservoir contributes a large proportion of the reservoir's inflows in summer and fall because most of the stream flow is diverted for irrigation. Ground water flow and its subsequent nutrient loading to the reservoir were studied from February 2010 through November 2011. The objectives were to: 1) characterize nutrient transport from the water table aquifer to the reservoir; 2) quantify and characterize the spatial variability of ground water flow and nutrient loading in a mountainous irrigated valley; and 3) estimate nitrate leaching to ground water from cropland, lawns and septic drain fields.

The first objective was achieved by monitoring stream flows, and modeling ground water flow and nutrient loading towards Pineview Reservoir. Ground water from the water table aquifer contributed 22 percent and 2.6 percent nitrate + nitrite nitrogen and total dissolved phosphorus, respectively, to the annual reservoir loads. The aquifer contributed a total inflow of 3.4 x 10^6 m^3 yr^{-1} (2 percent of the total inflows) to the reservoir. Large variations in both ground water nutrient concentrations (6 – 310 µg P L^{-1} as total dissolved phosphorus and 3.3 – 21 mg N L^{-1} as nitrate + nitrite) and ground water flows among aquifer subdivisions were observed.

Study of the second objective employed GIS-based interpolation techniques in analyzing the spatial distribution of ground water flow and nutrient loading towards the reservoir. Large spatial variations in ground water flows and nutrient loadings were observed. The 67 percent confidence intervals (geometric mean ± 1 standard deviation) for total dissolved phosphorus ranged from 0.014 - 0.400 kg P d^{-1}. Nitrate + nitrite nitrogen had a 67 percent confidence interval of 0.954 - 39.1 kg N d^{-1}. The variations were attributed to agricultural and domestic non-point sources.

Under the third objective, ground water nitrate loadings in the near-reservoir drainage area of the reservoir's major tributary, the South Fork of the Ogden River, were simulated in the GIS-based Nitrogen Loss and Environmental Assessment Package. Annual leaching rates (kg N ha^{-1} yr^{-1}) from drain-fields and the lawns were, respectively, more than 2.6- and 1.1-fold higher than the croplands. However, differences in the spatial extent of contributing sources resulted in 70- and 50-fold higher total leaching losses from croplands and lawns, respectively, than drain-fields.

The findings would help water managers, town planners, and stakeholders in their decisions relative to land use, water distribution and use to protect and/or improve water quality in the reservoir.

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Pineview Reservoir, near Utah's populous Wasatch Front, could play an important role in modulating water supply as water demands and water uses change in response to increasing population densities. The reservoir's water quality may decline if nitrogen and phosphorus additions to the reservoir are not controlled. Most of the water flowing into the reservoir in summer and fall is contributed by the shallow ground water. The quantity and quality of the shallow ground water to Pineview Reservoir were studied from February 2010 through November 2011. The objectives were to: 1) increase understanding of nitrogen and phosphorus transport from ground water to the reservoir; 2) understand the differences in ground water flows and transport of nitrogen and phosphorus from different locations in a mountainous irrigated valley; and 3) estimate the nitrate contributions of cropland, lawns, and onsite wastewater to ground water. Large variations in nitrogen and phosphorus transport from different locations and land uses were observed. This information will help water managers, town planners and water users to make informed decisions on how to protect or improve the reservoir's water quality.

DEDICATION

This dissertation is dedicated to my wife, children, parents, brothers, and sisters. You are special to me. Your moral, social, spiritual, and financial supports were not in vain. There were times when the going got tough but your encouraging words eased my life. I am greatly indebted to you all. May God bless and lead you!

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My parents, Mr. Josophat Nyanda and Mrs. Desiliya Nyanda, you are wonderful parents. You taught me to work hard, to be trustworthy, and to fear God. You sacrificed a lot of things including your comfort to provide financial support for my education. It is my prayer that the Almighty God should bless you abundantly. To my sisters and brothers, thank you for your encouragement, and financial and material support.

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Thomas Nyanda Reuben

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CHAPTER 1

INTRODUCTION

The contribution of ground water to nutrient loading in Pineview Reservoir in Ogden Valley, Utah, (Fig. 1) was studied from February 2010 through November 2011. Pineview Reservoir serves northern Utah communities with irrigation; recreation, including boating and fishing; hydroelectric power generation and summer municipal water supply. The reservoir is an impoundment of the Ogden River and has a capacity of approximately 140 million m^3 and a surface area of 1200 ha (Tetra Tech, 2002; UDEQ, 2006).

Pineview Reservoir receives drainage from surface and ground water present in Ogden Valley and the surrounding mountains. Water from the reservoir flows westward, down the Ogden River, joins the Weber River and flows into Great Salt Lake (Bozniak et al., 2001). The reservoir has been classified by the Utah Division of Water Quality (UDWQ) for provision of habitat and food chains for cold water aquatic life (Utah Department of Administrative Services, 2009). UDEQ (2006) reported that the reservoir only partially met the criteria for this beneficial use because of annual development of anoxic conditions in the cooler hypolimnion that forced fish and other animals into the warmer epilimnion. There had been an increasing concern on the presence of odor, taste and coloration of the reservoir water due to cyanobacterial and algal blooms. Decomposition of the dead biological material produced in the bloom demanded oxygen and probably contributed much to the recurrence of anoxia in succeeding summers.

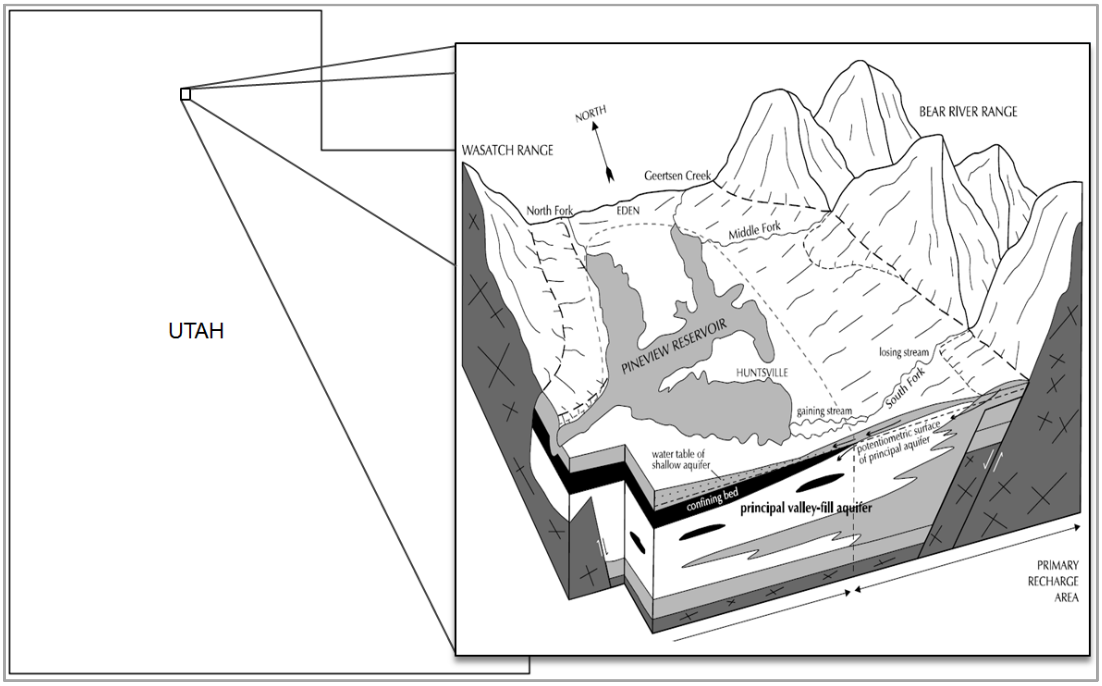

Figure 1. Pineview Reservoir location; and the hydrogeology of Ogden Valley, Utah (Snyder and Lowe, 1998)

Pineview Reservoir could play an important role in modulating water supply as water demands increase and water uses change in response to increasing population densities along Utah's Wasatch Front. It is therefore important for the water quality of the reservoir to be protected and, if possible, improved so that it may support both present and future desired uses. Understanding how natural and engineered systems could be used or managed to slow or reverse the eutrophication in Pineview Reservoir is essential.

Existing information about the geology, hydrology, and ground water and surface water quality of the Ogden Valley and Pineview Reservoir watershed helped inform the research reported here. Avery (1994) reviewed earlier hydro-geologic studies of Ogden Valley, described the valley's ground water hydrology and simulated ground water flows. The simulation of ground water flow rates into the reservoir was based on flows observed in three open-ended water barrels that were inserted in the sediments beneath Pineview Reservoir and flow observations from three streams namely the North, Middle, and South Forks of the Ogden River. His estimate of annual ground water discharge from the shallow unconfined aquifer to Pineview Reservoir was 33.3 million m^3 (27,000 ac-ft). There was uncertainty in the estimate due to sparse data.

The UDWQ had a more than 20 year history of monitoring water quality in streams that feed Pineview Reservoir and the data were available from the USEPA's STORET database. In addition, the Weber Basin Water Conservancy District (WBWCD) had been monitoring the reservoir and stream water quality for several years.

Pineview Reservoir has a Total Maximum Daily Load (TMDL) determination as required under section 303 (d) of the Federal Clean Water Act of 1972 (WPCF, 1987; USEPA, 2010). The TMDL study was completed and approved in 2002 (Tetra Tech, 2002). The study used an annual ground water flow of 24.7 million m^3 (20,000 ac-ft) to estimate the nutrient loading to Pineview Reservoir. The flow was estimated in 1988 from seeps coming out of some areas intermittently inundated by the reservoir (Weber Basin Water Quality Management Council, 1990). Nutrient loading computer simulations using SWAT (DiLuzio et al., 2001; Neitsch et al., 2001) and CE-QUAL-W2 (Chapra, 1997) models predicted that a reduction of 15 percent of the loads of both nitrogen and phosphorus would help meet the reservoir's beneficial use. Calibration of the CE-QUAL-W2 model was limited since only six sets of N and P data from the reservoir and tributaries (North, Middle, and South Forks of the Ogden River) were used yet the calibration normally requires extensive time series data (Tetra Tech, 2002). The ground water loadings of N and P to the reservoir were estimated using an estimated discharge of 24.7 million m^3 (Weber Basin Water Quality Management Council, 1990), a background concentration of 0.75 mg N L^{-1} (Wallace and Lowe, 1998), and an observed value of 20 μg P L^{-1} (Tetra Tech, 2002).

Lowe and Miner (1990) reported that there was an increase in septic tank absorption systems in the valley which may lead to increased nutrient loads in ground water. The nutrient contribution of these onsite wastewater treatment systems to Pineview Reservoir was not clearly known because of lack of data. Due to sparse data, the TMDL study for the reservoir estimated onsite wastewater system loads to the reservoir based on the number of people residing in the area and expected wastewater discharge and nutrient concentrations per household (Tetra Tech, 2002).

A study by the Utah Water Research Laboratory (UWRL) in collaboration with the Weber Basin Water Conservancy District (WBWCD) had shown that variation in nutrient sources influenced nutrient availability in the reservoir and internal nutrient loading provided additional nutrients (especially phosphorus) required for the algal blooms to occur. Internal nutrient loading was evidenced from an increase in phosphorus concentrations in the reservoir's hypolimnion prior to algal blooms in the year 2008 and 2009. Like in most years, the 2008 bloom occurred at the beginning of fall season.

Research motivation

Ground water contribution to the nutrients essential for primary production in Pineview Reservoir might be significant especially during the period when the blooms occurred. This was because the major streams ran almost dry at that time of the year due to irrigation water diversions. Avery (1994) reported that the major source of ground water inflows to Pineview Reservoir was the shallow unconfined aquifer. Uncertainty existed on what proportions of total inflows and nutrients to the reservoir were contributed by the shallow unconfined aquifer ground water. Avery (1994) and Tetra Tech (2002) recommended further surface and ground water studies in Ogden Valley in order to better understand their contributions to Pineview Reservoir. To our knowledge, there had not been any detailed studies on nutrient loading from the shallow unconfined aquifer to Pineview Reservoir. Need, therefore, existed to conduct detailed studies on the shallow unconfined aquifer ground water flows and their subsequent nutrient contributions towards primary production in the reservoir.

Research objectives

The overarching objective of the study was to expand the information available about the sources of nitrogen and phosphorus to Pineview Reservoir and to improve the ability to accurately describe how these nutrients were transported through the watershed to the reservoir. It was anticipated that this information would help water resource managers in their decisions relative to land use, water distribution and use (e.g., for irrigation), to protect and/or improve water quality in the reservoir. The study determined flow and nutrient loading contributions of the shallow unconfined aquifer to Pineview Reservoir, determined the extent to which the nutrients influenced the reservoir's primary production, and evaluated the significance of lawn and cropland fertilizers, and domestic wastewater to the loadings. Task-specific objectives were:

1. Evaluate the transport of nitrogen and phosphorus towards Pineview Reservoir through ground water in the shallow, unconfined aquifer by frequent monitoring of ground water flow rates, nitrate-nitrogen, total dissolved phosphorus and soluble reactive phosphorus in strategically located ground water monitoring wells;
2. Determine the spatial variation of ground water flow parameters and nutrient transport in an irrigated mountain valley;
3. Evaluate nitrogen loading to Pineview Reservoir from the shallow unconfined aquifer in the vicinity of the South Fork of the Ogden River by: 1) simulating N leaching from cropland, lawns and drain-fields using NLEAP-GIS 4.2; and 2) comparing observed ground water N concentrations with estimated N concentrations from the NLEAP simulation results.

Dissertation organization

Chapter 2 gives an overview of previous research work done on ground and surface water flows and nutrient loadings to Pineview Reservoir. The chapter outlines sources of nutrients to ground and surface water, and discusses pathways and pools on these nutrients in the environment. Chapter 2 also describes the impacts of nutrient loading to both ground and surface water. It also outlines procedures followed in determining ground and surface water nutrient concentrations and flows.

Chapter 3 presents research findings on the relative importance of ground water and surface water nutrient loadings to the reservoir. The chapter also discusses Pineview Reservoir's water quality attributes such as nutrients, chlorophyll *a* concentrations and trophic state.

Chapter 4 presents the spatial variability of ground water flow parameters and nutrient transport to the reservoir in an irrigated mountain valley. The chapter outlines GIS techniques that were applied to model ground water flow and nutrient loading to Pineview Reservoir from the water table aquifer in Ogden Valley, Utah where land use is predominantly irrigated agriculture.

Chapter 5 presents a report on NLEAP-GIS 4.2 simulations that were conducted to estimate nitrate leaching to ground water from cropland, lawns and septic system effluent drain fields. The chapter also presents comparisons of observed ground water nitrate concentrations and soil residual nitrate with those estimated from the simulation results. Simulation results for nitrate runoff, denitrification including nitrous oxide (N_2O) emissions, and ammonia volatilization are also reported in Chapter Five5. Chapter 6 outlines the summary, conclusions, and recommendations drawn from all of the dissertation research.

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CHAPTER 2

LITERATURE REVIEW

Abstract

Pineview Reservoir in Ogden Valley, Utah, is threatened to become eutrophic from the current mesotrophic system, if nutrient loading to the reservoir is not controlled. Review of literature on causes, effects, and control of eutrophication was conducted as part of the research work that focused on furthering understanding of physical, chemical and biological factors that are either directly or indirectly related to Pineview Reservoir's water quality. The literature review covered the following specific areas: 1) eutrophication; 2) point- and non-point sources of nutrients; 3) surface and ground water flows, and their subsequent nutrient loadings to surface water bodies such as lakes and reservoirs; 4) atmospheric sources of nutrients; and 5) nutrient limitation and algal bioassays. This chapter presents a summary of the literature reviewed in relation to the afore-mentioned specific areas.

Eutrophication

Too much loading of water bodies with nitrogen and phosphorus can lead to eutrophication. Chapra (1997) defined eutrophication as a situation whereby a water body is over fertilized. Over fertilization of water bodies results in algal and/or cyanobacterial blooms which affect water body uses such as fisheries, recreation, crop and animal production, and culinary (Carpenter et al., 1998). Serious health problems or death may occur when humans or livestock ingest toxin-producing cyanobacteria (Carpenter et al., 1998; Kotak et al., 1993; Lawton and Codd, 1991; Martin and Cooke, 1994; McComb and Davis, 1993). Smith and Schindler (2009) reported that anthropogenic sources of nutrients were a major contributor towards excessive plant growth (eutrophication) in surface water bodies. The nutrient that limits phytoplankton growth in most water bodies is phosphorus (NRC, 1993a). Scarcity of phosphorus is attributed to insolubility of phosphorus minerals, limited forms of gaseous phosphorus, and adsorption of phosphorus to fine-grained soil particles which makes the nutrient biologically unavailable under oxic conditions (Chapra, 1997). Eutrophication of surface water bodies due to nitrogen has become a common and growing concern over the years (NRC, 1992; Carpenter et al., 1998). A review by Lewis and Wurtsbaugh (2008) concluded that the probability of nutrient limitation by N was almost equal to that of P. Co-limitation of N and P has also been reported (Elser et al., 2007; Lewis and Wurtsbaugh, 2008; Tank and Dodds, 2003). A study on periphyton by Tank and Dodds (2003) reported significant algal responses in streams to addition of N alone or a combination of N and P. They reported that the co-limitation may be attributed to additive effects of N and P due to very low N and P concentrations in the water bodies. None of the assays studied by Tank and Dodds (2003) showed nutrient limitation due to P only.

Point source and non-point source pollution of surface water bodies

Point sources are those pollution sources which have a defined point of pollutant discharge into a river, lake, ocean, or reservoir. Nonpoint sources do not have a clearly defined point of discharge into the water body (Carpenter et al., 1998). Carpenter et al.

(1998) reported that in the United States, approximately 82 percent and 84 percent of the total nitrogen and phosphorus loads to surface water bodies were contributed by nonpoint sources.

A good proportion of the phosphorus applied to agricultural land is not utilized by plants and therefore accumulates in the environment (Carpenter et al., 1998; Isermann, 1990; NRC, 1993b). Caraco (1995) and Carpenter et al. (1998) reported that about 3 – 20 percent of the phosphorus applied to agricultural lands gets eroded or leached to surface water bodies. Transport of phosphorus through ground water may be more significant than it has been believed over the past years (Hanrahan, 2012) because significant phosphorus leaching may occur in soils whose P content exceeds what plants would require (McDowell et al., 2001; Sharpley and Smith, 1994). Similarly, organic P is reported to be more mobile than inorganic P hence the former may easily leach to and get transported through ground water (Eghball et al., 1996). Preferential flow could also lead to relatively high P transport through ground water (Kronvang et al., 2007; Mittelstet et al., 2011; Sharpley et al., 2003).

Isermann (1990) and NRC (1993b) reported that only 18 percent of the N input in fertilizer is removed from farms in produce in the United States and Europe thereby leaving an average annual surplus of 174 kg N ha^{-1}. A study conducted by Kraft and Stites (2003) showed that irrigation influenced ground water nitrate concentrations beneath the irrigated field. The study measured nitrate concentrations of ground water beneath an irrigated vegetable field and also calculated nitrate budgets based on fertilizer applications. Another study conducted by Zhu et al. (2005) in northern China found that approximately 6 to 16 percent of the nitrogen applied to irrigated wheat and corn had leached out of the crop root zone. They attributed this to over irrigation and application of more N fertilizer than required by the crops.

In summary, the fight against agricultural pollution of surface water bodies needs to integrate agronomic management and soil hydrology in environmental management to ensure that proper amounts of nutrients are applied to cropland in a proper manner and areas of high potential for erosion and subsurface transport of N and P are properly managed (McCoy and Corbett, 2009; McDowell et al., 2001; Mitsch et al., 1999; Sharpley and Smith, 1994; Whitmore and Schröder, 2007). Site-specific analyses and farmer involvement in implementation of best management practices (Carpenter et al., 1998; Sharpley et al., 2001) would help bring the desired results.

Impact of surface- and ground water on the quality of surface water bodies

Agricultural fertilizers and sewage discharges are some of the contributors of ground water nutrient loads (Foster et al., 1989; Harter, 2003; Leip et al., 2011; Macpherson and Sophocleous, 2003; Parkinson et al., 1999; Schiavo et al., 2006; Schröder et al., 2003; Slomp and Van Cappellen, 2004; Withers and Lord, 2002). Lowe and Miner (1990) reported that ground water from the unconfined aquifer in Ogden Valley made a significant contribution of nitrate-nitrogen to Pineview Reservoir due to septic systems and agriculture. Both nitrogen and phosphorus transport in ground water is mainly in dissolved forms while transport of phosphorus in streams is mainly in particulate forms (Carpenter et al., 1998). According to Carpenter et al. (1998), soil particles transport much phosphorus and more phosphorus would be lost from agricultural land to water bodies if the soil had lots of phosphorus. The phosphorus that is attached to soil particles may become available and be used by aquatic biota (Carpenter et al., 1998; NRC, 1992; Sharpley and Smith, 1994).

Ground water nitrogen and phosphorus

Nitrogen can be lost from ground water through processes such as denitrification and adsorption to aquifer materials (Slater and Capone, 1987; Slomp and Van Cappellen, 2004; Weiskel and Howes, 1992). Robertson (2008) reported that deeper zones in the shallow aquifer had lower nitrate concentrations than shallower zones due to denitrification. Less ammonium N and dissolved organic carbon (DOC) concentrations (0.1 mg L^{-1} and 3 mg L^{-1}, respectively) in the deeper strata were also reported. The low ammonium concentrations were reported to have resulted from oxidation of the septic system effluent (Robertson, 2008). Phosphorus content of ground water can be reduced by its adsorption to organic matter and precipitation with metals such as Ca^{2+}, Fe^{3+} and Al^{3+} (Robertson, 2008; Weiskel and Howes, 1992). Presence of oxidizing aquifer conditions lowers the transport of phosphorus in ground water by providing a conducive environment for P to precipitate out of solution (Kroeger et al., 2008). Low pH (<5) has also been reported to reduce P movement in soil solution by enhancing its sequestration (Domagalski and Johnson, 2011). A septic plume study by Weiskel and Howes (1992) on medium to coarse sand soil (<0.1 percent clay) showed that septic systems contributed less to eutrophication of surface water bodies due to phosphorus than by nitrogen because much of the phosphorus got attached to soil particles and precipitated with iron and aluminum oxides. A study by Robertson (2008) on a septic system plume in a sandy calcareous soil in Ontario showed that P was not permanently removed from solution and limited removal of P from solution existed due to low concentrations (<0.1 mg L^{-1}) of dissolved iron. Robertson (2008) reported that the study had shown that P which had passed the vadose zone was very likely to remain mobile and impact surface water down-gradient.

Movement of P through the soil to ground water can be influenced by the presence of organic matter (Eghball et al., 1996). Eghball et al. (1996) reported high concentrations of dissolved P at a soil depth of 1.8 m beneath an irrigated corn field in Nebraska where both inorganic fertilizers and manure were applied. They said this might be attributed to either chemical reactions of P with organic manure constituents or movement of organic manure itself. Eghball et al. (1996) stated that aquifers with shallow water tables were prone to high P influx especially in areas with coarse textured soils. A study by Anderson and Magdoff (2005) attributed high concentrations of P in ground water to release of orthophosphate from soluble organic P. DOC has also been reported to enhance P movement in soils by reducing its adsorption to soil particles (Kang et al., 2011). Phosphorus movement beneath the ground surface can also be influenced by preferential flow as reported by Mittelstet et al. (2011) who found that preferential flow resulted in significantly higher subsurface P transport than surface runoff in an alluvial floodplain.

Atmospheric sources

Nitrogen can be fixed from the atmosphere and enter surface water bodies through biological and/or physical processes. Biological fixation of nitrogen is done by cyanobacteria and other bacteria capable of doing this, while physical fixation is done through precipitation events that are accompanied by lightning. Other origins of atmospheric nitrogen are agriculture and fossil fuel burning (Carpenter et al., 1998; Vitousek et al., 1997). Jaworski et al. (1997) compared fluxes of nitrogen from 33 rivers in the United States and found that there was positive correlation between the nitrogen in rivers and atmospheric deposition of nitrate.

Atmospheric sources of P also play a role in addition of nutrients to water bodies. For example, Jassby et al. (1994) reported four times as much soluble reactive phosphorus (SRP) deposition as surface runoff loading to Lake Tahoe. Jassby et al. (1994) reported that wet deposition accounted for a 10-yr average SRP loading rate of 0.37 µmol P m^{-2} d^{-1} to Lake Tahoe. An earlier study by Cole et al. (1990) had shown that average atmospheric SRP loading to Mirror Lake in New Hampshire in 1987 was reported 0.47 µmol P m^{-2} d^{-1}.

Nitrogen transport simulation using NLEAP-GIS 4.2

Nitrogen transport can be simulated using a number of modeling frameworks one of which is the Nitrogen Loss and Environmental Assessment Package (NLEAP-GIS 4.2). NLEAP-GIS 4.2 is capable of performing multiple simulations of the fate of nitrogen in the environment. Some of the inputs for the package include soil properties, climate data, and crop and management scenarios. The package is capable of downloading climate and soil data from NRCS websites such as the Soil Survey Geographic (SSURGO) Database. Simulation results from the package can be used to select best management practices that could be applied to a given watershed in order to reduce nutrient loading of water bodies (Delgado et al., 2010). Delgado et al. (2008) tested the tool and found that it accurately predicted nitrate leaching losses and atmospheric N losses for a number of locations.

Nutrient limitation and algal bioassays

Leibig's law of the minimum states that growth of any organism is constrained if one of the essential nutrients is in limited supply (Dodds, 2002). Algal and cyanobacterial blooms are mostly controlled by limited supplies of nitrogen and/or phosphorus. Phosphorus is the most common limiting nutrient to algal growth in freshwater bodies but both N and P loading need to be minimized in order to control algal blooms (NRC, 1993a). According to NRC (1993b), agricultural N and P loadings to surface water bodies have increased in the United States.

It is pertinent to determine which nutrient(s) is limiting development of phytoplankton blooms in order to be able to understand how the quality of a water body could be preserved or improved by ensuring that the supply of the limiting nutrient to the water body is controlled. One way in which the limited nutrients are identified is by conducting nutrient bioassay studies either in situ or in a laboratory environment (Dodds, 2002; Miller et al., 1978; Nyholm and Lyngby, 1988). Despite being more representative to the natural environment, in-situ studies are rarely done due to other limiting factors such as the complexity of the environment which may result in difficulties in having control over the bioassay (Dodds, 2002). It is therefore common practice to conduct nutrient bioassay studies in a laboratory setting. APHA (1995) outlined the procedures for conducting a laboratory-based algal bioassay.

Algal bioassays have been conducted over the years to determine whether nitrogen and/or phosphorus limit algal production in specific water bodies (Elser et al., 2007). Elser et al. (2007) reviewed different literature on nutrient limitation assays that were conducted in different climatic zones and different surface water bodies including freshwater systems. The review showed that nitrogen and phosphorus were equally important in influencing primary production in freshwater bodies as shown by significant increases in phytoplankton growth due to simultaneous additions of the two nutrients (Elser et al., 2007). Single nutrient limitations have also been reported in the literature. An algal nutrient bioassay conducted by Forsberg et al. (1975) to assess the effect of advanced waste water treatment and sewage diversion had shown that the limiting nutrient to growth of *Selenastrum capricornutum* in two-thirds of the samples was nitrogen. Forsberg et al. (1975) reported that phosphorus limited algal growth when total P concentrations were below 0.05 mg L^{-1}.

A bioassay conducted by Dobolyi and Ördög (1981) gave some conflicting results with the one that was conducted in a previous study for the same water body. The study by Dobolyi and Ördög (1981) showed that Lake Balaton was phosphorus limited. The strain used in their study was the green alga *Selenastrum capricornutum*. These results differed from a previous study that used a green alga strain *Scenedesmus obtusiusculus* CHODAT in which the limiting nutrient was found to be nitrogen. Dobolyi and Ördög (1981) reported that the previous study failed to agree with the anticipated results based on the lake water chemistry and they attributed the failure to the algae strain *Scenedesmus obtusiusculus* CHODAT which they deemed unfit for determination of nutrient limitation. It is possible that the differences in the results may imply different nutrient requirements of the respective algal species (Tank and Dodds, 2003).

Iron

Iron mostly exists in two oxidative states as ferric ion (Fe III), and ferrous ion (Fe II). Under reduced conditions, iron exists as Fe II while its oxidized state is Fe III. Iron in its oxidized state will complex with other compounds (such as oxygen, phosphate and sulfate) and precipitate out of solution while reduced iron salts are more soluble. Presence of the different forms of iron in the environment impacts bioavailability of both nitrogen and phosphorus. Korom (1992) reported that reduced iron acts as a source of electrons for denitrification of NO_3^- when denitrifying bacteria are present.

Complexation of ferric iron with phosphate under oxic conditions maintains presence of phosphorus in lake or reservoir sediments (Marsden, 1989). The phosphorus gets released into solution when anoxic conditions develop (Marsden, 1989). This provides the additional phosphorus needed for phytoplankton production.

Dissolved organic carbon

Dissolved organic compounds provide a source of energy, carbon and other nutrients to the process of denitrification carried out by bacteria in which nitrate is reduced to nitrogen gas. A study conducted by Green et al. (2008) to evaluate processes controlling denitrification in agricultural land showed that nitrate accepts electrons which could be donated by dissolved organic carbon. Green et al. (2008) reported that dissolved organic carbon usually correlated with nitrate in addition to specific conductance, Ca^{2+}, Mg^{2+}, K^{+}, Cl^{-}, and SO_4^{2-}. The study further showed that denitrification rates were relatively low in shallow aquifers underlying agricultural land. The study also showed that there was strong correlation between nitrate and other agricultural chemicals which implied that the nitrate observed was of agricultural origin (Green et al., 2008).

Water infiltration, percolation, and aquifers

Infiltration is the process through which water enters the soil profile under the influence of gravity and capillary forces. When the rate of infiltration becomes constant, movement of water occurs through water saturated soil micro pores under the influence of gravity. This process is referred to as percolation. Deep percolating water becomes or joins ground water in a storage unit called an aquifer (Brooks et al., 1997).

An aquifer is a soil material that holds water which can be abstracted by means of wells and the water sometimes comes to the surface through springs. Aquifers that are bound by impermeable or semi-impermeable soil or rock layers both on top and at the bottom are termed confined while those with only a bottom impediment to vertical water flow are unconfined. Water in a confined aquifer is held under pressure hence these aquifers are referred to as artesian aquifers. Unconfined aquifers are also referred to as water table aquifers because the level of ground water in this type of aquifer is called a water table. Some unconfined aquifers occur as small pockets of water which do not cover a wide area. Such aquifers are known as perched, unconfined aquifers (Brooks et al., 1997; Todd and Mays, 2005).

Ground water in Ogden Valley's Pineview Reservoir area occurs in confined, unconfined and perched aquifer formations (Fig. 1). The center of the southern part of the valley comprises both unconfined and confined aquifer formations. The unconfined aquifer overlies the confined aquifer and the two aquifers are separated by a silt clay layer. The unconfined aquifer overlying the confined aquifer is also referred to as the shallow water table aquifer (Avery, 1994).

Ground water flow determination

The direction of ground water flow (flow lines) can be determined by connecting higher and lower elevation contour lines with perpendicular lines (Brooks et al., 1997; Todd and Mays, 2005). The rate of ground water flow can be computed from the Darcy equation. Aquifer hydraulic conductivities for use in the Darcy equation can either be determined in the field from slug tests or obtained from literature (Todd and Mays, 2005). Landon et al. (2001) recommended field determination of hydraulic conductivities in order to obtain values that more closely represent the aquifer in question.

Water elevation monitoring and barometric pressure compensation

High frequency monitoring of ground water elevations can be achieved through the use of pressure transducers (Fisher and Healy, 2008). Rasmussen and Crawford (1997) reported the need for simultaneous measurements of the barometric pressure and water levels in wells which are not exposed to the atmosphere. If barometric pressure is not measured beneath the well cap, the wells need to be vented to the atmosphere so that the pressure inside and outside the well is at equilibrium (Rasmussen and Crawford, 1997). Price (2009) stated the need for an additional device monitoring atmospheric pressure whether or not the pressure transducer being used is vented. The additional device is used for barometric pressure compensation of the absolute pressure data (Price, 2009; Rasmussen and Crawford, 1997).

Estimate of ground water discharge and loading

Watershed water balance methods have been used to estimate ground water discharge (Avery, 1994; Kroeger et al., 2008; Tetra Tech, 2002). This approach uses a water balance equation in which ground water discharge is the unknown. The problem with this approach is that uncertainties are lumped together in the unknown variable therefore making the estimated value less accurate.

Pineview Reservoir's TMDL study applied a water balance model called the GWLF to estimate the ground water flow and its subsequent nutrient loading from the shallow unconfined aquifer to the reservoir (Tetra Tech, 2002). Respective background nutrient concentrations of 0.75 mg dissolved N L^{-1} and 0.02 mg dissolved P L^{-1} obtained from Wallace and Lowe (1998) and an observed sampling data value (Tetra Tech, 2002) were used in the loading estimates. Summer nutrient loadings were based on an annual ground water flow of 24.7 million m^3 that was reported by the Weber Basin Water Quality Management Council (1990).

Avery (1994) simulated ground water discharge from the shallow unconfined aquifer to Pineview Reservoir and estimated that it was approximately 33.3 million m^3 yr^{-1}. The simulation applied a three dimensional finite difference computer model which was calibrated using a step-by-step approach to obtain what was referred to as "acceptable values of water-level altitude, water level change, and flow rate." The simulation of ground water flow was hampered by sparse data and a recommendation was made for collection of more data on surface water conditions and additional observation wells data in order to refine the simulation (Avery, 1994).

Based on the information above, this study was conducted to bridge the information gap that existed in knowledge of the flow and nutrient contributions of the shallow, unconfined aquifer to Pineview Reservoir. The study was also done to increase the quality of information in order to reduce uncertainties in use and management decisions related to improvement or preservation of the reservoir water quality. This is because any decisions made regarding use and management of the reservoir would have socio-economic impacts to reservoir managers, including communities in Ogden Valley.

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CHAPTER 3

PINEVIEW RESERVOIR NUTRIENT LOADING, UNLOADING AND THE ROLE OF GROUND WATER IN THE ESTIMATES*

Abstract

A Total Maximum Daily Load (TMDL) study for Pineview Reservoir was completed in 2002. As is often the case, the data used in the TMDL analysis were sparse. Concerns over the accuracy and implementation of the TMDL led the Utah Water Research Laboratory and the Weber Basin Water Conservancy District to collaborate on a study of the reservoir starting in October 2007. An objective of the research work was to expand the information available about the sources of nitrogen and phosphorus to Pineview Reservoir and to improve the description accuracy of nutrient transport through the watershed. It was determined that: 1) thermal stratification occurred in most of the reservoir and that the hypolimnion became anoxic by late July; 2) phosphorus and nitrogen accumulated in the anoxic hypolimnion and were exported with reservoir withdrawals; 3) nutrient concentrations and phytoplankton production responded to water column turnover; and 4) turnover and nutrient mixing into the water column appeared to be related to both draw-off of the hypolimnion and wind-mixing under near-isothermal

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conditions. These findings led to a detailed study of nutrient transport from the watershed to Pineview Reservoir. Grab sampling and high frequency monitoring of both ground water and surface water were conducted. Nitrogen and phosphorus transport through surface and ground water were determined. The study has shown that ground water loadings of dissolved nitrogen and soluble reactive phosphorus were 35 - 40 percent lower than those estimated in the TMDL. Nitrate-N loading from surface water was 1.8 fold higher and total phosphorus loading was 2.4 fold higher than was estimated in the TMDL study.

Introduction

In the U.S. intermountain west, eutrophication of reservoirs is typically associated with increasing phosphorus concentrations in the water column and the resultant growth of cyanobacteria and algae to unacceptable concentrations (Hein, 2006). Water transparency decreases, taste and odor may increase and hypolimnetic dissolved oxygen (DO) may decrease to concentrations unusable by fish as biomass decomposes on or near the sediments. A reservoir's aesthetic value may decrease to unacceptable levels, recreational uses may decrease and taste and odor treatment costs for municipal water supply may reach unaffordable levels. Negative impacts may also occur to downstream ecosystems when anoxic water is released from the bottom of a reservoir. Water quality managers, including government environmental protection agencies, strive to slow or even reverse the eutrophication process. The U.S. Federal Clean Water Act, as administered by the U.S. Environmental Protection Agency, requires states to determine total maximum daily loads (TMDLs) of pollutants to water-bodies including reservoirs. Nutrients, especially phosphorus (P) and nitrogen (N), are often listed as pollutants that must be controlled to bring reservoirs into compliance with beneficial use criteria.

The depth of study and the quality of the TMDL process is often less than desired due to funding and time limitations. Estimation of the TMDL is generally computed using existing data and water quality models. Available data are often sporadic and limited in scope, and the use of anecdotal information may be attractive given data limitations. The data limitations may therefore result in important, inadvertent omissions in the development of the TMDL.

The TMDL for Pineview Reservoir in Ogden Valley, Weber County, Utah, was approved in 2002 (Tetra Tech, 2002). Pineview Reservoir is an impoundment of the Ogden River, and its principal tributaries include the North, Middle, and South Forks of the Ogden River, Geertsen Creek, and Spring Creek. The initial phase of the reservoir was completed in 1937, and it was expanded to its current capacity in 1957. It has a storage capacity of approximately 140 million m^3 (110,000 ac-ft) with a surface area of 1200 ha (2900 ac) and a maximum depth of 25 m (81 ft). It is a multi-use reservoir that provides water for irrigation and municipal uses; recreational fishing, boating, and water sports; and a means of flood control for the community of Ogden, which is located approximately 11 km (7.0 mi) downstream.

The TMDL study used data available from public databases and management organization records. Most of the more recent literature on the reservoir and its watershed was reviewed (Doyuran, 1972; Lowe and Wallace, 1999; Weber Basin Water Quality

Management Council, 1990). However, an important ground water resource study of the valley surrounding Pineview Reservoir that was conducted by the U.S. Geological Survey (Avery, 1994) was not cited. The data analysis done in the TMDL could not conclusively identify either N or P as the growth limiting nutrient. Recommendations for limiting N and P loading from irrigation, on-site wastewater treatment systems, livestock manure, and rangeland were made along with cost estimates for alternatives for achieving load reduction goals. Internal cycling of phosphorus from the reservoir sediments was assumed to be negligible because the model predicted that DO concentrations would fall below 0.2 mg L^{-1} for only a few days each year. Ground water discharging to the reservoir from the unconfined (water table) aquifer was estimated to carry approximately 600 kg P and 22,000 kg N per year. Modeling predicted that the highest N and P loading rates from ground water would occur in June and July when irrigation water application rates were highest.

The Weber Basin Water Conservancy District (WBWCD) is a purveyor of treated municipal water and irrigation water (untreated, secondary water supply) in Weber and Davis counties and has rights to a substantial fraction of the water stored in Pineview Reservoir. Following the completion of the Pineview Reservoir TMDL, a water quality manager at the WBWCD raised questions about the accuracy of the nutrient load estimates and the predicted biological response because of the sparse data available. Concerns were also raised about the practicality of controlling nutrient loads sufficiently using approaches recommended in the TMDL report. In response to these concerns, WBWCD and the Utah Water Research Laboratory at Utah State University began collaboration on Pineview Reservoir water quality investigations, and data collection began in October 2007. The objectives of this project included characterizing the loading of N and P to Pineview Reservoir and describing the fate of the P entering the reservoir, including internal cycling and export.

Approach and methods

Pineview Reservoir hydrology is snowmelt driven. In years with near-normal or above-normal winter and spring precipitation, the reservoir fills to capacity by late May. Peak withdrawals are in July and August in response to irrigation demands. Water is withdrawn through the outlet near the bottom of the dam throughout the year to supply hydroelectric works. Seasonal withdrawals include supply for a municipal water treatment plant and irrigation. Reservoir instrumentation and sampling schedules were designed to provide more frequent data in anticipation of thermal stratification in the early summer, hypolimnion drawdown, and destratification in the early fall (Worwood, 2011). Water quality attributes including temperature, DO, pH, and conductivity were measured using *in situ* sondes at five locations (Fig. 2), which were anticipated to represent the reservoir.

A Van Dorn sampler was used to collect grab samples at three depths through the water column at each sampling location. Grab sample collection and measurement of discharge were also conducted on five major surface water tributaries. Samples were analyzed for nutrient concentrations according to the methods listed in Table 1.

Concentrations below detection limits were estimated by imputation when possible (Gilliom and Helsel, 1986).

Pineview Reservoir inflows include ground water contributions from both unconfined (water table) and confined aquifers. The northern part and southern margins of Ogden Valley have unconfined aquifer formations, while the center of the southern part of the valley is confined in the lower, principal aquifer and unconfined in the overlying, shallow, water table aquifer (Avery, 1994). Figure 3 shows the aquifer formations of the valley.

Pineview Reservoir is located above the silt-clay confining layer and, because upward flow from the confined aquifer is very slow, its principal ground water recharge comes from the unconfined aquifer. Primary recharge of the shallow aquifer above the confining bed comes from streamflow, precipitation, and the unconfined aquifer beyond the confining layer margin (Snyder and Lowe, 1998; Tetra Tech, 2002).

In order to further the understanding of N and P loads to the unconfined aquifer and their subsequent loading to Pineview Reservoir, five monitoring wells were constructed in February 2010 (Fig. 2). All the wells were finished in the unconfined aquifer overlying the confining layer and were located less than 100 m (330 ft) from the shoreline in areas with easy access and anticipated ground water flow direction toward the reservoir (Fig. 33, Appendix B). The wells were 5.1 cm (2 in.) i.d. PVC with a 1.5 m (5 ft) screen of slot width 0.25 mm. Their average depth was 8.2 m. Ground water elevations in the wells were monitored every 12 h using pressure loggers (Onset Computer Corp., Pocasset, Mass.).

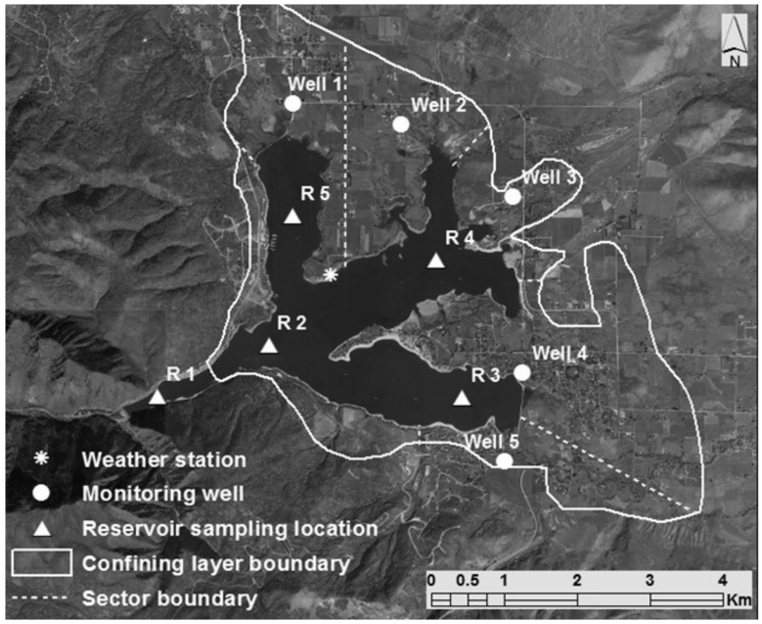

Figure 2. Reservoir sampling and ground water monitoring well locations in Ogden Valley (ESRI). Reservoir sampling locations are labeled R1 through R5

Table 1. Laboratory analysis methods (APHA, 1995)

Variable	Method
Soluble reactive phosphorus	SM 4500-P B & P E
Total phosphorus	SM 4500-P E
Total dissolved phosphorus	SM 4500-P B & P E
Nitrate + nitrite-nitrogen	SM 4500-NO_3^- F
Ammonium-nitrogen	SM 4500-NH_3 G
Dissolved organic carbon	SM 5310 C
Total dissolved iron	SM 3111 B
pH	SM4500-H+ B
Conductivity	SM2510 B
Chlorophyll *a*	HPLC EPA 447.0[a]

[a] = Arar (1997)

Direct measurements of water table elevations were made in conjunction with well water sampling at periods ranging from weekly to monthly using an electronic water level indicator. The measurements were used as reference water table elevations when converting pressure logger data to water table elevations in HOBOware, the software accompanying the pressure loggers. Data from the loggers were compensated for barometric pressure (Rasmussen and Crawford, 1997) in HOBOware using barometric pressure data obtained from the weather station located on the reservoir bank (Fig. 2). Water table elevations were expressed in units of meters above mean sea level based on geodetic surveys conducted at each well. Ground water contour maps were then generated by ordinary kriging using the exponential semivariogram model in ArcGIS 3D Analyst. From the resulting contour maps, hydraulic gradients for each of the wells were computed by dividing the change in elevation across contour lines with the horizontal distance between them.

Slug tests were conducted in all monitoring wells to determine the hydraulic properties of the aquifer. The tests involved bailing out 1 L of water from each well. Respective well drawdown and recovery data were logged using pressure loggers. Slug test data were analyzed, and hydraulic conductivities and Darcy fluxes were calculated (Todd and Mays, 2005). The resulting parameters and observed water table elevations relative to reservoir elevations (Bureau of Reclamation, 2011) were used to estimate daily ground water flow from the shallow, unconfined aquifer to Pineview Reservoir.

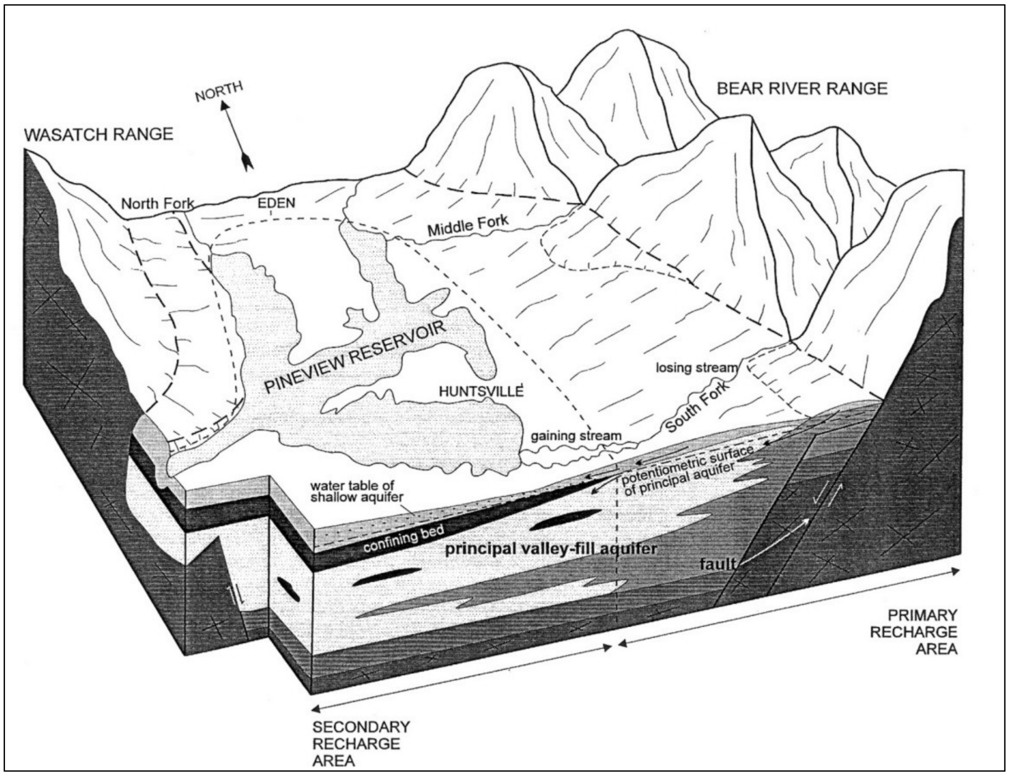

Figure 3. Hydro-geologic formation of Ogden Valley (Snyder and Lowe, 1998)

Ground water samples were bailed or pumped with a bladder pump from the five wells at least once a month and analyzed. The water was pumped through polyethylene tubing and a glass chamber using a portable 4.2 cm (1.66 in.) bladder pump. The chamber had slots into which probes for measuring temperature pH, DO, and EC were screwed. The apparatus were rinsed by pumping distilled deionized water through the assembled system before inserting the pump into each well. The cut-off time for the rinse was when the EC readings were ≤ 2 μS/cm. The cut-off point was determined through field experience having discovered how difficult it was to get an EC reading of zero. Ground water grab samples were collected after variations among three successive meter readings were less than or equal to 1 percent during continuous well pumping.

Water quality attributes determined included soluble reactive P (SRP), total dissolved (0.45 μm filtered) P (TDP), nitrate-N (NO_3 + NO_2-N), ammonium-N (NH_4-N), total dissolved organic carbon (DOC), total dissolved iron, temperature, pH, and conductivity (Table 1). Daily tributary nutrient concentrations and flow rates were estimated from the grab sample concentrations using the rank-data distribution method (Lee, 2008). Ground water nutrient concentrations were linearly interpolated to estimate daily concentrations. Daily loads were computed from the product of surface water or ground water flow rates and nutrient concentrations.

Ground water nutrient concentrations for the sectors south and west of the reservoir, where no wells were located, were estimated as the geometric mean of the concentrations from the five wells. The geometric mean was also applied in estimating the two sectors' hydraulic gradient and hydraulic conductivity for subsequent computation of ground water flows and loads from these sectors. Sector boundaries were arbitrarily assigned by assuming that they occurred in the areas where the ground water contours representing adjacent wells formed steep gradients and intersected with the confining layer boundary (Fig. 2). Data analysis to determine significant differences was done using one-way analysis of variance (ANOVA) in R (R Development Core Team, 2011).

Results and discussion

Stream inflows had average SRP and TP concentrations of 9.7 and 42 μg P L^{-1}, respectively, and an average nitrate-N concentration of 0.25 mg N L^{-1} from 1 May 2010 through 30 April 2011. Total stream inflow to the reservoir for this period was approximately 170 million m^3 (140,000 ac-ft). The surface water load of TP was approximately 19,000 kg P, of which 14 percent was SRP. The SRP load was 32 percent lower than the TMDL estimate of 3,700 kg P $year^{-1}$ (Table 2). This difference probably resulted from differences in both flow rate estimates and nutrient concentrations given the limited data available for the TMDL study. The nitrate-N load for the same period was approximately 49,000 kg N, which is significantly higher than the TMDL estimate.

Table 2. Comparisons of Pineview Reservoir's water inflows and nutrient loadings estimated in the TMDL study (Tetra Tech, 2002) with those determined in the current study (2008-2011)

Source and Parameter	Unit	TMDL Study	Current Study
Ground water			
Flow	$m^3\ year^{-1}$	2.5×10^7	3.4×10^6
Soluble reactive/ortho-P	kg $year^{-1}$	590	360
Total dissolved P	kg $year^{-1}$	ND[a]	490
Nitrate-N	kg $year^{-1}$	22,000	14,000
Tributaries			
Flow	$m^3\ year^{-1}$	1.3×10^8	1.7×10^8
Soluble reactive P	kg $year^{-1}$	3,700	2,500
Total P	kg $year^{-1}$	7,900	19,000
Nitrate-N	kg $year^{-1}$	27,000	49,000

[a] ND = not determined.

The annual ground water discharge into Pineview Reservoir from the shallow unconfined aquifer was determined to be 3.4 million m^3 (2,700 ac-ft). This quantity was computed from Darcy fluxes of 23 ±35 cm d^{-1} (mean ±1 SD). The daily ground water discharge from the shallow, unconfined aquifer to Pineview Reservoir rose steadily between 22 May and 13 October 2010 (Fig. 4). This rise was probably related to irrigation since the irrigation season in Ogden Valley runs from May through October. The sustained high flows in the fall of 2010 through the spring of 2011 were probably caused by unusually high rainfall and snowmelt experienced in Utah during this period. The daily average precipitation rate for August 2010 through March 2011 was 0.28 cm d^{-1}, approximately 1.7-fold higher than that for May 2009 through June 2010.

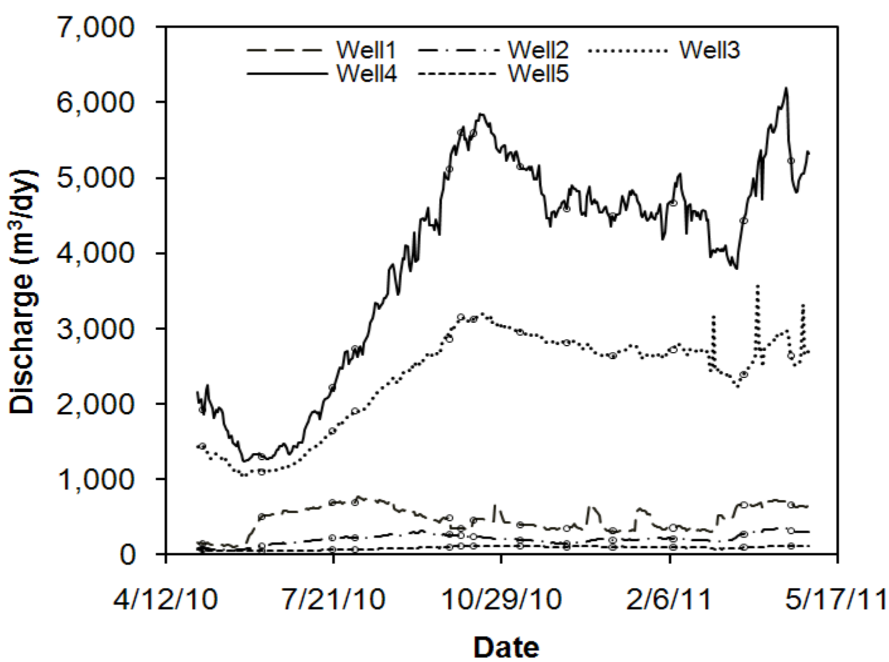

Figure 4. Daily ground water discharges from the shallow, unconfined aquifer to Pineview Reservoir for the period from May 1, 2010 through April 30, 2011. Circles denote days on which sampling was done

SRP concentrations for wells 4 and 5 were consistently higher than for the other three wells (Fig. 5). The two wells had average SRP concentrations of 220 and 280 μg P L^{-1}, respectively. Similarly, TDP concentrations for wells 4 and 5 were usually higher than for the other wells (Fig. 6). The average TDP concentrations for these two wells were 320 and 310 μg P L^{-1}, respectively. The average SRP and TDP concentrations for the five wells were both significantly higher than the background concentration of 20 μg P L^{-1}that was used during the TMDL study to estimate reservoir phosphorus loading from the unconfined aquifer.

Figure 7 shows that well 1 had pulses of nitrate-N in summer and spring that consequently resulted in the highest average concentration among the wells. Its average concentration was 9.7 ±11 mg N L^{-1}, while wells 2 through 5 had nitrate-N

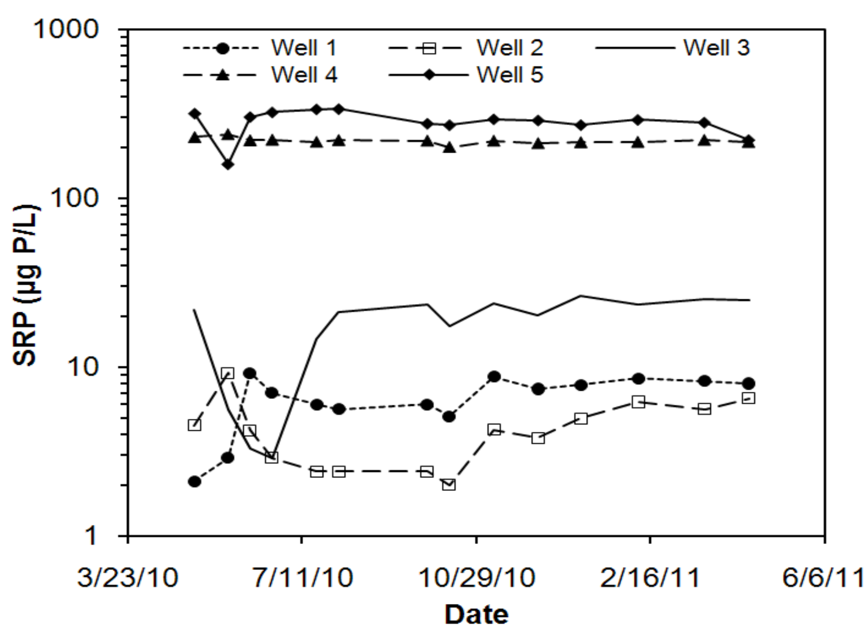

Figure 5. SRP concentrations for Ogden Valley’s shallow ground water monitoring wells

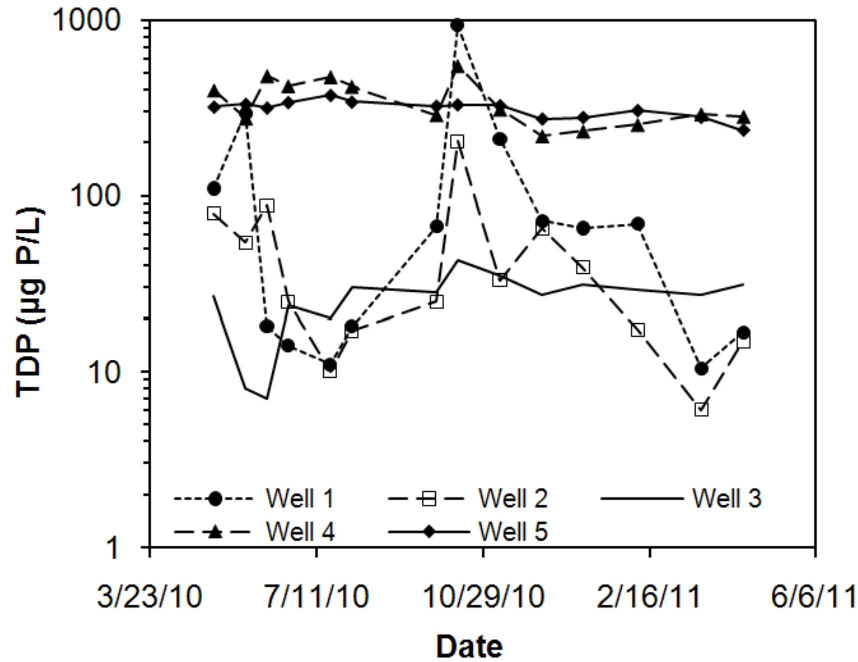

Figure 6. TDP concentrations for Ogden Valley's ground water monitoring wells

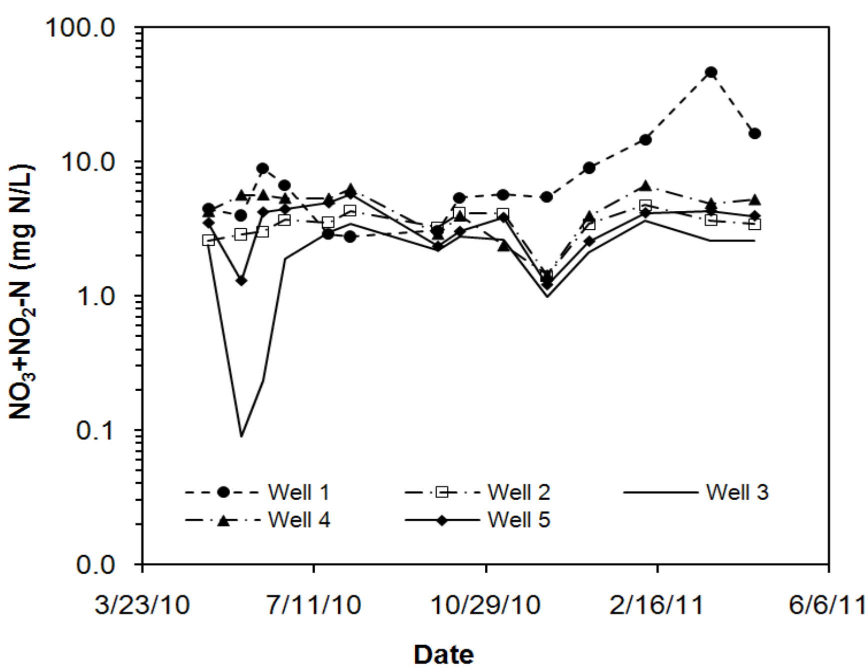

Figure 7. NO_3 + NO_2-N concentrations for ground water monitoring wells in Ogden Valley

concentrations of 3.4 ±0.8, 2.2 ±1.1, 4.6 ±1.5, and 3.5 ±1.3 mg N L^{-1}, respectively. With few exceptions, ammonium-N concentrations in all the wells were below the detection limit of 0.04 mg N L^{-1}.

The highest average DOC concentration was observed in well 5, followed by well 4 (Fig. 8). High concentrations of DOC (Table 17, Appendix F) and nitrate-N in these two wells suggest that the sources of nutrients in these wells could be related to wastewater. The annual median water temperature in the wells was 11°C, and DO concentrations ranged between 2.3 mg L^{-1}, in well 1, and 8.8 mg L^{-1} in well 4. Well 1 had the highest annual average dissolved iron concentration of 0.03 ±0.05 mg Fe L^{-1}, followed by well 4 (0.02 ±0.02 mg Fe L^{-1}). Well 1 was located near a major roadway and had an early February electrical conductivity (EC) of 3,340 μS cm^{-1}, indicating that it was influenced by road salting, but the annual median EC in the other wells was 437 μS cm^{-1}. The pH in all the wells ranged from 6.0 to 8.0.

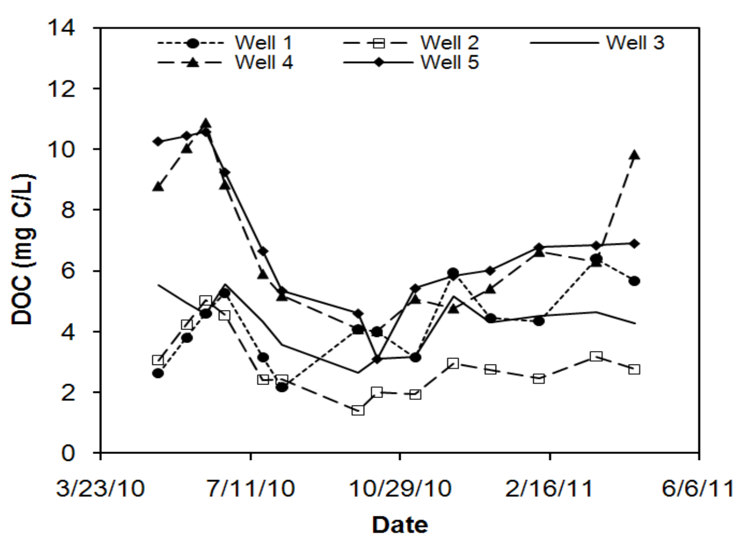

Figure 8. DOC concentrations for Ogden Valley's ground water monitoring wells

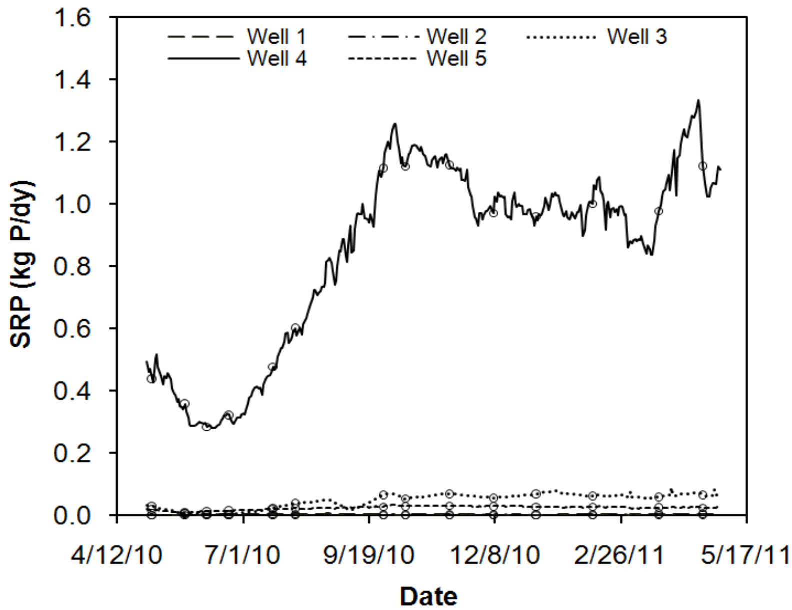

Figure 9. Daily SRP loads from the shallow, unconfined aquifer to Pineview Reservoir. Circles denote days on which grab sampling was done

Figure 9 shows that the daily SRP loadings from the well 4 sector were substantially higher than from the other sectors. ANOVA for the results showed that the SRP loadings were significantly different ($p \leq 0.05$) among all the well sectors. The high SRP loading from the well 4 sector is a reflection of both the high ground water flows and the high SRP concentrations (Table 16, Appendix E) observed from this well.

The well 4 sector had the highest daily TDP loads compared to the other sectors (Fig. 10). ANOVA showed that the daily TDP loads were significantly different ($p < 0.05$) among all the sectors. The mean daily TDP load for the well 4 sector was 2.4 kg P. The dynamic nature of the loading rate was surprising. The high TDP loads might be a result of nutrient flux from on-site wastewater disposal and irrigated land transported by high ground water flows. Higher frequency monitoring, perhaps twice monthly, in well 4

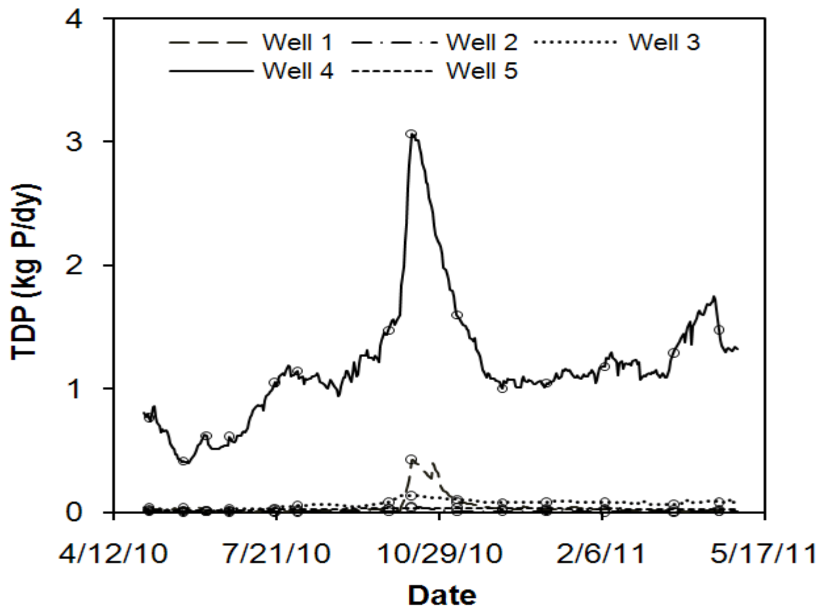

Figure 10. Daily TDP loads from the shallow, unconfined aquifer to Pineview Reservoir. Circles denote days on which grab sampling was done

may provide a more accurate description of the significantly high but short-lived nutrient loads.

Daily nitrate-N loading from the sectors represented by each well ranged from 0.06 to 33 kg N (Fig. 11). The well 4 sector had the highest mean daily nitrate-N load of approximately 17 kg N, followed by the well 1 (11 kg N) and well 3 (6.0 kg N) sectors. These variations in loads from one location to another reflect aquifer heterogeneity, as shown in Figure 4, and the nitrate-N concentration variability presented in Figure 7.

DOC loading from the well 4 sector to the reservoir was significantly higher than from all the other sectors. This was probably because this sector had the highest ground water flow and consistently high DOC concentrations. This well is located in Huntsville Town, which has a septic system density of approximately 1.7 systems ha^{-1}.

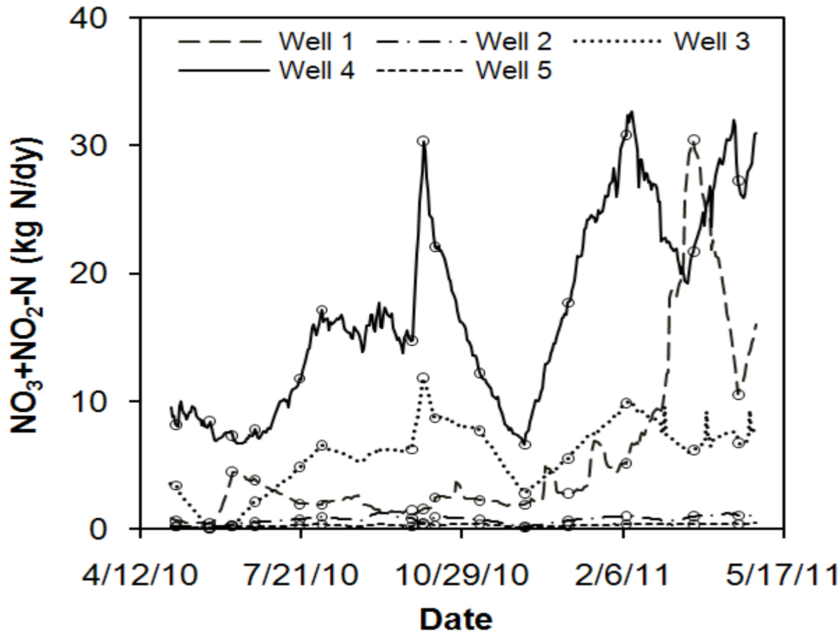

Figure 11. Daily NO_3 + NO_2-N loading from the shallow, unconfined aquifer to Pineview Reservoir. Circles denote days on which grab sampling was done

The high DOC concentrations support the hypothesis that wastewater influences the nutrient load to Pineview Reservoir in this sector.

ANOVA of TDP versus SRP concentrations from the grab samples showed no significant difference between the two parameters within each well except well 3, where TDP concentrations were significantly higher ($p \leq 0.05$) than SRP. This implies that, in general, nutrient loads to ground water from agriculture, horticulture, and on-site wastewater may be important since the land use in the areas where the wells are located is mostly agriculture and low-density housing.

Approximately 13 percent of the SRP loading to Pineview Reservoir was contributed by ground water (Table 2). Similarly, nitrate-N loadings from ground water were only about 20 percent lower than those from surface water inflows. The nitrogen loadings from ground water to Pineview Reservoir were approximately 36 percent lower than the estimate made by the TMDL study. Table 2 also shows that stream nitrogen loadings determined in this study were nearly two times higher than the TMDL prediction. The TMDL study's predicted SRP loading from the streams was higher, while the TP loading was substantially lower than that determined in the present study. Annual phosphorus loadings to Pineview Reservoir through ground water were lower than those reported in the TMDL study, while tributary loads were higher than the TMDL estimates.

Internal phosphorus loading played a significant role in producing high but intermittent phosphorus concentrations in the reservoir, as evidenced by the high hypolimnetic total phosphorus concentrations (Fig. 12) when DO concentrations fell below approximately 2 mg L^{-1} (Marsden, 1989).

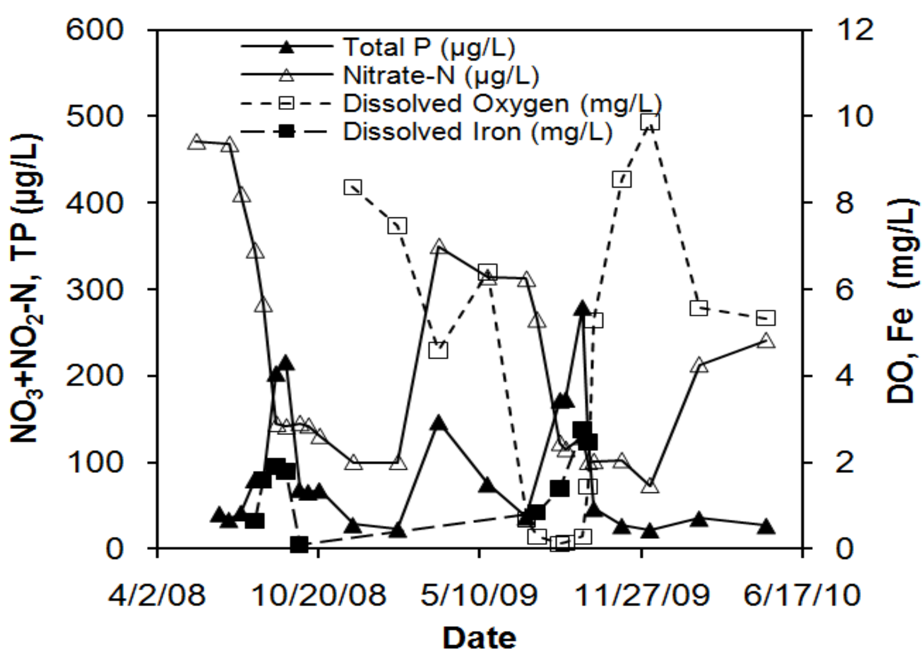

Figure 12. Nutrient concentrations for the Pineview Reservoir mid-reservoir sampling site (R2), near-bottom depth

Contrary to the conclusions made by the TMDL study, it was found that internal cycling of nutrients, especially P, is occurring in Pineview Reservoir and that annually observed phytoplankton blooms can be attributed to the release of benthic nutrients. It was estimated that 14,800 kg P were exported through water withdrawals between 15 April 2009 and 14 April 2010. This large P export resulted from the release of hypolimnetic water throughout the summer irrigation season. Given the total P import estimate found in the present study of 19,500 kg $year^{-1}$, the accumulation of P in the reservoir is substantially mitigated by its export. The high hypolimnetic TP concentrations were chronologically followed by high chlorophyll *a* concentrations (Fig. 13) in the years 2008 and 2009, indicating that internal loading of P may have provided the additional bioavailable P needed to initiate phytoplankton blooms.

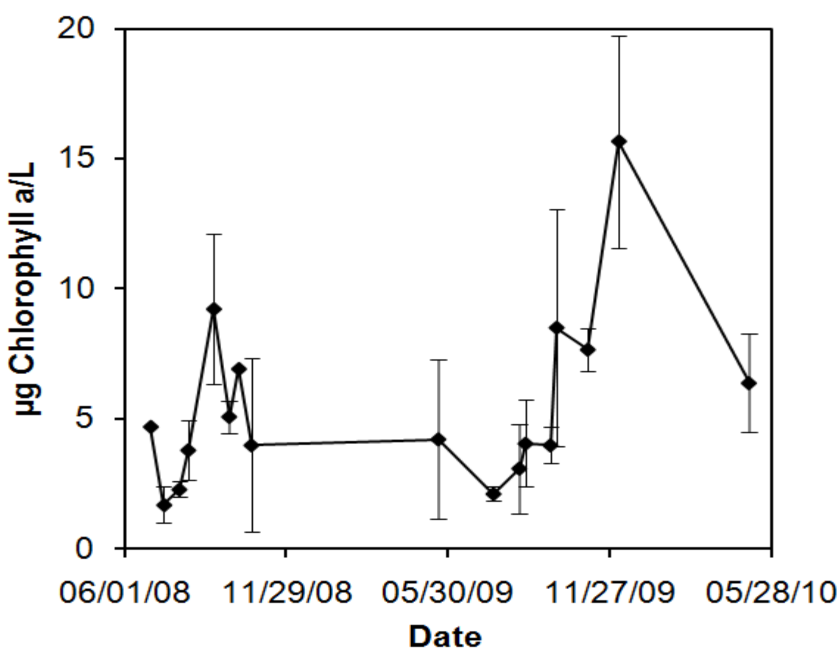

Figure 13. Average surface water chlorophyll a concentrations at the five Pineview Reservoir sampling locations (Fig. 2). The error bars are 95 percent confidence intervals

Chlorophyll a results have also shown that Pineview Reservoir's trophic state is mostly oligotrophic to mesotrophic (supported by 82 percent of the samples). The TMDL report showed that the reservoir's trophic status may be mesotrophic to eutrophic. However, the current study has shown that the trophic status reported in the TMDL may exist for only 18 percent of the year.

Conclusion

Loading of nitrogen and phosphorus to Pineview Reservoir in Ogden Valley and the fate of the phosphorus entering the reservoir including internal cycling and export have been characterized. The information available from the TMDL study has been significantly enhanced.

Surface water loads of nitrogen and phosphorus were 78 percent and 98 percent of the total and highest during spring runoff associated with snow melting. Internal cycling of phosphorus from the reservoir sediments coincided with the initiation of phytoplankton blooms. Export of phosphorus via release of hypolimnetic water mitigates the accumulation of phosphorus in the reservoir sediments.

Ground water nutrient loadings from the shallow, unconfined aquifer to the reservoir varied significantly from one sector to another, reflecting the heterogeneity of the aquifer and nutrient concentrations. Annual phosphorus loadings to Pineview Reservoir through ground water were lower than those reported in the TMDL study, while tributary loads were generally higher than the TMDL estimates. Relatively high nitrate-N and dissolved organic carbon concentrations that accompanied higher total dissolved phosphorus in ground water, near or within residential areas, implied that wastewater influenced the nutrient loads in these sectors. Further studies to provide direct evidence of the impact of septic systems on nutrient loading to the reservoir need to be carried out.

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CHAPTER 4

GROUND WATER FLOW SPATIAL VARIABILITY AND NUTRIENT TRANSPORT TO A RESERVOIR IN AN IRRIGATED MOUNTAIN VALLEY

Abstract

A study was conducted to quantify and characterize the spatial distribution of ground water flow and nutrient loading in the mountainous and irrigated agriculture dominated Ogden Valley, Utah, for 15 November 2010 through 14 November 2011. Large differences in flow and nutrient loading estimates from previous studies motivated the study. Arc GIS kriging techniques were employed in analyzing high frequency ground water flow and grab sample nutrient concentration data.

Results from the study showed large spatial variations in ground water flows and nutrient loadings towards Pineview Reservoir. Spatial variation in flows was attributed to variations in hydraulic gradients and saturated thickness while nutrient loading variations were attributed to nutrient flushing to ground water due to snowmelt and irrigation water. Both agricultural and domestic non-point sources appeared to influence the nutrient loadings. Uncertainty in spatial variations due to a large hydraulic conductivity range of 0.86 – 22 m d^{-1} and possible existence of preferential flow paths were believed to have been minimized by aggregation of low and high flows in the results.

The median total dissolved phosphorus concentration for nine wells (104 µg P L^{-1}) was more than 3-fold higher than the median for five wells (32 µg P L^{-1}), signifying large spatial variations. Much spatial variation in flow rate was observed, with the largest confidence interval of 1,518 – 5,077 m^3 d^{-1} on 6 May and the smallest (447-1,814 m^3 d^{-1})

on 27 June 2011. The wide confidence interval observed in spring may be attributed to rapid increases in reservoir elevations due to spring runoff emanating from snowmelt in the mountains and spatially variable changes in water table elevation. These fluctuations probably caused large variations in the hydraulic gradients and saturated thickness due to differences in cell locations relative to the reservoir shoreline.

Introduction

Eutrophication of reservoirs in the US intermountain west is usually attributed to increasing phosphorus concentrations in the water column which results in cyanobacteria and algae growth to unacceptable concentrations (Hein, 2006). Cyanobacteria and algae growth affect the beneficial uses of the reservoir by lowering its aesthetic value through development of taste and odor. Dissolved oxygen concentrations in the water column also decline due to oxygen depletion emanating from phytoplankton respiration and decomposition of dead plankton. Low dissolved oxygen concentration makes the reservoir less conducive to aquatic life resulting in death of fish and other aquatic fauna. The US Environmental Protection Agency implemented the US Federal Clean Water Act under which states are required to determine the Total Maximum Daily Loads (TMDLs) for reservoir and/or lake water pollutants. Most TMDLs list nitrogen (N) and phosphorus (P) as the nutrients influencing primary production in reservoirs and lakes. Control of N and P loading to water bodies is therefore of prime importance towards maintaining the beneficial uses of these water bodies.

Pineview Reservoir is one of the reservoirs whose TMDL studies stipulated the need for controlling both nitrogen and phosphorus in order to abate eutrophication (Tetra Tech, 2002). The reservoir is located in Ogden Valley, approximately 11 km east of Ogden City in Weber County, Utah, USA. It is an impoundment of the Ogden River and receives water from the North, Middle, and South Forks of the Ogden River, Geertsen Creek, Spring Creek, various other smaller tributaries and ground water. The maximum storage capacity of the reservoir is approximately $140x10^6$ m^3 and the annual inflow is approximately $170x10^6$ m^3 $year^{-1}$. Pineview Reservoir's TMDL study was conducted following increasing concerns about development of algal/cyanobacteria blooms beginning late August through early September every year. The TMDL study recommended a 15 percent reduction in nitrogen and phosphorus loadings to the reservoir in order for the reservoir to meet its designated beneficial uses (Tetra Tech, 2002).

Ground water flow and subsequent nutrient loading estimates to Pineview Reservoir have been made before (Avery, 1994; Miner et al., 1990; Tetra Tech, 2002; see Chapter 3). The Clean Lakes Study estimated that the annual ground water contribution from the water table aquifer to Pineview Reservoir was approximately 25 x 10^6 m^3 (Miner et al., 1990). Apparently, the ground water flows were estimated based on data from five monitoring wells studied during the 1988 irrigation season but the method used to calculate the ground water flow was not described. Avery (1994) simulated ground water flows in Ogden Valley and estimated that the annual ground water contribution from the water table aquifer to Pineview Reservoir was approximately 33 x 10^6 m^3. Avery's study encountered sparse data problems especially for the water table aquifer and therefore recommended further studies involving more monitoring wells over a longer monitoring period (Avery, 1994). Tetra Tech (2002) used the flow estimated by the Clean Lakes Study to estimate N and P loading to the reservoir. They also recommended that more studies be conducted to fill data gaps.

Chapter 3 reported a collaborative study conducted by the Utah Water Research Laboratory and the Weber Basin Water Conservancy District whose objective was to increase understanding of ground water nutrient loading relative to surface water nutrient loads to Pineview Reservoir. They determined surface and ground water flow rates and nutrient (nitrogen, phosphorus, iron, and dissolved organic carbon) loadings as well as the fate of phosphorus in Pineview Reservoir. They reported that nitrate + nitrite (NO_3 + NO_2) and total dissolved phosphorus (TDP) loading contributions from the water table aquifer to Pineview Reservoir were approximately 22 percent and 2.6 percent, respectively, of the total annual loads. These proportions originated from water table aquifer inflow of 3.4×10^6 m^3 yr^{-1} which only accounted for approximately 2.0 percent of the total annual reservoir inflow. Aquifer heterogeneity resulted in variations in ground water flows among aquifer subdivisions. Large variations in average ground water nutrient concentrations (6 – 310 µg P L^{-1} as TDP and 3.3 – 21 mg N L^{-1} as NO_3 + NO_2) among the aquifer subdivisions were also reported. The combined effect of aquifer heterogeneity and variability in nutrient concentrations resulted in large variations (0 – 2 kg P d^{-1} and 0.06 – 33 kg N d^{-1}) in nutrient loading among the subdivisions (see Chapter 3) The uncertainty and potentially large inaccuracy in the previous studies prompted further study of ground water flow and nutrient loadings in Ogden Valley in order to improve the analysis.

Consequently, a ground water loading analysis was conducted in the Pineview Reservoir area using data from 15 November 2010 through 14 November 2011. The objective of the research was to quantify spatial variability of ground water flow, ground water nutrient concentrations and nutrient transport towards Pineview Reservoir, and to improve the ground water flow and nutrient loading estimates for the reservoir using this information. Achievement of the objectives would provide useful information for management decisions aimed at protecting the reservoir water quality. Knowledge of the spatial variability of the ground water flows, nutrient concentrations and nutrient transport would guide management decisions on whether or not control of nutrient loadings to the reservoir would be achieved by implementing best management practices in zones. The zoning of the best management practices would depend on the extent of the spatial variability.

Approach and methods

Snow is the major form of precipitation in Ogden Valley. High intensity snow melt occurs in the spring resulting in high stream flows, which usually peak in April or May. A normal water year (Oct – Sep) in Ogden Valley's Huntsville area receives a total precipitation of 579 mm (WRCC, 2012). Based on 20 to 24 years of precipitation data from Pineview-Dam weather station (WRCC, 2012), precipitation in the 2009/2010 water year was approximately 18 percent below normal while the 2010/2011 precipitation was

46 percent above normal. Ground water recharge in the valley is mainly from stream flow, snow melt, rainfall and irrigation (Snyder and Lowe, 1998; Tetra Tech, 2002). The irrigation season in Ogden Valley begins by mid-May and ends in late October. Most of the stream flow is diverted for irrigation in summer. This implies ground water flow into Pineview Reservoir constitutes a larger proportion of the total inflows in summer than spring. The principal source of ground water to Pineview Reservoir is probably the shallow water table aquifer which overlies the principal (confined) aquifer (Fig. 14) in the center of the southern part of the valley (Avery, 1994).

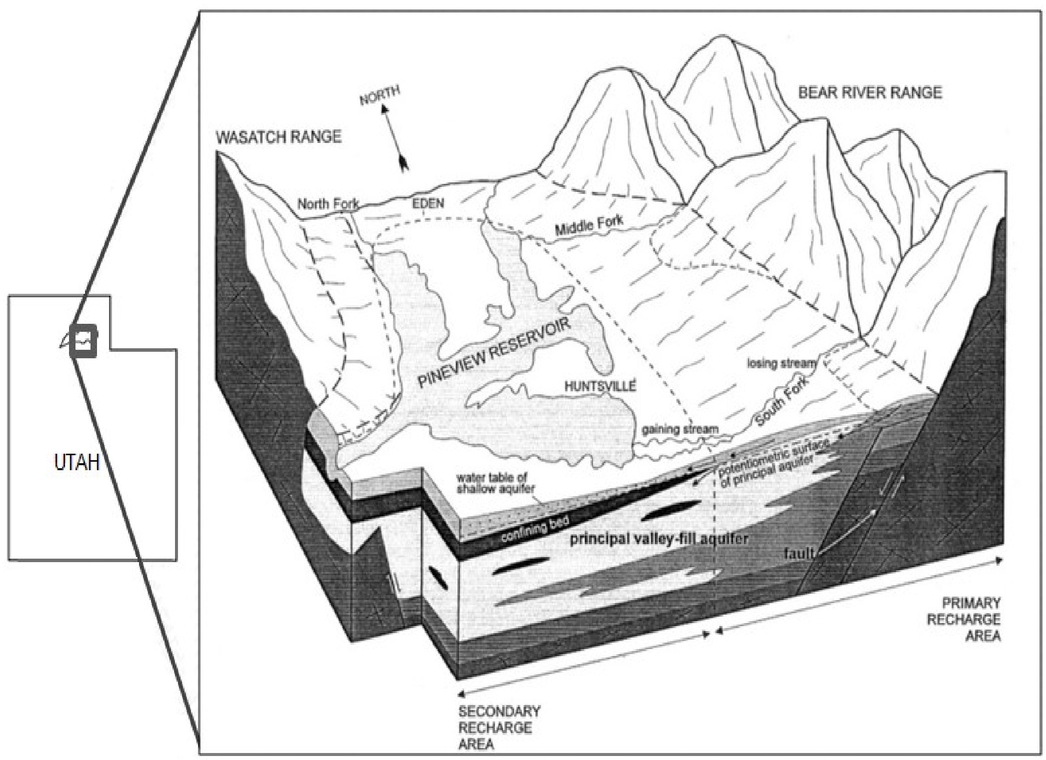

Figure 14. Pineview Reservoir and Ogden Valley hydrogeology (Snyder and Lowe, 1998)

Four monitoring wells were constructed in the vicinity of Pineview Reservoir in addition to the five that were studied by Reuben et al. (see Chapter 3). Two of the additional wells, wells 6 and 7, were located near the shoreline south and west of the reservoir in order to obtain a better representation of these parts of the water table aquifer. Well 8 was constructed 530 m west of the first well located in Huntsville (well 4) because the existing monitoring well had relatively high nutrient concentrations and flows in comparison with most of the other four. Well 9 was situated between wells 3 and 4 in order to refine the resolution for nutrient transport in the area since well 3 had relatively low nutrient loading. Figure 15 shows the well locations. The wells were constructed in a similar manner with the initial five wells (5.1 cm (2 in.) i.d. PVC with a screen mesh size

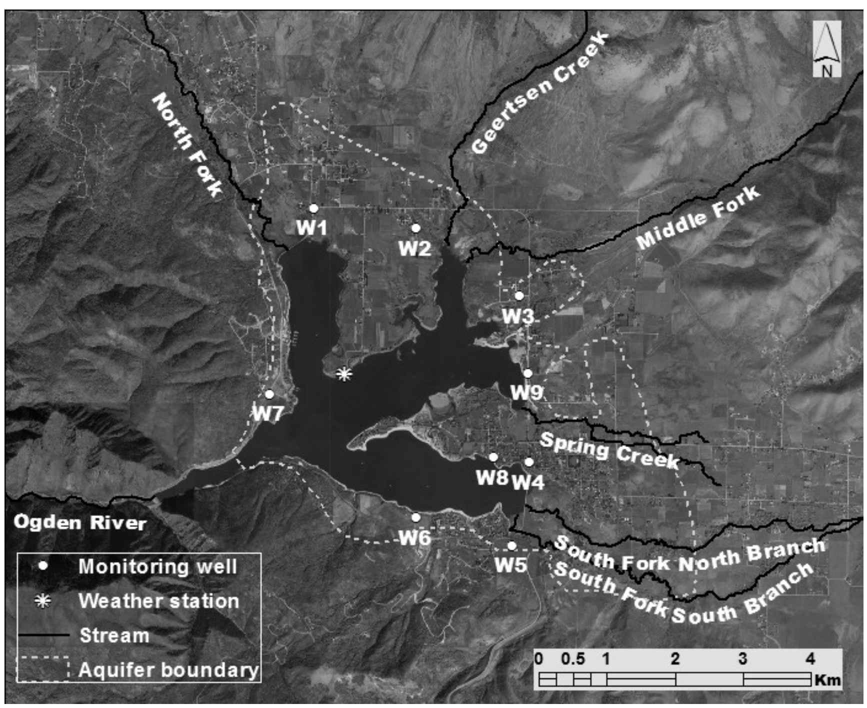

Figure 15. Map (ESRI) showing locations of monitoring wells, streams and weather station. Wells are labeled W1 through W9 while the weather station is represented by an asterisk

of 0.25 mm) except for the screen length which was 3.0 m (10 ft). All the wells fully penetrated the shallow, unconfined aquifer and were constructed as close to the reservoir as possible (Figs. 33 and 34, Appendix B). The average depth for all nine wells was 8.8 m (Table 20, Appendix I). Hydraulic conductivities at each well location were determined from slug tests (Todd and Mays, 2005). Geodetic surveys to establish elevations of the well caps above mean sea level were also conducted.

Nitrate + nitrite nitrogen (NO_3-N), soluble reactive phosphorus (SRP) and total dissolved phosphorus (TDP) concentrations from the nine monitoring wells around Pineview Reservoir were monitored for 198 days from 1 May through 14 November 2011. Total dissolved iron and dissolved organic carbon (DOC) concentrations were also monitored. Ground water pH, electrical conductivity (EC) and dissolved oxygen (DO) were measured onsite using probes and a meter during grab sampling events. Ground water was pumped through polyethylene tubing and a poly (methyl methacralate) chamber using a portable 4.2 cm (1.66 in.) bladder pump. The chamber had O-ring-sealed ports into which the probes were pressed to measure the aforementioned water parameters. Prior to inserting the pump into each well, the apparatus was rinsed by pumping reagent grade (18 MΩ/cm) deionized water through the system until the EC readings were ≤ 2 μS/cm.

Grab samples were collected from a discharge tube connected to the chamber outlet when steady meter readings (≤1 percent change between three successive readings) were observed during continuous well pumping of at least 5 minutes. Similar standard operating procedures (Table 3) to the ones followed by Reuben et al. (see Chapter 3)

Table 3. Laboratory analysis methods (APHA, 1995)

Variable	Method
Soluble reactive phosphorus (SRP)	SM 4500-P B & P E
Total dissolved phosphorus (TDP)	SM 4500-P B & P E
Nitrate + nitrite-nitrogen (NO_3-N)	SM 4500-NO_3^- F
Dissolved organic carbon (DOC)	SM 5310 C
Total dissolved iron	SM 3111 B
pH	SM4500-H+ B
Electrical conductivity (EC)	SM2510 B

were applied during sample collection, preservation, transportation and laboratory analyses. Quality assurance and quality control techniques described in Standard Methods for the Examination of Water and Wastewater (APHA, 1995) and user manuals for respective instruments were closely followed. All instruments used during field measurements were calibrated prior to each field exercise.

The grab samples were tested for serial autocorrelation to ensure that the variances in the means were not influenced by autocorrelation (Salas et al., 1980; Yue et al., 2002). The observed nutrient concentrations (Tables 15 through 18, Appendices D through G, respectively) were linearly interpolated over time to estimate daily nutrient concentrations used to compute daily spatial nutrient loadings.

Water table elevations for the nine wells were monitored twice each day at 6:00 am and 6:00 pm, local time, using pressure loggers (Onset Computer Corp., Pocasset, Mass.). Pressure data from the project's weather station located on the bank of Pineview

Reservoir (Fig. 15) were used to compensate the logger data for barometric pressure (Rasmussen and Crawford, 1997). The compensation was done in HOBOware using Pressure Compensation Assistant (Onset Computer Corp., Pocasset, Mass.). All water table elevations were converted to meters above mean sea level following topographic surveys conducted on each well cap.

Daily elevations for Pineview Reservoir water surface were obtained from the US Bureau of Reclamation online database (Bureau of Reclamation, 2012) and converted from feet (NAD 27/NAVD 29, 12N) to meters (NAD 83/NAVD 88, 12N) above mean sea level using CORPSCON 6.0 software developed by the US Army Corps of Engineers. An attributes table of daily reservoir and water table elevations was spatially joined with a point feature shapefile comprising the nine well locations and 290 reservoir points. The reservoir points were arbitrarily assigned throughout the reservoir to constrain water table contours from crossing the reservoir. Reservoir water surface elevations were assumed to be uniform across the reservoir each day. The spatially joined point feature shapefile and daily locational elevations were interpolated through a kriging procedure in ArcGIS 3D Analyst. A semivariogram model fitted with an exponential equation was applied in the process. The hydraulic gradient cells representing well locations were clipped and the resulting clip used in computing ground water flows and nutrient loading. Hydraulic conductivities were log transformed after observing that a log transform would make the data more normally distributed. Log normal distributions of hydraulic conductivities have been discussed by others (Buckland, 1987; Loáiciga et al., 2006; Verbovšek 2008; Zhai and Benson, 2006;). A kriging procedure was applied to the log transformed hydraulic conductivities and antilogarithms of the results were used in computation of daily Darcy fluxes, flows and nutrient loadings at respective well locations. It was assumed that the spatial distribution of the hydraulic conductivities was constant. The Darcy fluxes, flows and nutrient loadings were also interpolated by kriging to establish the spatial distribution of the respective parameters. Figure 16 summarizes the inputs, processes and outputs of the ground water flow and nutrient loading modeling procedure.

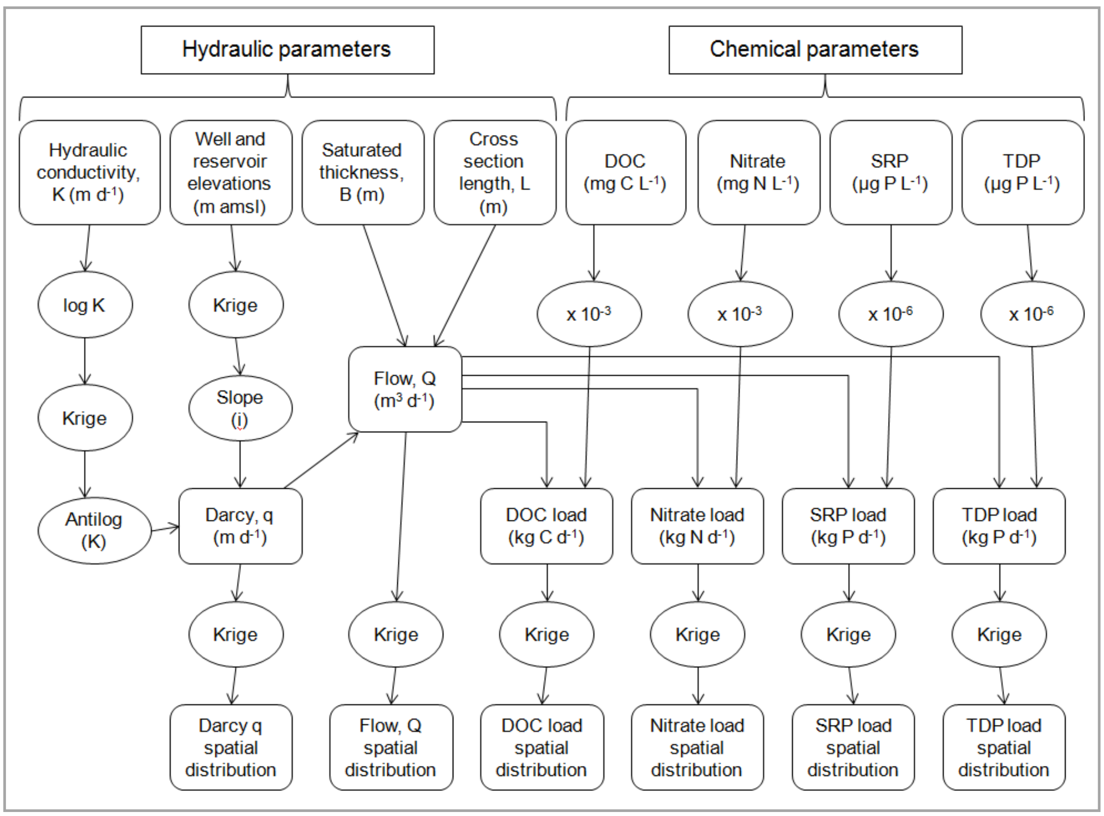

Figure 16. Conceptual diagram for ArcGIS modeling of ground water nutrient loadings. The chemical parameters were measured nutrient concentrations (Appendices D through G) that were linearly interpolated into daily concentrations.

The model calibration variables for respective hydraulic and nutrient loading parameters were determined by iterations. The iterations involved manually altering the exponential model parameters in ArcGIS Model Builder until the model more accurately predicted the measured values for the calibration day (used 3 May 2011 data). Calibrated models were validated by running each model on data from a different date picked arbitrarily. The validation exercise showed that the models more accurately predicted the respective parameters since the differences between the measured and predicted parameters were within ±10 percent. Table 4 is a summary of the calibrated model variables for respective parameters. Kriging (ESRI, 2012) of the daily flow and nutrient loadings in ArcGIS was conducted by running a batch command on Python Scripts generated from the Model Builder. For each kriging output type, the total number of 20 m by 20 m output cells was 47,809 on each of the 198 days.

Darcy flux, ground water flow and nutrient loading results from the kriging were log-transformed to make the data approximately log normally distributed. Kriging output data whose magnitudes were less than or equal to 10^{-6} (m d^{-1}, m^3 d^{-1}, and kg d^{-1}, respectively) were screened out because 1) their contributions to the flows and/or loads were assumed negligible; and 2) most of the data cells with such values were situated in the reservoir. The reservoir was considered a ground water sink because its water surface elevations were below water table elevations in the wells throughout the study period.

Table 4. Exponential model calibration variables applied in the kriging process in ArcGIS

Input parameter	Lag size	Major range	Partial sill	Nugget	Neighbors[b]
Water surface elevations	1530	5754	1500	0	12
DOC, Nitrate, SRP, TDP loads	1530	5754	8	0	12
Hydraulic conductivities (K)	500	5754	2.34[a]	0	9
Darcy fluxes (q)	1530	5754	0.30	0	9

[a] =calibration parameter used to krige log transformed hydraulic conductivities
[b] =number of nearest neighbors used to interpolate the value at a given location or cell. Number of neighbors for kriging K equals number of K values (for nine well locations) from slug tests.

The geometric means and SD of the censored data were used to construct ± 1 SD confidence intervals for each of the 198 days. This was similar to the procedures of Delhomme (1978). Processing of the database files (198 files per output parameter; 47,809 data points per file) generated from the kriging procedure, and construction of the confidence intervals for the geometric means were achieved through Visual Basic for Applications programming.

A multiple linear regression equation (Appendix J) was employed to extrapolate daily median ground water flows and associated nutrient loadings for the first part (15 November 2010 through 30 April 2011) of the study year in order to estimate the spatial median flow and nutrient loadings. The equation was developed by regressing daily median modeled ground water flows against daily reservoir elevations and ground water elevations for 1 May through 14 November 2011. Reservoir elevations and ground water elevations from wells 1, 2, 3, 5 and 7 explained a significant portion of the variance in modeled daily median ground water flows ($p < 0.0001$; $\alpha = 0.05$). The final regression model included only these significant explanatory variables. The coefficient of determination, r^2, for the predicted versus modeled median ground water flows was 0.998; implying that the explanatory variables were adequate predictors. The regression equation was applied using corresponding daily reservoir elevations and ground water elevations from the five selected wells observed from 15 November 2010 through 30 April 2011 to estimate daily median ground water flows. Daily median SRP, TDP and nitrate loadings were computed from the product of linearly interpolated median concentrations and daily median spatial flows modeled using procedures summarized in Figure 16. The daily median spatial flows and nutrient loadings were aggregated into the median flow and respective one-year median nutrient loadings (15 November 2010 through 14 November 2011).

Results and discussion

Ground water quality monitoring in the Ogden Valley's Pineview Reservoir area for 1 May through 14 November 2011 showed that the average TDP concentration for the nine wells was 200 ± 270 μg P L^{-1} (arithmetic mean ± 1 SD). Well 9 generally had the highest TDP concentrations with a median of 673 μg P L^{-1} (Table 5). The median TDP concentration from well 9 was more than twice the concentrations of wells 4 and 5 whose median concentrations were 249 and 304 μg P L^{-1}, respectively. Autocorrelation tests

Table 5. A summary of TDP concentrations from nine monitoring wells

Well	Minimum[a]	Median[a]	Maximum[a]	Mean	RSE[b]
1	5	12	17	12	17
2	14	17	42	20	22
3	25	32	42	32	7.7
4	226	249	443	277	12
5	247	304	318	294	3.5
6	67	154	342	165	23
7	12	19	45	25	23
8	64	107	947	238	60
9	424	673	1,265	727	19

[a]: All concentrations in μg P L^{-1}. The number of sampling events was six. [b]: Relative Standard Error (percent).

conducted following the approach reported by Salas et al. (1980) and Yue et al. (2002) indicated inexistence of serial autocorrelation of nutrient concentrations among the grab samples. Wells 1, 3, 4, 5, and 9 had relatively low variance in TDP concentrations (Relative Standard Error (RSE) ≤ 20 percent) than the other four wells (Table 5). Thus, 55 percent of the wells had relatively low temporal variations in TDP concentrations.

The highest nitrate-N (NO_3+NO_2-N) concentration (28 mg N L^{-1}) during the study period was observed from well 1 on 3 May 2011. The average nitrate-N concentration for well 1 was 10 and median 7.0 mg N L^{-1} (Table 6). The average nitrate N concentration for the nine wells was 4.3 ± 4.3 and the median concentration 3.9 mg N L^{-1}. Wells 2, 3, 4, 6, and 8 had relatively low temporal variance in nitrate-N concentrations (RSE ≤ 20

Table 6. A summary of $NO_3 + NO_2$-N concentrations from nine monitoring wells

Well	Minimum[a]	Median[a]	Maximum[a]	Mean	RSE[b]
1	3.6	7.0	28	10	37
2	3.9	5.0	5.3	4.7	5.2
3	2.5	3.0	3.3	2.9	4.2
4	5.3	6.6	7.2	6.4	4.5
5	1.2	4.6	8.8	4.3	26
6	0.3	0.5	0.7	0.5	8.2
7	0.4	0.6	1.4	0.7	23
8	0.1	4.2	4.9	3.7	20
9	2.0	2.9	13	4.9	37

[a]: All concentrations in mg N L^{-1}. The number of sampling events was six. [b]: Relative Standard Error (percent).

percent) than the other four wells (Table 6). Two observations from well 1 (on 3 May and 14 November 2011, respectively) and one from well 9 (3 May 2011) had nitrate-N concentrations that exceeded the drinking water Maximum Contaminant Level of 10 mg N L^{-1} set by the USEPA (2012). These nitrate N concentrations may reflect the dynamic nature of nitrate-N loading in the north western and eastern parts of the water table aquifer. Reuben et al. (see Chapter 3) also reported that 28 percent of the samples from well 1 exceeded the Maximum Contaminant Level for nitrate.

Well 9 also had relatively high median SRP concentrations (452 μg P L^{-1}) followed by wells 5 and 4 whose median concentrations were 268 and 209 μg P L^{-1}, respectively. Large spatial variations were observed in DO concentrations. Well 9 registered the lowest average DO concentration of 1.98 mg L^{-1} and well 8 had the lowest individual DO concentration of 0.11 mg L^{-1}. No anoxic conditions (DO < 2 mg L^{-1}) were observed in the other seven wells. Autocorrelation tests for all nine wells showed that no serial autocorrelation existed among SRP concentrations among sampling days. Low temporal variations in SRP concentrations were observed in all wells as evidenced from the low RSE values presented in Table 7. Corresponding increases in SRP, TDP and iron concentrations occurred in well 9 on both days during which anoxic conditions existed while well 8 did not experience such a phenomenon. Statistical difference existed among the mean DOC concentrations for well 8 and 9 (as witnessed from their non-overlapping 67 percent confidence intervals (4.6±1.7 and 8.6±1.8 mg C L^{-1}, respectively).

Table 7. A summary of SRP concentrations from nine monitoring wells

Well	Minimum[a]	Median[a]	Maximum[a]	Mean	RSE[b]
1	7.1	9.4	11	9.3	5.7
2	4.7	6.1	6.8	5.8	6.3
3	22	24	26	24	2.4
4	200	209	224	210	1.6
5	143	268	288	248	8.9
6	60	100	118	99	8.7
7	4.7	7.5	9.4	7.4	8.8
8	52	87	108	85	9.1
9	191	452	493	394	13

[a]: All concentrations in µg P L^{-1}. The number of sampling events was six. [b]: Relative Standard Error (percent).

The confidence intervals (mean ± 1 SD) for the geometric means of ground water flows from the water table aquifer towards Pineview Reservoir showed much spatial variation with the largest confidence interval of 1,518 – 5,077 m^3 d^{-1} on 6 May and the smallest (447-1,814 m^3 d^{-1}) on 27 June 2011 (Fig. 17). The wide confidence interval observed in spring may be attributed to rapid increases in reservoir elevations due to spring runoff emanating from snowmelt in the mountains and spatially variable changes in water table elevation. These fluctuations may have caused large variations in the hydraulic gradients due to differences in cell locations relative to the reservoir shoreline.

The impact of reservoir elevation changes may have been less for cells farther away from the reservoir than those relatively close. The relatively stable confidence

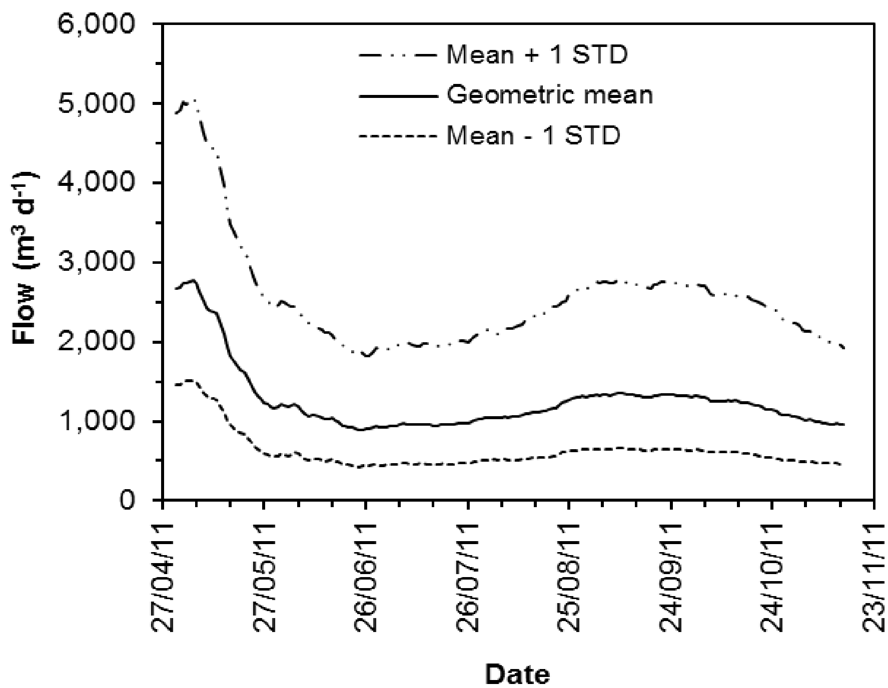

Figure 17. Spatial variation of estimated daily ground water flows towards Pineview Reservoir. The variations are presented as the daily geometric means plus or minus one standard deviation. The variation is based on computations summarized in Figure 16

intervals observed in summer may be attributed to more stable hydraulic gradients experienced during this time of the year due to relatively slow drawdown of the reservoir and stabilization of ground water elevations due to irrigation. The advent of a gradual increase in the confidence interval in fall could be due to joint effects of a more rapid decline in reservoir elevation and variations in water table elevations (Fig. 18) due to variable cumulative effects of deep percolating irrigation water. The decline in the reservoir elevations resulted from water withdrawals for irrigation during the period when there were relatively low stream inflows due to stream diversions for irrigation. This hypothesis is supported by the decline in the mean ground water flow and the confidence interval at the end of fall, a time when the irrigation season and consequently, stream diversions end in Ogden Valley.

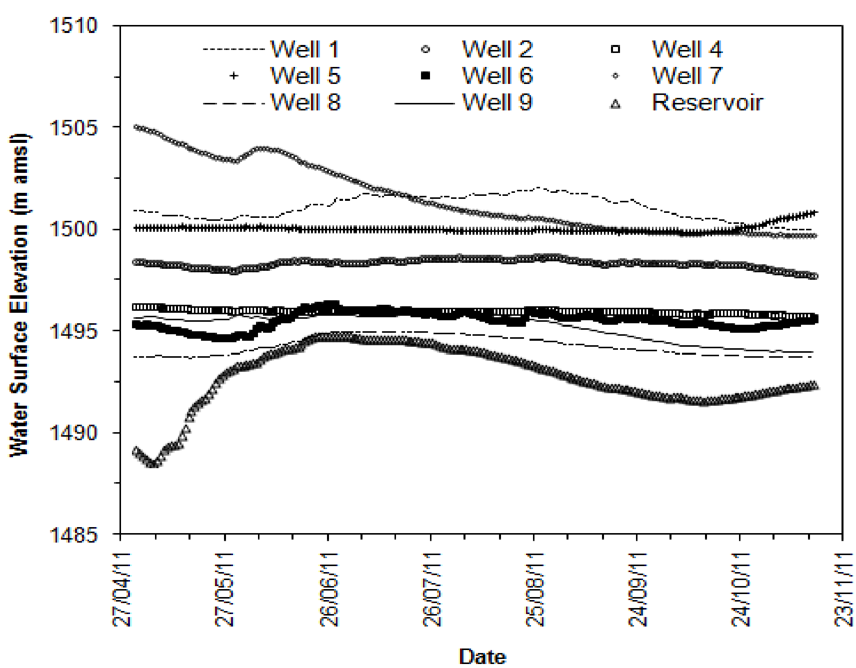

Figure 18. Daily water elevations for Pineview Reservoir and ground water monitoring wells. Well 3 was left out because its profile was similar to well 2

The median TDP concentrations in wells 1, 2, 3, and 7 were relatively lower than from the other five wells. Well 9's median TDP concentration was more than 4-fold higher than well 8, and more than 20-fold higher than wells 1, 2, 3, and 7. Wells 4 and 5 were more than twice as much as well 8. These differences reflected the spatial variability of TDP in the valley. The confidence interval for TDP loading generally followed a similar trend to that of ground water flows. The widest TDP confidence interval (0.032 - 0.400 kg P d^{-1}) occurred on 6 May and the narrowest (0.014 - 0.093 kg P d^{-1}) on 25 June 2011. These two events (closely) coincided with similar occurrences for flows and may signify the overall influence of the spatial distribution of ground water flow on the distribution of TDP loading towards Pineview Reservoir and any similar reservoir or lake.

A sharp increase in TDP confidence intervals was observed in fall while the intervals in summer were narrow but short-lived (Fig. 19) compared to the narrow intervals for the flows. The sharp increase in the TDP loading confidence interval could to a greater extent be attributed to relatively higher TDP concentrations observed from wells 8 and 9 (Table 15, Appendix D). Wells 4 and 5 may also have influenced the increase to some extent since their TDP concentrations had a generally gentle rise during this period. The wide confidence intervals in fall imply that significant spatial variability of TDP loading may exist in Ogden Valley. The variability may be attributed to differences in loading of TDP from both agriculture and domestic wastewater in different locations in the watershed.

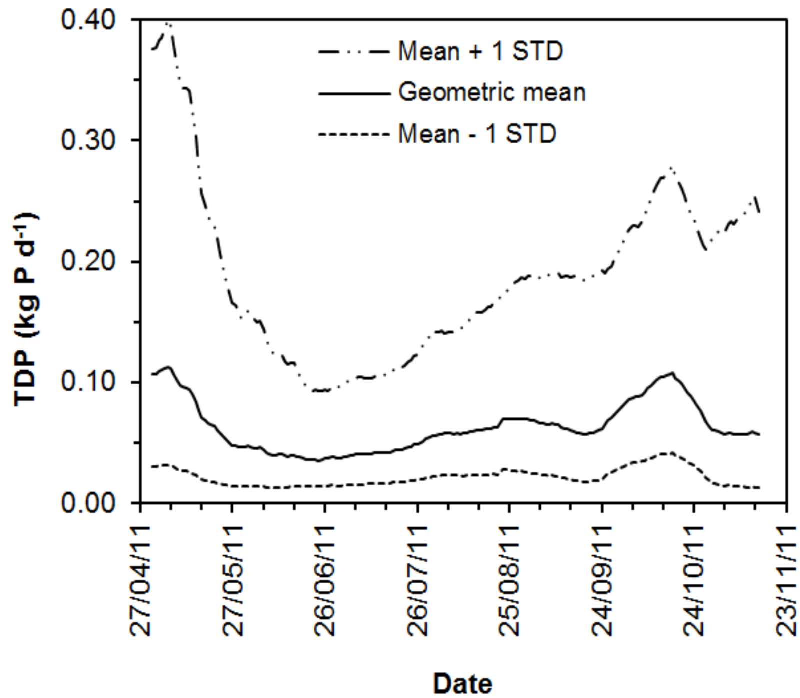

Figure 19. Spatial variation of daily ground water TDP loadings to Pineview Reservoir. Corresponding geometric means and standard deviations of log TDP loadings were either added or subtracted from each other. Antilogarithms were taken on the results

The confidence interval for SRP loading had a similar general trend to that of TDP whereby short-lived narrow confidence intervals occurred in summer and the interval widened in fall (Fig. 20). The widest and narrowest SRP confidence intervals of 0.017- 0.331 and 0.008 - 0.073 kg P d^{-1} were, respectively, observed on 6 May and 26 June 2011. A more distinct difference between SRP and TDP confidence interval trends occurred in fall during which the SRP confidence interval had a plateau top while TDP had peaks and a valley. These differences may signify the spatial variation in loading proportions of SRP to dissolved organic P (DOP) from different locations in the watershed as can be seen in Figure 21.

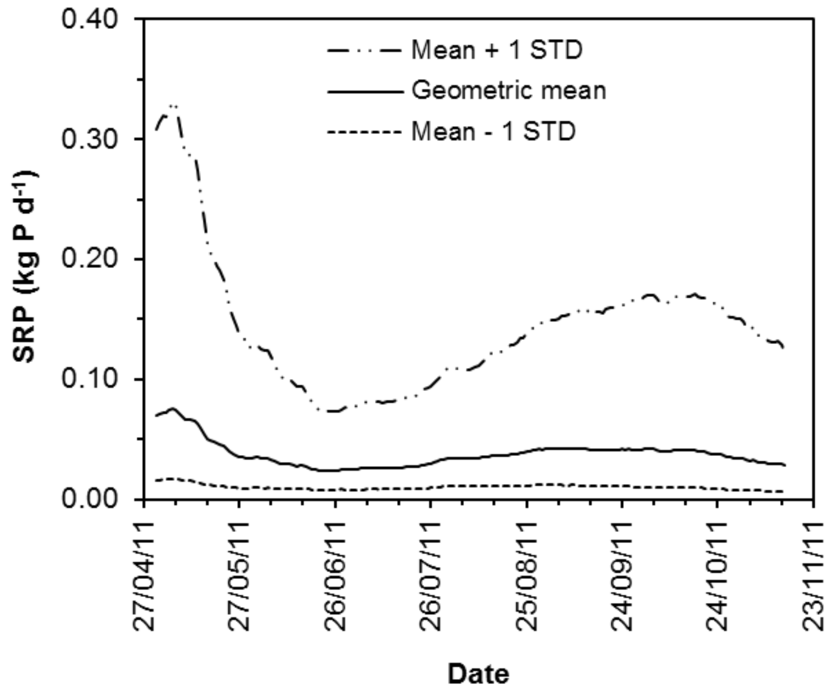

Figure 20. Spatial variation of daily ground water SRP loadings to Pineview Reservoir. Corresponding geometric means and standard deviations of log SRP loadings were either added or subtracted from each other. Antilogarithms were taken on the results

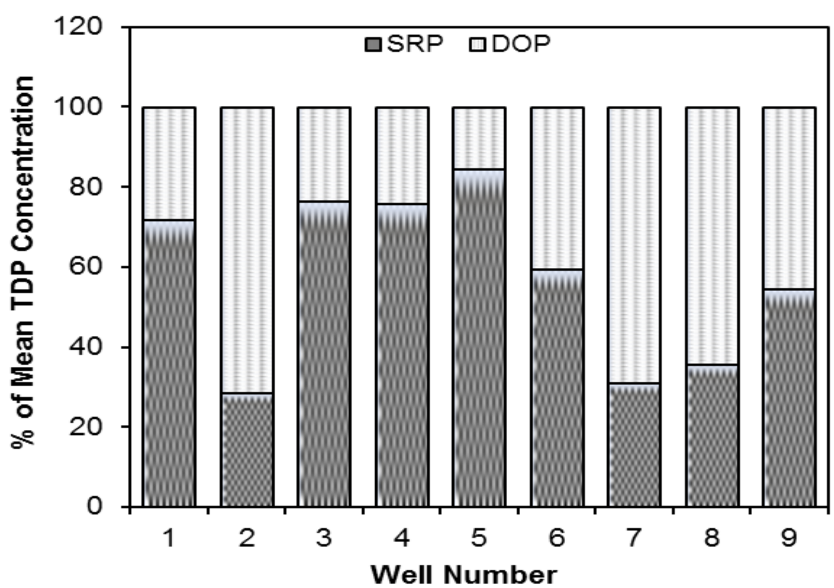

Figure 21. Relative SRP and DOP proportions from nine Ogden Valley monitoring wells. SRP and TDP were measured while DOP was estimated from the difference between the two

The confidence intervals for nitrate ($NO_3 + NO_2$-N) loadings had a similar trend to that of the flows except for a narrower interval and a relatively stable lower limit from summer through fall (Fig. 22). The relatively stable lower limit could be attributed to a more stable geometric mean (or a relatively stable variance) due to low variability in nitrate concentrations among the ground water cells. The widest interval (6.38 - 39.1 kg N L^{-1}) was observed on 3 May and the narrowest (0.954 - 8.12 kg N L^{-1}) on 26 June 2011.

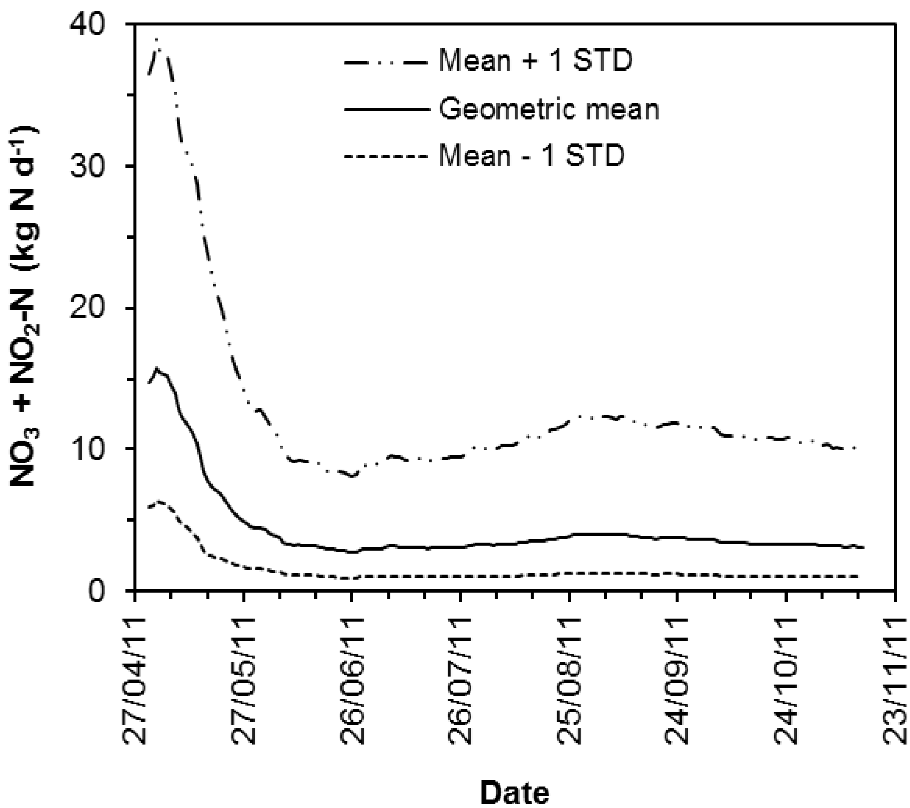

Figure 22. Spatial variation of ground water $NO_3 + NO_2$-N loadings to Pineview Reservoir. Corresponding geometric means and standard deviations of log $NO_3 + NO_2$-N loadings were either added or subtracted from each other. Antilogarithms were taken on the results

Daily median ground water flows (for 15 November 2010 through 14 November 2011) determined from the kriging procedure results and from the multiple linear regression extrapolations are presented in Figure 23. The daily median ground water flows ranged from 0.001 x 10^6 (14 November 2011) to 0.003 x 10^6 m^3 d^{-1} (16 April 2011) with an annual sum of 0.6 x 10^6 m^3. The reservoir elevation on 14 November was almost three meters higher than its elevation on 16 April 2011 while the average ground water elevation on the former day was approximately 3 m lower than the latter. The average ground water hydraulic gradient for 14 November was approximately half that on 16 April. The results verify the influence of both ground water and reservoir elevations on the flows in Ogden Valley.

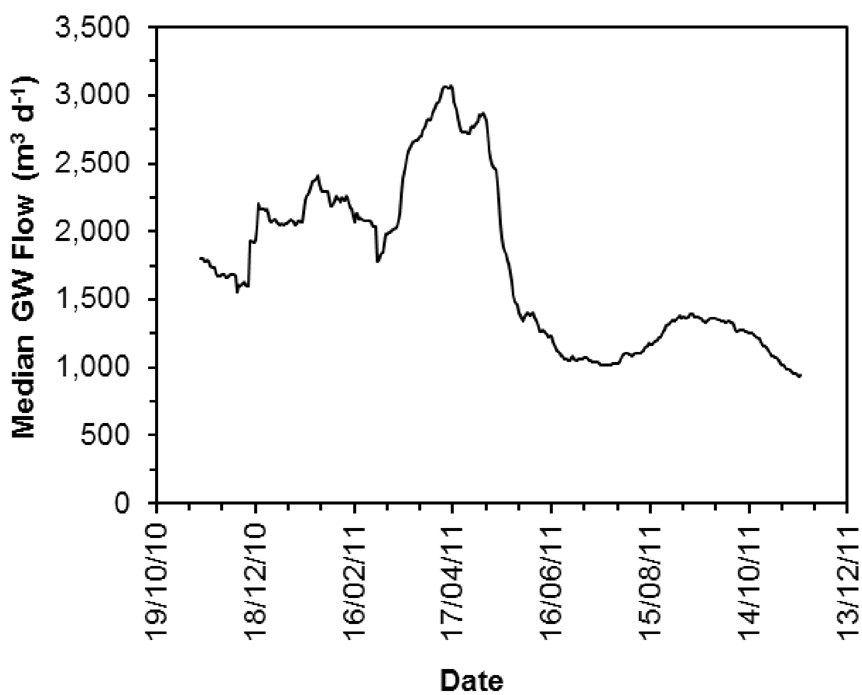

Figure 23. Median ground water flows for 15 November 2010 through 14 November 2011. Daily median flows for 1 May through 14 November 2011 were computed from kriging results. Flows prior to 1 May 2011 were estimated from a multiple regression equation using daily reservoir and well elevations

The daily median SRP and TDP loadings through the water table aquifer to Pineview Reservoir computed from the product of the daily median flows and corresponding median SRP and TDP concentrations are presented in Figure 24. The annual total median SRP and TDP loadings were 50 and 62 kg P, respectively. The general trends of both SRP and TDP median daily loadings were similar to that of the daily median ground water flows. More pronounced differences in the trends existed during the last quarter of the study period. The differences could be attributed to large variations in nutrient concentrations during this quarter as depicted from Figures 19 and 20. Ground water return flow from irrigated land may have influenced these nutrient concentration variations.

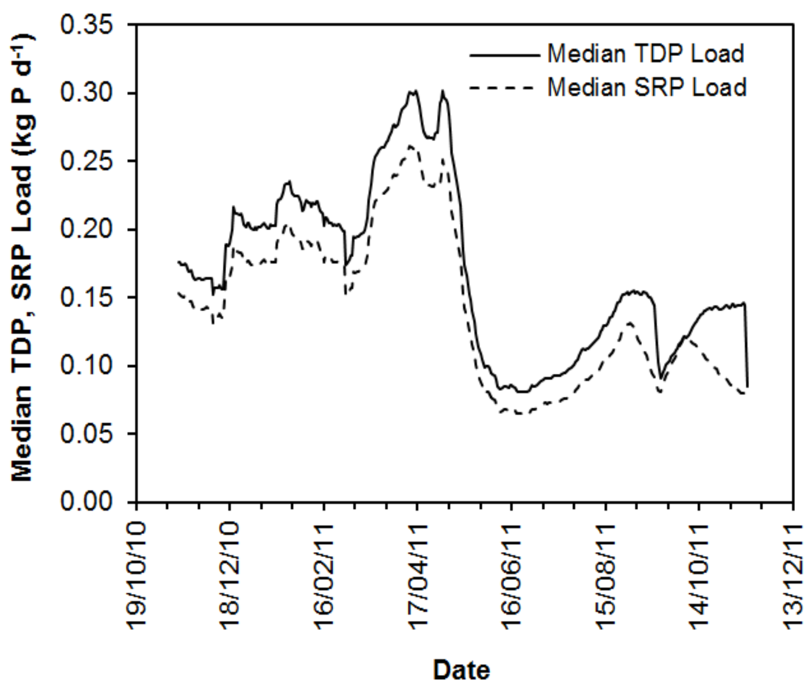

Figure 24. Daily median SRP and TDP ground water loadings to Pineview Reservoir

Figure 25 represents median daily loadings of NO_3+NO_2-N to Pineview Reservoir through the water table aquifer. The annual total for the median NO_3+NO_2-N loadings for the period from 15 November 2010 through 14 November 2011 was approximately 2,460 kg. A similar trend to that of SRP and TDP occurred during the first half of the period. Irrigation return flow through ground water appeared to have a large influence on the phosphorus and nitrate loadings in fall 2011 as evidenced from the zigzag fashion of respective daily median loadings. This is because the variations coincided with the irrigation season. The variability in the spatial distribution of the ground water flows and nutrient loadings in Ogden Valley may also be attributed to differences in ground water hydraulic gradients at various locations within the water table aquifer's drainage area. The influence of hydraulic conductivities on the spatial variability of the flows and

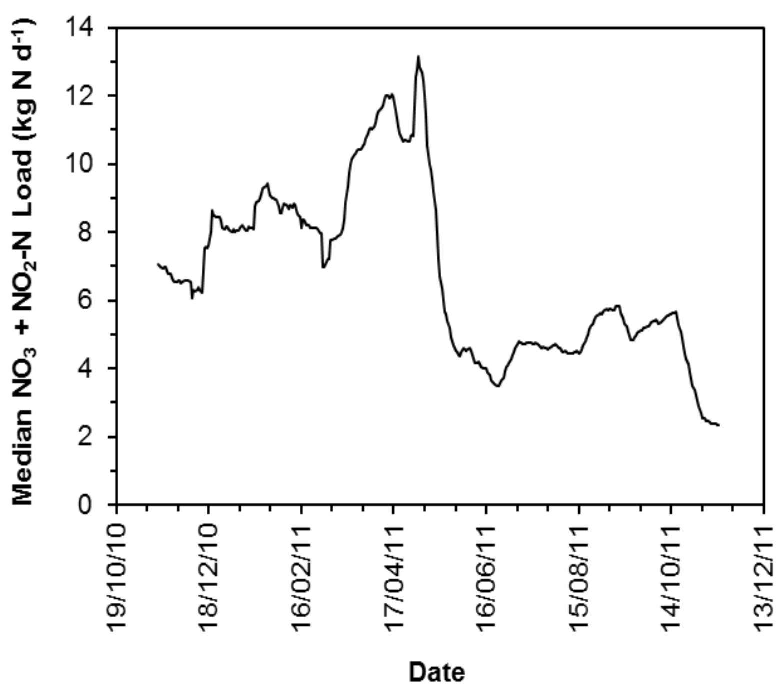

Figure 25. Daily median NO_3+NO_2-N ground water loadings to Pineview Reservoir

nutrient loads may have been significant because the conductivities varied largely regardless of how close the wells were. For example, the hydraulic conductivity at well 4 was 22 m d^{-1} while well 8 had 0.86 m d^{-1} yet these wells were only 530 m apart. Time and resource constraints prevented the study from conducting a detailed exploration on the spatial distribution of the hydraulic conductivities and presence and effect of preferential flow paths. The uncertainty associated with these two parameters is believed to have been reduced or factored out by aggregation of low and high flows.

The overall median TDP concentration (24.8 μg P L^{-1}) for wells 1 through 5 during the current study was similar to the overall median concentration (24.6 μg P L^{-1}) observed in the study by Reuben et al. (see Chapter 3). Inclusion of four additional monitoring wells to the five that Reuben et al. (see Chapter 3) studied, raised the median TDP concentration by three-fold. The median SRP concentration (31.8 μg P L^{-1}) for five wells was approximately half that observed by Reuben et al. (see Chapter 3) and approximately three-fold lower than the median for nine wells. The similarity in median TDP concentrations observed in the current study from wells 1 through 5 with those from Reuben et al. (see Chapter 3) imply that the five wells had minimal temporal variations from 1 May 2010 through 14 November 2011. The comparison of medians from five wells with those from nine wells implies that large variations existed in SRP and TDP loadings among different locations in the watershed. The median NO_3+NO_2-N concentration (5.0 mg N L^{-1}) for five wells in the recent study was similar to the median for nine wells (3.9 mg N L^{-1}) and to the one observed in the preceding study (4.0 mg N L^{-1}) by Reuben et al. (see Chapter 3). The similar central tendency of the NO_3+NO_2-N concentrations may imply similarity in nitrate loading trends from different locations in the watershed.

The 67 percent confidence intervals (geometric mean ± 1 SD) for the ground water flows from the water table aquifer towards Pineview Reservoir showed much spatial variation with the largest confidence interval of 1,518 – 5,077 m^3 d^{-1} on 6 May and the smallest (447-1,814 m^3 d^{-1}) on 27 June 2011 (Fig. 17). The wide confidence interval observed in spring may be attributed to rapid increases in reservoir elevations due to spring runoff emanating from snowmelt in the mountains and spatially variable changes in water table elevation. These fluctuations may have caused large variations in the hydraulic gradients due to differences in cell locations relative to the reservoir shoreline.

The spatial variability in nutrient loadings may also be attributed to variability in nutrient contributions from non-point sources (agriculture and domestic wastewater) from various locations in the watershed. For example, well 9 was located close to a less dense residential area dominated by irrigated agriculture upland. Well 4 was situated down gradient of a relatively high density residential area; downtown Huntsville. Irrigated crop production occurred within and east of this town and it is possible that some of the nutrients observed in well 4 (and possibly well 8) originated from the cropland. Well 5 was located in an area surrounded by approximately 50 hectares of irrigated land and was within a radius of 300 m from a less dense residential area. Well 7 was located approximately 160 m down gradient from a predominantly residential area. Domestic wastewater in the residential areas was disposed through onsite wastewater treatment systems. The nutrient loadings observed from the study could therefore be attributed to both domestic and agricultural sources as reported by Reuben et al. (see Chapter 3). The rates of nutrient loadings from these sources appear to vary both spatially and seasonally. Spring snowmelt and runoff appeared to have the greatest influence on the spatial variability of nutrient loadings to the reservoir. This was probably because snowmelt events resulted in deep percolation of water and consequently flushed nutrients into ground water. The variability in space in the nutrient loadings were influenced by the differences in nutrient sources and differences in hydraulic conductivities.

These differences support the hypothesis that large spatial variations in nutrient concentrations exist across the water table aquifer. This implies that more monitoring wells may be required in order to have a better resolution of ground water nutrient concentrations (and subsequent nutrient loadings) around Pineview Reservoir. Use of just a few monitoring wells may either over or under estimate ground water nutrient loadings to the reservoir due to the spatial variability of the nutrient concentrations. The study managed to capture primary evidence of the spatial variations in nutrient concentrations. Need for more frequent monitoring of the ground water concentrations was not established because temporal variations in nutrient concentrations were generally very minimal.

It is common practice to use a deterministic approach in studying ground water because the method is more straightforward than a spatially variable approach. However, the deterministic approach is deficient in dealing with flow and nutrient loading variations resulting from the non-homogeneous nature of the explanatory variables. This implies that the deterministic approach is likely to either over or under-estimate the response variable than the spatial variable approach would do. The disadvantage with the spatial variable approach is that ground water flow variables, e.g. hydraulic conductivity, are usually not normally distributed in nature. It is common practice to log-transform hydraulic conductivities to make the data more normally distributed. This practice may adversely affect the upper limits of the hydraulic conductivity values of the distribution since lognormal Probability Density Functions are reported to filter out upper bounds (Loáiciga et al., 2006). The present study did not experience this problem since the upper limit of the measured hydraulic conductivities was only 2 percent higher than its corresponding kriged value. The spatial variable approach taken in this study was able to better quantify and characterize the nature of ground water flow and nutrient loading in Ogden Valley and may be applicable to similar environments elsewhere.

Knowledge of the spatial distribution of ground water flow and subsequent nutrient loadings to Pineview Reservoir and other surface water bodies would be helpful in the development and implementation of best management practices to help reduce eutrophication. This is because targeted best management practices would be employed for specific parts of the watershed depending on the degree and source of the spatially distributed nutrient loads.

Conclusion

The spatial distribution of ground water flow parameters and nutrient loading towards a reservoir in an irrigated mountain valley have been quantified and characterized. Large variations were observed in the spatial distribution of the flows, total dissolved phosphorus and nitrate-nitrogen loadings. The spatial variations could be generally attributed to the dynamic nature of the influence of reservoir elevations and irrigation return flow on hydraulic gradients, spatial variations of nutrient sources, and hydraulic conductivities. At least five out of the nine wells sampled in this study had relatively low temporal variation of respective nitrate-N, SRP, and TDP concentrations. This implied that the need for high frequency monitoring of the nutrient concentrations may be low relative to what was anticipated prior to the study.

Hydraulic gradients and nutrient loadings were strongly related to snowmelt and irrigation. Snowmelt influenced spatial variation of hydraulic gradients through rapid increases in reservoir elevations. Resonance of the hydraulic gradients to changes in the reservoir elevations varied among wells depending on their geographic locations relative to the reservoir shoreline. Variations in the spatial distribution of ground water flow in summer and fall were related to both the influence of irrigation water accumulation on ground water elevations and reservoir water surface elevation decline. The influence of irrigation return flow through ground water was evidenced from the coincidence of the rise in nutrient loading spatial variations with the irrigation season. Spring snowmelt in an intermountain irrigated and rural community valley acts as a strong driving force for nutrient transport through ground water. This was evidenced from the higher nitrogen and phosphorus loadings observed during high snowmelt period than at any other time of the year.

Some degree of uncertainty surrounded the interpretation of the influence of hydraulic conductivities on the spatial distribution of the flows and nutrient loading because a hydraulic conductivity range of 0.86 to 22 m d^{-1} was observed from the nine sampled wells. Probable existence of preferential flow paths in the aquifer system is another uncertainty not addressed in the study. These uncertainties may have affected the accuracy in estimating the ground water flows and nutrient loadings. The overall effects of these uncertainties have probably been minimized by aggregative effects of low and high flows in the estimates. Future studies need to increase the number of wells in order to gain a better description of the spatial variations in the conductivities and minimize the uncertainty due to preferential flow.

Much spatial variation in flows was observed with the largest 67 percent confidence interval of 1,518 – 5,077 m^3 d^{-1} on 6 May and the smallest (447-1,814 m^3 d^{-1}) on 27 June 2011. The wide confidence interval observed in spring may be attributed to rapid increases in reservoir elevations due to spring runoff emanating from snowmelt in the mountains and spatially variable changes in water table elevation. These fluctuations may have caused large variations in the hydraulic gradients due to differences in cell locations relative to the reservoir shoreline.

Application of the spatial variable approach helped incorporate the nature of spatial variations of the hydraulic gradients, flows and nutrient sources in estimation of ground water flows and nutrient loadings in a water table aquifer. The spatial characteristics of the flow and nutrient loadings could not have been captured if just a few localized wells were used to estimate the flows. It is imperative to know the spatial distribution of the flow and nutrient loading parameters in order to have a better estimate of the flows and loadings. This was evidenced from the highly variable nature of the spatial distribution of the flows and nutrient loadings observed in the study.

Ground water flow and nutrient loading estimates from previous studies have been improved and a better understanding of ground water flow and nutrient loading distribution established. Previous studies in which ground water flows and nutrient loadings from the shallow unconfined aquifer were estimated did not take the spatial variability of these parameters into consideration. This may have resulted in either over- or underestimation of the ground water flows and their subsequent nutrient loadings because the present study has shown that there is large spatial variability in the hydraulic characteristics of the aquifer and nutrient loading sources. For example, the ground water flow estimate of 25 x 10^6 m^3 made by the Clean Lakes Study (Miner et al., 1990) was more than 13 times higher than the annual sum of the upper limits of the daily 67 percent confidence intervals. The former flow estimate may have been affected by uncertainties due to the spatial variability of the aquifer hydraulic properties. The current study has also provided more accurate estimates of the flows and nutrient loadings because it involved high frequency monitoring of the ground water elevations in the shallow, unconfined aquifer surrounding the reservoir.

The new ground water flow and nutrient loading estimates may provide insight to managers, town planners, and water users on how the reservoir water quality can be protected or improved through implementation of best management practices that are tailor made for different zones in Ogden Valley. Implementation of the best management practices through the zonal approach may minimize the cost of the interventions since specific zones would have specific interventions for controlling nutrient loading to ground water and its subsequent transport to Pineview Reservoir.

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CHAPTER 5

ESTIMATED GROUND WATER NITRATE LOADINGS FROM LAWNS, IRRIGATED CROPLAND, AND ONSITE WASTEWATER

Abstract

Nitrate-nitrogen loadings from irrigated croplands, lawns, and onsite wastewater drain-fields were simulated using the NLEAP-GIS 4.2 model in Ogden Valley, Utah. The study determined the influence of domestic waste water and nitrogen fertilizers applied to lawns and fields on nitrate + nitrite N (nitrate-N) loadings to the shallow, unconfined aquifer in the drainage area of the South Fork of the Ogden River. Ground water flow data from the preceding study were used to estimate ground water nitrate-N concentrations from the NLEAP-GIS 4.2 simulated leaching losses. Measured soil residual nitrate-N for soil cores from two wells were compared with NLEAP simulated soil residual. The study showed that the annual leaching rates from the drain-fields and the lawns were, respectively, more than 2.6- and 1.1-fold higher than the croplands. Total leaching losses from the croplands and lawns were, respectively, 70- and 50-fold higher than total loads from drain-fields. Lawns and drain-fields had lower total leaching losses than the cropland because the total area was lower than that of the cropland. The model predicted that a 50 percent reduction in lawn fertilizer application rate would result in a decline in leaching that was 4-fold higher than that from 50 percent reduction in irrigation water application rate. The potential for using NLEAP in estimating soil residual nitrate-N and ground water nitrate-N concentrations was deemed high. Estimated soil residual

nitrate-N (4.0 mg N L^{-1}) was similar to the average observed residual nitrate-N whose 95 percent confidence interval (C.I.) (arithmetic mean ± 2 SD) was 0 – 8.7 mg N L^{-1}. The 95 percent C.I. (1.6 – 2.2 mg N L^{-1}) for the estimated nitrate-N concentrations overlapped with the 95 percent C.I. (1.2 – 9.0 mg N L^{-1}) for the nitrate-N concentrations measured from two wells in the study area. The study recommended 1) inclusion of lawn management in pollution control programs; 2) implementation of best management practices for croplands; 3) management of septic systems and drain-fields; and 4) building a lawn management scenario into NLEAP-GIS 4.2.

Introduction

Water quality deterioration in lakes and reservoirs is usually attributed to cyanobacteria and algae growth resulting from increasing concentrations of nitrogen (N) and phosphorus (P). Hein (2006) reported on this problem in lakes and reservoirs in the U.S. intermountain west. Cyanobacteria and algae blooms impede beneficial uses of the surface water bodies including fishing, boating, swimming and aesthetic enjoyment. Pineview Reservoir is one of the reservoirs in the intermountain west whose beneficial uses have been affected by the blooms. The reservoir is designated a cold water aquatic life habitat but it is managed as a warm water aquatic habitat due to dissolved oxygen and temperature, impairment (Tetra Tech, 2002). Other beneficial uses of the reservoir include irrigation water supply, summer municipal water supply, hydro-electric power generation, swimming, and boating.

Pineview Reservoir is located in Ogden Valley, approximately 11 km east of Ogden City and adjacent to Huntsville Town in Weber County, Utah. The reservoir has a storage capacity of approximately 140 million m^3, a surface area of 1200 ha and a maximum depth of 25 m (Weber Basin Water Quality Management Council, 1990; Winkelaar, 2010). Pineview Reservoir receives water from both surface and ground water sources. Most of the reservoir's surface water inflows come from Geertsen Creek, Spring Creek, and the Ogden River's North, Middle, and South Fork tributaries. The downstream reach of the South Fork of the Ogden River splits into two tributaries known as the North and South Branches (N.B. and S.B.) prior to entering the reservoir. Ground water inflows are mainly from the shallow, unconfined (water table) aquifer which is separated from the underlying confined aquifer by a relatively impermeable silt-clay layer (Avery, 1994; Snyder and Lowe, 1998; Tetra Tech, 2002).

Pineview Reservoir's Total Maximum Daily Load (TMDL) study, completed in 2002, stated the need to reduce nitrogen and phosphorus loadings to the reservoir in order to meet its beneficial use. The intended N and P reduction goal was to be achieved through control of nutrient loadings from irrigated land, on-site wastewater treatment systems, livestock manure, and rangeland. As is often the case in TMDL studies, the data used in the Pineview Reservoir study were sparse and this limited the information from which the study could draw its conclusions. The study recommended that more studies be conducted to further understand the surface and ground water contributions towards nutrient loadings to Pineview Reservoir (Tetra Tech, 2002).

The Utah Water Research Laboratory collaborated with the Weber Basin Water Conservancy District in ground and surface water studies aimed at deepening understanding of proportionate contributions of ground and surface water towards N and P loadings to Pineview Reservoir (see Chapter 3). Reuben et al. (see Chapter 3) reported that the water table aquifer nitrate-N ($NO_3 + NO_2 - N$) and total dissolved phosphorus loadings to Pineview Reservoir were, respectively, 22 percent and 3 percent of the total annual loads but that it contributed only 2 percent of the total reservoir inflows. Large variations were observed in the ground water nutrient loadings from different locations in the water table aquifer and were attributed to the variations in both aquifer heterogeneity and nutrient concentrations.

Research findings by Reuben et al. (see Chapter 3) prompted Reuben and Sorensen (2012) to conduct a study that employed GIS kriging techniques in assessing the spatial distribution of ground water flows and nutrient loadings from the water table aquifer to Pineview Reservoir. They estimated daily flows that ranged from 447 ± 1,814 to 1,518 ± 5,077 m^3 (mean ± 1 SD) for the period from 1 May 2010 through 14 November 2011. Reported median ground water flows for 13 November 2010 through 14 November 2011 ranged from 939 to 3,063 $m^3\ d^{-1}$. The study by Reuben and Sorensen (2012) reported the existence of large spatial variations in ground water flows and nutrient loadings. Snowmelt in spring and irrigation return flow in late summer and early fall were believed to influence both ground water flows and nutrient loadings (Reuben and Sorensen, 2012). The seasonal nutrient loading variations observed by Reuben and Sorensen (2012) implied that irrigated agriculture and other non-point nutrient sources such as lawns and onsite wastewater treatment systems played a major role in supplying nutrients to ground water.

Hydraulic conductivities for two ground water monitoring wells (wells 4 and 8, Fig. 26) located in Huntsville Town (sector 4-8) represented the lower (0.86 m d^{-1}) and upper (22 m d^{-1}) limits of the hydraulic conductivity range observed in all nine water table aquifer wells surrounding the reservoir (Reuben and Sorensen, 2012). The study by Reuben and Sorensen (2012) also showed that well 8 registered the lowest minimum nitrate-N concentration while nitrate-N concentrations from well 4 were consistently higher than most of the other eight monitoring wells.

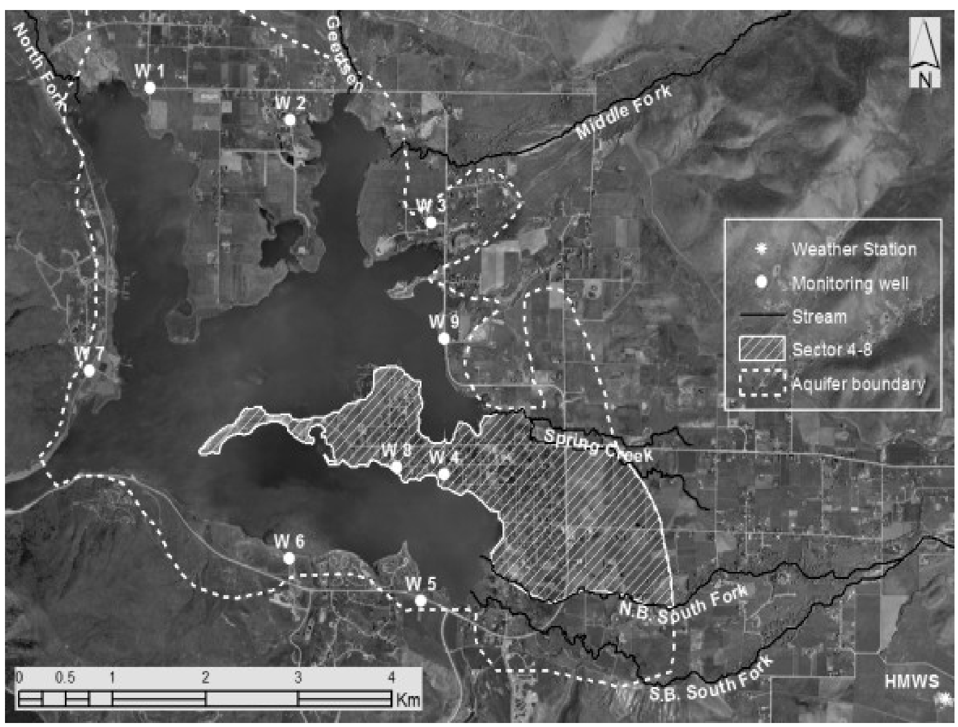

Figure 26. Map of Ogden Valley (ESRI) showing sector 4-8, ground water monitoring wells (W1 through W9), streams and the Huntsville Monastery Weather Station (HMWS). Map modified from Reuben et al. (see Chapter 3) and Reuben and Sorensen (2012)

The present study was initiated to focus on nitrate-N contributions from irrigated agriculture, lawns and onsite wastewater discharges because the ground water proportion of nitrate-N contributions to the annual reservoir loadings was much higher than that of P (see Chapter 3). This study needed to be conducted in the Huntsville area because 1) the range for the hydraulic conductivities for the monitoring wells in this area (wells 4 and 8) bracketed the range for the entire water table aquifer; and 2) single factor analysis of variance showed that the means of the nitrate-N concentrations from wells 4 and 8 were statistically different ($p < 0.05$) in 2011. The study of nitrogen leaching in the Huntsville area was also essential because a bioassay study that was conducted in 2008 had shown that both nitrogen and phosphorus limited primary production in Pineview Reservoir (Table 14, Appendix C). It was necessary to study nitrogen leaching in order to gain more insight on its leaching to ground water and explore best management practices that would help reduce the leaching losses around the reservoir since most eutrophication control measures target phosphorus (Rast and Thornton, 1996) as opposed to nitrogen.

The study consisted principally of performing and analyzing GIS-based Nitrogen Loss and Environmental Assessment Package (NLEAP-GIS 4.2) simulations of nitrate-N leaching from irrigated cropland, lawns and onsite wastewater drain-fields overlying the water table aquifer in sector 4-8 (Fig. 26). It was conducted for two years: from 1 January 2010 through 31 December 2011. An objective of the study was to identify site-specific best management practices that would help reduce ground water nitrate-N loadings to Pineview Reservoir.

Approach and methods

Ground water in Ogden Valley exists in perched, confined (artesian), and unconfined (water table) aquifer formations (Avery, 1994). Pineview Reservoir was finished in the silt-clay layer that separates the confined aquifer and the shallow unconfined aquifer in the center of the southern part of the valley (Avery, 1994). Ground water inflows to the reservoir are mainly from the shallow, unconfined aquifer (Avery, 1994) which is primarily recharged by precipitation, irrigation, seepage from streams, and ground water flow from an adjacent unconfined aquifer beyond the artesian aquifer boundary (Snyder and Lowe, 1998; Tetra Tech, 2002). The average annual precipitation in Ogden Valley is 558 mm (22.0 in.) (WRCC, 2012). Precipitation in the valley is snow dominated and the high mountain areas receive the highest snowfall (Tetra Tech, 2002). Another source of recharge for the shallow unconfined aquifer in Ogden Valley is wastewater disposal from onsite wastewater treatment systems (OWWTS).

Nitrate-N leaching simulations for irrigated croplands, lawns, and OWWTS drain-fields in sector 4-8 were conducted using NLEAP-GIS 4.2 following procedures outlined by Delgado et al. (2010). The crops for which the simulations were conducted were alfalfa, spring wheat, grass pasture, grass hay and turf grass. A GIS shapefile containing information on the geographic locations of the crop fields and their respective crop types was obtained from the NRCS field office in Ogden, Utah, in 2009. Impervious surfaces were removed from the shapefile using the *editor toolbar* environment in Arc GIS. Missing crop field and lawn parcels were manually added to the shapefile using 2012 Google imagery. Nitrate + nitrite N simulations were run for seven years (2005 through

2011) for the croplands and lawns while simulations for the drain-fields were run for five years (2007 through 2011). The years for which simulation results were compared were 2010 and 2011. The other years were incorporated in the simulations to ensure that nearly steady state conditions existed. Figure 27 is a concept diagram showing the parameters for which the simulations were done.

The generalized concept diagram was developed based on the following simplifying assumptions: 1) no direct discharge of wastewater to the reservoir or its tributaries since no point source discharge permits existed in the valley; 2) no surface application of onsite wastewater in sector 4-8; 3) no direct discharge of septic system

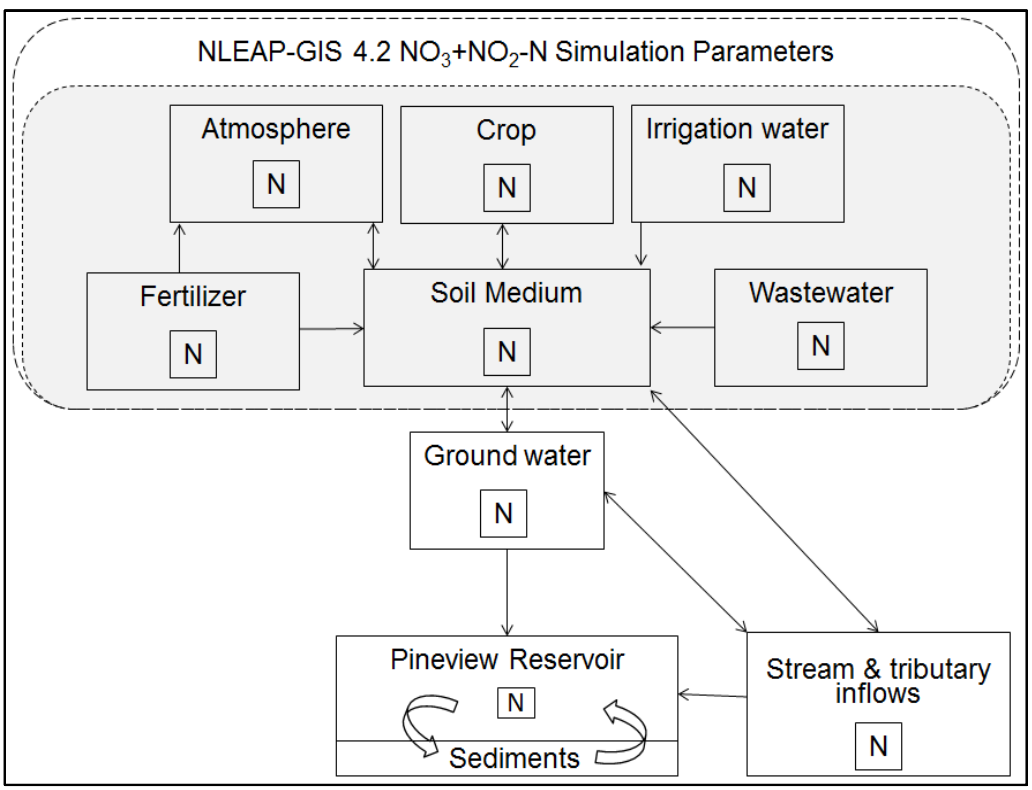

Figure 27. A generalized concept diagram for nitrate-N pools and transport simulations conducted for sector 4-8 using the NLEAP-GIS 4.2 program (Delgado et al., 2010). Simulated parameters are presented on a gray background

effluent into ground water, i.e. the drain-fields were designed and installed following guidelines and regulations stipulated by the EPA (USEPA, 1995, 2002); and 4) nutrient movement from the reservoir to the shallow unconfined aquifer was negligible because daily water table elevations in the wells were predominantly greater than corresponding reservoir elevations. However, the reservoir elevations were above water table elevations in well 8 on 20 June through 22 June 2011 by 4 ± 0.5 cm. The effects of the higher reservoir elevations were considered negligible since the differences in elevations were relatively small, may have been less than the error in measuring the reservoir and or well water elevations, and were short-lived (Reuben and Sorensen, 2012).

NLEAP-GIS 4.2 (Delgado et al., 2010) was used because 1) it is easy to access and process online soil and climate data; 2) it has the capability to facilitate examination of nitrate-N pools and transport at field, watershed and regional scales; and 3) viewing of the simulation results in GIS layers simplifies planning and/or implementation of site-specific best management practices. The program has a Microsoft Excel user interface for imputing soil layer data, climatic data, and management scenarios for which simulations of nitrogen pools and pathways in the environment are conducted (Delgado et al., 2010; Shaffer et al., 2010). Simulation outputs are displayed in Excel and can be exported to GIS through a GIS database file link (Delgado et al., 2010).

Input data on fertilizer application rates to crops were obtained from the Utah Fertilizer Guide (USU Cooperative Extension, 2010) and through personal communication (James Barnhill, Utah State University Agricultural Extension Agent, 20 June 2012). Crop water requirement and irrigation interval data were obtained from a crop consumptive use report prepared by the UAES (1994). Data on fertilizer and irrigation application rates for lawns were obtained from Sagers (1990) and the Utah Department of Water Resources website (UDWR, 2012), respectively. The actual amount of irrigation water applied to the croplands was estimated by dividing the crop water requirements by 0.6, the average application efficiency for conventional furrows and basins (with or without furrows) computed from efficiencies reported by Eisenhauer et al. (2011).

Irrigation water was assumed to have a nitrate-N concentration of 0.25 mg N L^{-1}, the arithmetic mean for the South Fork of the Ogden River (31 observations from the North Branch and 32 from the South Branch of the South Fork) from 1 January 2010 through 22 August 2011. It was befitting to use data from the South Fork of Ogden River because most of the irrigation water in sector 4-8 comes from this stream. The concentration was similar to that reported by Reuben et al. (see Chapter 3) for all the five major streams in Ogden Valley from 1 May 2010 through 30 April 2011. Soil data for the Ogden Valley were downloaded from the NRCS Soil Survey Geographic (SSURGO) online database (NRCS, 2012a) using NLEAP-GIS 4.2 soil download tool (Delgado et al., 2010). Properties of the soils present in sector 4-8 in the valley are presented in Table 8.

Table 8. Properties of soils present in sector 4-8 (Huntsville Town area) in Ogden Valley, Utah (NRCS, 2012a)

Soil Series	Symbol	Bulk density (g cm^{-3})	Percent OM content	Hydrology type	Drainage properties
Canburn silt loam	Cb	1.39	1.92	D	Poorly drained
Eastcan loam	EaA	1.31	1.87	C	Moderately well drained
Parleys loam	PaA	1.31	1.32	B	Well drained
Phoebe fine sandy loam	PhA	1.42	1.47	B	Well drained
Sunset loam	SwA	1.50	1.29	B	Somewhat well drained
Utaba cobbly loam, warm	UbA	2.11	0.66	A	Well drained

Note: Initial residual nitrate-N = 22.5 kg N ha^{-1} (20 lbs. ac^{-1}).

Turf grass was not among the list of crops included in NLEAP-GIS 4.2 crop input file. Use of corn silage (whose properties were present in the NLEAP program's crop input file) as a surrogate crop to turf grass was recommended (Jorge Delgado, USDA-ARS-SPNR, personal communication by email, 28 June 2012) since they both belong to the grass family. A turf grass crop with similar properties to those of corn silage was added to the crop input file to serve the purpose. The expected yield was changed from 22 (for corn silage) to 3 kg per hectare (for grass).

To implement the model, we assumed that fertilizer applications to the lawns were at a rate of 49 kg N ha^{-1} (1 lb. N per 1000 ft^2) at an application interval of 5 weeks (Sagers, 1990) during the months of April through September. Planting and harvesting were assumed to occur every year because the capability for yearly simulations for perennial crops was not available. Weekly harvests of turf were incorporated in the simulation to represent weekly mowing. Cuttings were not removed from the lawns to mimic the practice followed by most lawn owners. The last harvest event in each year was assumed to occur in November to ensure that the cold months of the year did not have any turf grass inputs. This was based on the assumption that the effect of the turf on nitrate-N pools and transport during this time of the year would be almost negligible due to freezing conditions which render the turf dormant. Annual replanting of the turf was scheduled in April to mimic springing up of a hypothetical perennial turf stand.

Input parameters for water application to the lawns included sprinkler irrigation at recommended application rates of 13 mm (0.5 in.) per irrigation event and varying irrigation intervals of six days in April and September, four in May, three in June through August, and ten in October (UDWR, 2012). The recommended irrigation application rate was lower than the rate (25.4 mm) at which average Utah homeowners watered their lawns (UDWR, 2012). It was befitting to use the recommended irrigation application rates because fertilizer applications to the lawns were also based on the recommended rates.

NLEAP-GIS 4.2 did not have a provision for simulation of septic system effluent applications to drain-fields but had the capability to simulate application of wastewater treatment plant sludge by injection or incorporation. A suggestion to simulate septic system effluent as wet sludge application by injection was considered reasonable if the application is uniform (Jorge Delgado, USDA-ARS-SPNR, personal communication by email, 28 June 2012). The input data requirements for a sewage sludge simulation in

NLEAP-GIS 4.2 were the mass loading rate of wet sludge, sludge water content (percent), C:N ratio, and the percent (dry basis) organic matter (OM), nitrate-N, and NH_4-N content (Delgado et al., 2010). The septic effluent application for each drain-field was assumed to be uniformly distributed, and at the design rate of 568 L d^{-1} (150 gpd) per bedroom served (USEPA, 2002, 1995). The properties of the simulated septic effluent were assumed to be 0.01 percent nitrate-N, 0.09 percent NH_4-N in the suspended solids (USEPA, 1995) and a C:N ratio of 10, and 0.05 percent OM in the total mass of effluent applied.

The C:N ratio for septic effluent was based on the report that it rarely exceeds 10 (Judith Sims, Utah State University, personal communication be email, 31 October 2012) and on the C:N ratio for microbes (Brady, 1977). The percent OM was estimated from a septic effluent 5-day biological oxygen demand (BOD_5) of 240 mg L^{-1} for La Pine, Oregon, septic systems reported by WSDH (2004). The septic effluent BOD_5 was converted to chemical oxygen demand (Lawton and Codd, 1991) using a BOD_5/COD ratio of 0.37 (Eckenfelder and Musterman, 1995) and the resulting COD was converted to effluent carbon concentration using the assumption that all of the COD is due to the oxidation of carbon to CO_2. The resulting carbon concentration (243 mg L^{-1}) was converted to OM concentration by multiplying the carbon concentration by a factor of 1.97 from the soils literature (Howard, 1965).

The effluent mass loading rate was estimated from the number of bedrooms served by each drain-field, the design application rate, drain-field area, the density of pure water (1 g cm^{-3}) at 25°C, and the total mass of solids. The total organic solids mass that would contribute nitrogenous compounds was assumed to be the concentrations of dissolved (OM) and suspended solid constituents (TSS). The TSS concentration was assumed as 48 mg L^{-1} based on reported median septic system effluent concentration for La Pine, Oregon (WSDH, 2004). The TSS concentration was close to the minimum USEPA representative TSS concentration of 50 mg L^{-1} for septic system effluent (USEPA, 2002; WSDH, 2004). Septic system effluent water content was estimated by deducting the estimated total solids percent of 0.05 percent (527 mg L^{-1} total solids computed using the preceding steps) from 100 percent.

Data (in Microsoft Excel and GIS shapefile formats) on the number of bedrooms per housing unit, and sizes and geographic locations of the drain-fields for permitted OWWTS were obtained from the Weber-Morgan Health Department in 2012. It was assumed that all residential housing units in sector 4-8 had OWWTS despite not being included on the list of permitted housing units supplied by the Morgan-Weber Health Department. Approximately 60 percent of the residential units in sector 4-8 did not have number of bedroom and drain field area data. These were assigned the median number of bedrooms per housing unit and the median drain-field area that were computed from the available data's nearly geometric distribution. An intercept of the drain-field GIS spatial layer with the soil type layer (NRCS, 2012a) for sector 4-8 resulted in 322 drain-field locations. The septic effluent mass loading rates applied in the NLEAP-GIS 4.2 drain-field simulations had a median of 217 and a mean of 220 (standard deviation = 150) metric tons (MT) ha^{-1} yr^{-1}. The effluent loading rates were grouped into fourteen discrete

10 MT ha^{-1} d^{-1} bins. The bins were represented by the bin averages that ranged from 80 – 612 MT ha^{-1} d^{-1}.

It is common practice in Ogden Valley for drain-fields to be overlain by grass lawns which are sprinkler irrigated hence other events imputed into NLEAP-GIS 4.2 for simulation included planting, irrigation, and harvesting of the grass. Management practices for the drain-field grass were the same as those discussed under lawn turf except that the drain-field grass did not receive any commercial fertilizers.

Climatic data for the Huntsville Monastery Weather Station (Fig. 27) were downloaded from the Water Resources Center website (WRCC, 2012) for the period from 1 January 2005 through 31 December 2011. Missing climatic data (approx. 10 percent) were imputed by linear interpolation. The imputation may have increased the uncertainty associated with the climatic data but the error was tolerable since 90 percent of the data were available. The censored climatic data were formatted and saved as a text file which was uploaded into the NLEAP-GIS 4.2 user interface through the program's *convert climate* tool (Delgado et al., 2010).

Tillage operations for the croplands, lawns and drain-fields were only applied in the first year of the rotation to mimic reported farming practices for Ogden Valley (James Barnhill, Utah State University Agricultural Extension Agent, 20 June 2012). Planting of all the crops was assumed to occur on 20 April each year based on UAES (1994). Alfalfa fields were subjected to a six year crop rotation under which the first year was planted to spring wheat and the next six years alfalfa (James Barnhill, Utah State University Agricultural Extension Agent, 20 June 2012). No rotations were employed for the grass pasture, grass hay or turf grass. A monoculture of spring wheat was simulated as a baseline scenario. Under this management practice, the events carried out for the spring wheat planted in the first year of the alfalfa/wheat rotation were simulated for seven years. Incorporation of a baseline crop in the simulations was done to mimic the approach followed by Shumway et al. (2012) who used a corn monoculture. In contrast to the approach followed by Shumway et al. (2012), tillage operations in the current study were only included in the initial year in order to mimic the tillage practice followed in Ogden Valley. Simulations were done for a total of seven years to ensure that nearly steady state conditions existed during the target years (2010 and 2011). More details about the cropping events are presented in Table 9.

Soil sample cores were collected to respective depths when drilling wells 8 and 9 on 11 April 2011. The soil cores were collected to total depths of 12.7 and 7.8 m, respectively. They were analyzed for physical and chemical properties including nitrate-N, pH and texture (Table 21, Appendix K). The lab analyses were conducted at the Utah State University Analytical Laboratory following analytical methods outlined by Sparks et al. (1996). The 2N KCl extractable nitrate-N results were compared with the residual nitrate resulting from the NLEAP-GIS 4.2 simulations for April and December, 2011.

Daily ground water flow data for sector 4-8 were extracted from the flow spatial distribution raster shapefile developed by Reuben and Sorensen (2012) for the shallow unconfined aquifer in Ogden Valley. Extraction of the data was done using python script

Table 9. Management scenario events for nitrate-N pool and pathway simulations for sector 4-8 croplands, lawns, and drain-fields[a]

Management scenario	Length of growing season (days)	Irrigation depth (mm yr^{-1})	Fertilizer N applied (kg N ha^{-1} yr^{-1}) [b]
Alfalfa/wheat rotation	163 (125) [c]	1231 (835)[b]	0 (112) [c]
Turf grass	179	610	245
Wheat monoculture	125	835	112
Drain field	179	610	0
Grass pasture	174	957	56
Grass hay	174	1077	26

[a] :All data obtained or modified from the literature (Beddes and Kratsch, 2008; Sagers, 1990; UAES, 1994; UDWR, 2012; USU Cooperative Extension, 2010); [b] Urea ($(NH_2)_2CO$) fertilizer; [c] Numbers in parenthesis are for a spring wheat crop grown in the first year of the rotation.

batch command in Arc GIS by applying the clip feature tool and converting the resulting raster layers to point features which were then converted to database files. The geometric mean, median and standard deviation of the daily ground water flows were computed from the resulting database files using the procedure reported by Reuben and Sorensen (2012).

Ground water nitrate-N concentrations were estimated by dividing the sum of the NLEAP-GIS 4.2 simulated nitrate-N leaching losses (from the croplands, lawns and drain-fields) for 1 May through 14 November 2011 by the corresponding ground water flows obtained from the GIS database file developed by Reuben and Sorensen (2012). The geometric mean (0.4 mg N L^{-1}) of the background concentration data collected for the study reported by Wallace and Lowe (1998) was added to both the upper and lower limits of the 95 percent confidence interval for the resulting concentrations. The additions were based on the assumption that sector 4-8 received a steady inflow of ground water with a mean concentration of 0.4 mg N L^{-1} from the up-gradient, deep, unconfined aquifer. A comparison was made between the 95 percent confidence interval for the mean predicted ground water nitrate-N concentration and the confidence interval for the measured concentrations from wells 4 and 8 (Table 18, Appendix G) during the same period. Six sets of grab sampling measurements were used to compute the 95 percent confidence interval for the mean measured concentration.

Lawn simulation for a 50 percent lower N application rate than the recommended 49 kg N ha^{-1} per application was conducted to test its effect on N losses and residual. A similar comparison was made between simulation results for an irrigation application rate of 12.7 mm per irrigation event and 25.4 mm per event.

Results and discussion

Summary statistics for the mean annual nitrate-N leaching losses for 2010 and 2011 are presented in Table 10. Lawns and drain-fields were predicted to have high nitrate-N leaching rates (184 ± 21 and 76 ± 16 kg N ha^{-1} yr^{-1} (mean ± 1SD), respectively) relative to agricultural land. The high leaching rates from lawns may be attributed to high fertilizer application rates that the lawns were subjected to in order to maintain high quality (Sagers, 1990). High leaching rates from the drain-fields may be attributed to high sewage effluent loading rates in well drained soils. The influence of soil drainage type on nitrate-N leaching rates can be observed from Figure 28. Only the median sewage

Table 10. Summary statistics for annual nitrate-N leaching from fields, lawns and drain-fields for the period from 1 January 2010 through 31 December 2011

Management scenario	Nitrate-N leaching (kg N ha^{-1} yr^{-1})[a]				
	Minimum	Maximum	Median	Mean	SD
Grass hay	70	96	80	82	9
Grass pasture	76	106	95	93	12
Alfalfa/wheat rotation[b]	4	10	4	5	2
Lawn turf	122	194	191	184	21
Drain-fields[c]	44	123	71	76	16
Wheat monoculture	11	33	12	15	6

[a]: Mean annual leaching rates for 2010 and 2011; [b]: Spring wheat in 2005 and alfalfa from 2006 - 2011; [c]: Drain-field statistics are for fourteen management scenarios (effluent loading rates)

management scenario (i.e., 217 MT ha^{-1} d^{-1} application rate) has been presented in Figure 28 because it was the only one whose respective drain-fields existed in all the six soil types present in sector 4-8. The annual leaching rate of 76 ± 16 kg N ha^{-1} yr^{-1} for all the drain-fields was similar to the leaching rate of 75 ± 15 kg N ha^{-1} yr^{-1} from the fields under the median sewage effluent application rate.

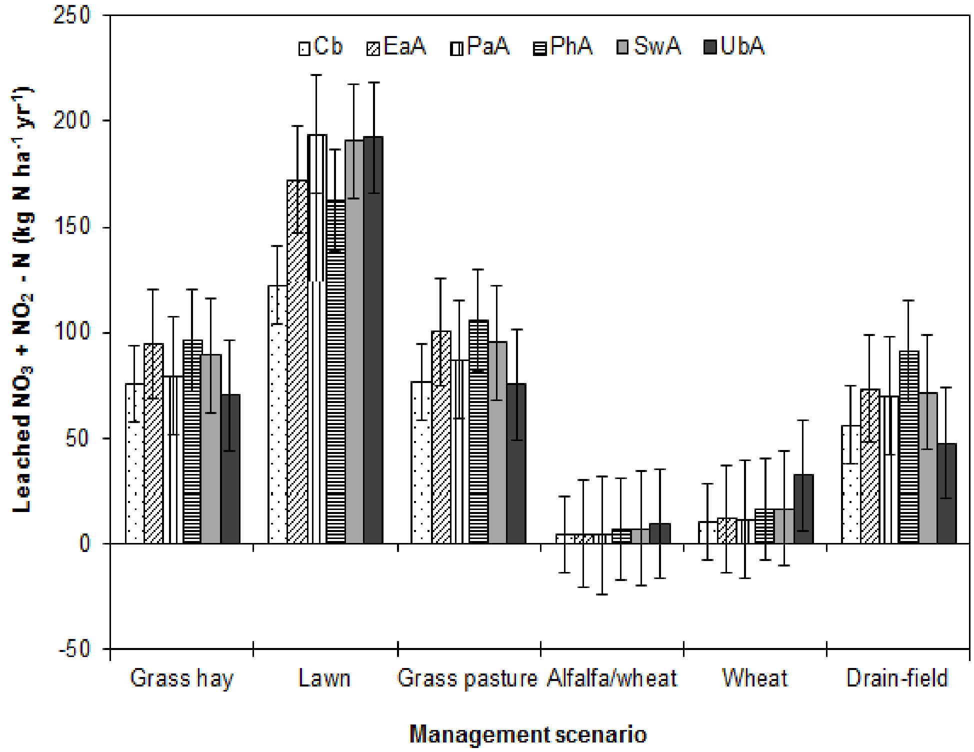

Figure 28. Mean annual nitrate-N leaching rates for different soil types under different management scenarios. The error bars are ± one standard error. The codes for different soil types are given in Table 8

The spatial distributions of the mean annual leaching rates from various management scenarios during the period from 1 January 2010 through 31 December 2011 are presented in Figures 29 through 31. The figure presents fields that closely represented the management practices followed in Ogden Valley's sector 4-8 and a hypothetical baseline scenario under which all the fields were assumed to be under a wheat monoculture for a period of seven years. As depicted in Figures 29 through 31, the mean annual leaching rates from lawns and drain-fields were generally higher than crop fields. The crop field map (Fig. 31) shows that alfalfa fields had the least nitrate-N leaching losses compared to the other crops. The low leaching losses from the alfalfa fields

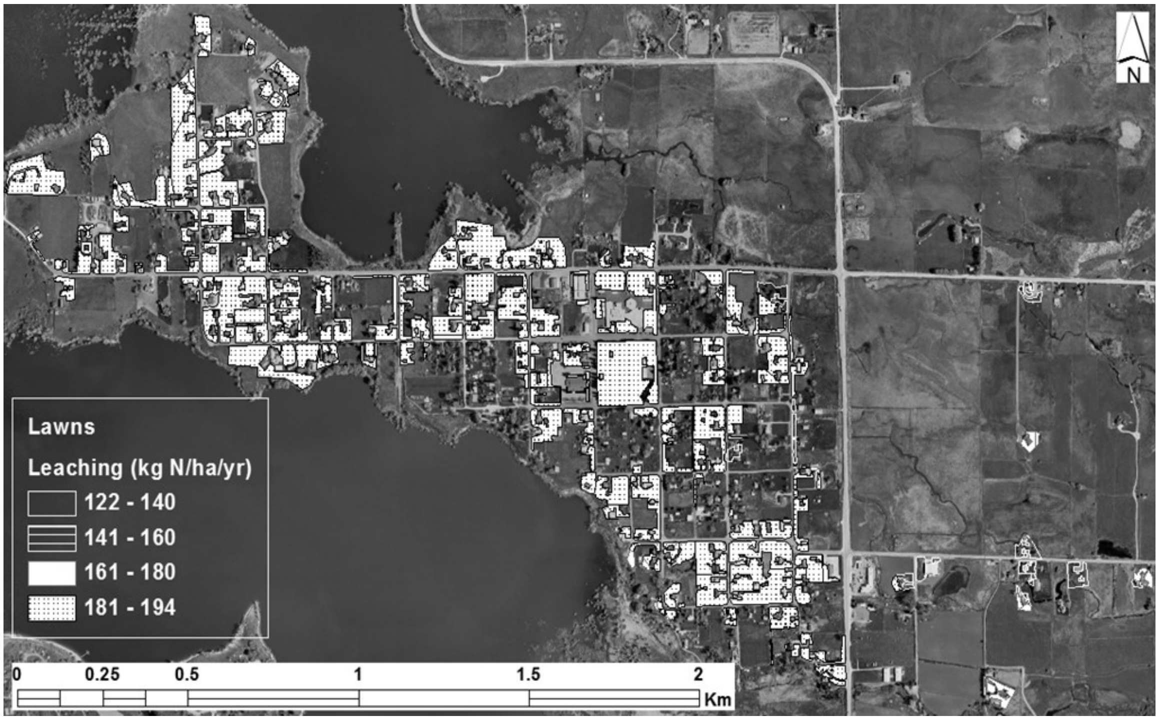

Figure 29. Simulated nitrate-N leaching rates from lawns in sector 4-8

Figure 30. Simulated nitrate-N leaching rates from drain-fields in sector 4-8

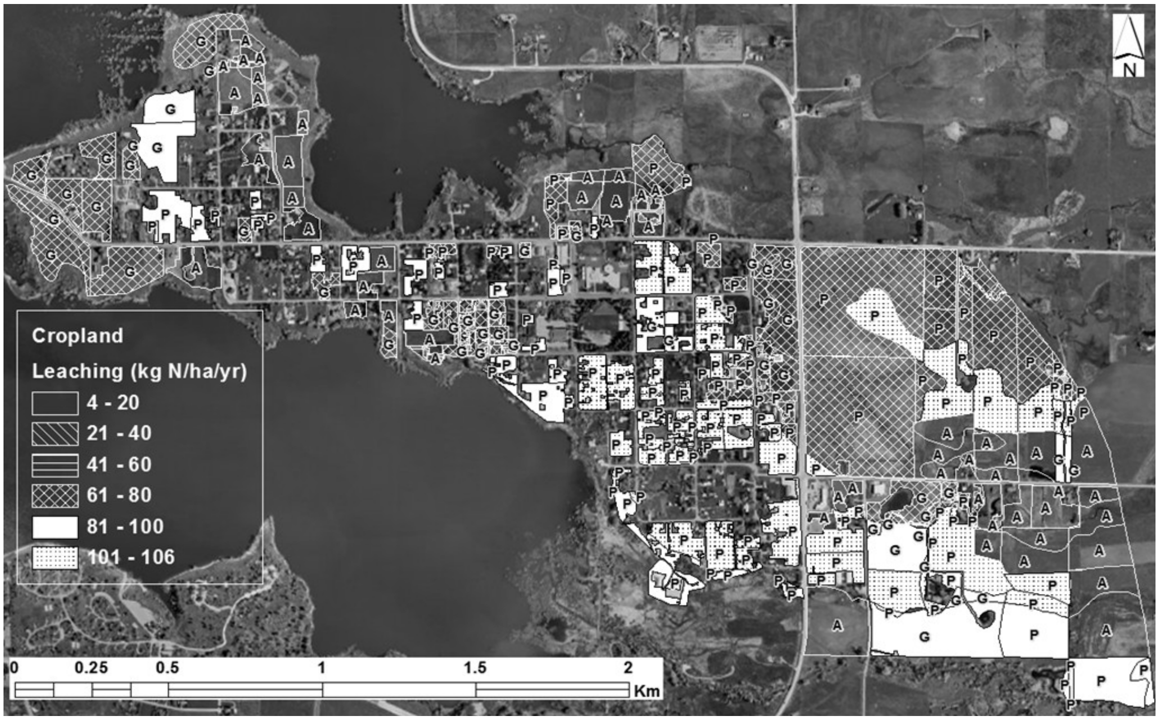

Figure 31. Simulated nitrate-N leaching rates from croplands in sector 4-8. A, G and P represent alfalfa/wheat rotation (year 1 wheat; next 6 years alfalfa), grass hay, and grass pasture, respectively. The fields are demarcated according to soil and crop types

resulted because the alfalfa fields received no nitrogen fertilizer applications during the last six years of the alfalfa/wheat rotation since alfalfa is a nitrogen fixer.

The baseline scenario (wheat monoculture) reported lower nitrate-N leaching rates (15 ± 6 kg N ha^{-1} yr^{-1}) than other crop management practices except the alfalfa/wheat rotation. The relatively lower leaching rates from the baseline scenario compared to grass and pasture fields reflected the high nitrate-N uptake capabilities of a wheat crop. This would signify the role of wheat (and any similar crop such as corn) in lowering the soil residual nitrate pool thereby reducing nitrate-N leaching potential. The baseline scenario map also portrayed differences in leaching rates that could be attributed to the different drainage properties of the soils and organic matter content since well drained soils associated with low organic matter content (i.e. UbA, SwA and PaA in Table 8) had higher leaching rates than the other soils.

The overall mean annual contributions of the cropland, lawns and drain-fields towards the soil nitrate-N pool and various losses simulated for sector 4-8 in Ogden Valley using NLEAP-GIS 4.2 program are presented in Table 11. The overall contribution of drain-fields (200 kg N) to nitrate-N loading of ground water in sector 4-8 in Ogden Valley was 50-fold and 70-fold lower than those of the lawns (10,030 kg N) and cropland (14,150 kg N), respectively, despite drain-fields having a similar leaching-rate confidence interval (77 ± 16 kg N ha^{-1} yr^{-1}) to croplands (71 ± 37 kg N ha^{-1} yr^{-1}). This is because the total drain-field area (2.6 ha) was 20-fold and 80-fold lower than the total land area under lawns and croplands, respectively. The relatively high leaching losses from lawns were due to relatively high leaching rates of 184 ± 21 kg N ha^{-1} yr^{-1} that may be attributed to relatively high fertilizer application rates (Table 9).

Lawns also had a relatively high annual nitrate-N residual of 13, 690 kg (Table 11) due to relatively high rates of soil nitrate-N accumulation (Figure 32). Again, this may be due to relatively high fertilizer application rates. The relatively high residual nitrate-N on the lawns imply that future N leaching in sector 4-8 would be more sustained by the lawns than the other fields if all lawns, croplands, and drain-fields received no additional nitrogen in succeeding years. A positive correlation between residual N and leaching losses has been reported in the literature (Shumway et al., 2012).

Table 11. Annual nitrate-N pools and losses simulated using NLEAP-GIS 4.2 for sector 4-8

Nitrate-N residual or loss	Cropland (kg N yr^{-1})	Lawns (kg N yr^{-1})	Drain-fields (kg N yr^{-1})	Baseline scenario (kg N yr^{-1})
Leaching	14,150	10,030	200	3,800
Denitrification	2,050	1,720	30	1,850
Emissions	460	680	5	760
Runoff	40	10	0	40
Volatilization	980	3,960	5	3,270
Residual	9,170	13,690	180	3,570

Note: Total land area (ha) for which the results were based: cropland (alfalfa/wheat rotation, grass pasture and pasture hay) = 209; lawns = 53; drain-fields = 2.6; and baseline scenario = 262.

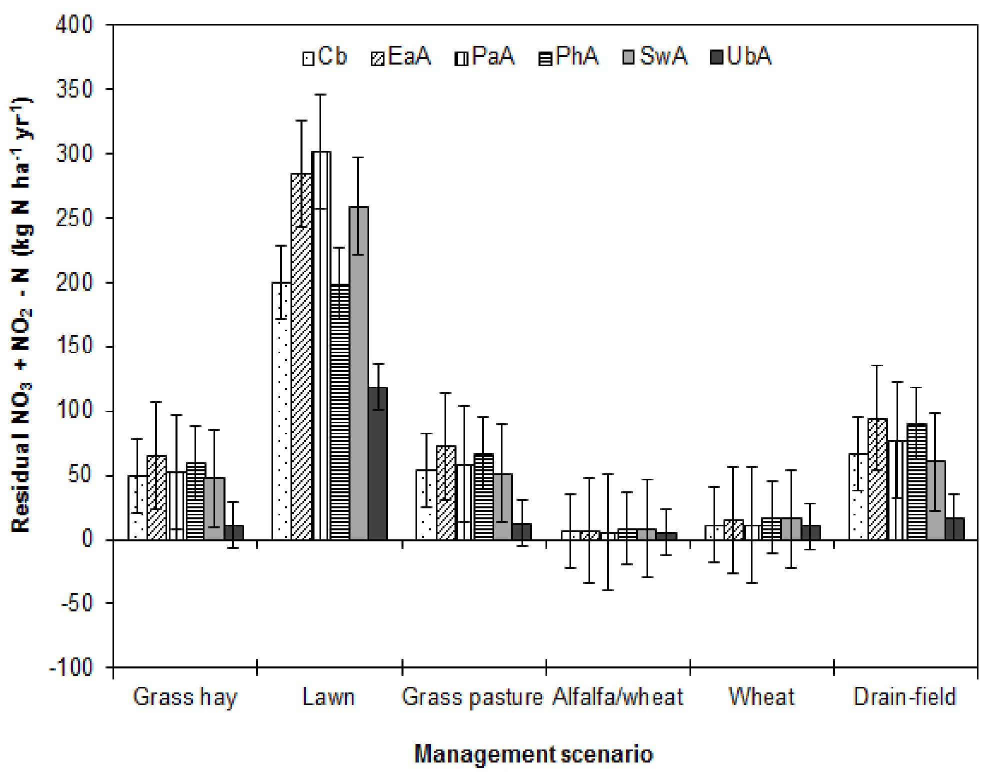

Figure 32. Residual nitrate-N for different soils under different management practices. The error bars are ± one standard error

The overall NLEAP simulated residual nitrate-N in the top 1.5 m soil profile for April 2011 from all the management practices (croplands with rotation, lawns and drain-fields), excluding spring wheat monoculture, was 2.7 mg N kg^{-1} soil. The annual average residual nitrate-N estimated from the NLEAP simulations was 4.0 mg N kg^{-1} soil. The predicted residual nitrate-N values were higher than the extractable nitrate-N of <1.25 mg N kg^{-1} (Table 21, Appendix K) measured from soil cores obtained during the construction of well 8 on 11 April 2011. This difference may also reflect the effect of spatial variability of soil nitrate-N content in sector 4-8 since only well 8 soil cores were analyzed in this sector and the overlying land use is a road side with weeds and grasses that are not fertilized.

Soil residual nitrate-N spatial variation was evidenced from soil core analysis for well 9 (located outside sector 4-8, within 1.4 km from well 8) whose extractable nitrate-N content ranged from 1.9 to 17 mg N kg^{-1} at depths 2.7 through 7.8 m beneath the soil surface (Table 21, Appendix K). Well 9 was located along a major highway but within 35 m from a septic system, 75 m from a barn and a corral, 100 m from an animal feeding location, and 100 m from a seldom (if any) used septic system. The hypothesis that spatial variability may have caused the difference between the simulated and measured extractable nitrate-N is further supported by the nitrate-N concentration data reported by Reuben and Sorensen (2012) for 1 May through 14 November 2011. Six water samples from well 4 had a median nitrate-N concentration that was more than 1.5 fold higher than a similar number of samples from well 8 (Reuben and Sorensen, 2012). Results from the current study imply that the NLEAP estimate of soil residual nitrate-N was reasonable and adequately reflects the field scenario. This is on the basis that the 95 percent confidence interval for all soil residual nitrate-N measurements from wells 8 and 9 (0 – 8.7 mg N L^{-1}) contains the estimated soil residual of 4.0 mg N L^{-1}). The confidence interval was constructed on the assumption that all soil residual nitrate-N values below detection limit were equal to the detection limit value of 1.255 mg N L^{-1}.

The predicted overall residual nitrate-N for the lawns (12±3.4, mean±1SD) was higher than the drain fields (3.4±1.1) and crop lands (2.1±1.2 mg N kg^{-1} soil). These results imply that the amount of nitrogen applied to the lawns exceed their requirement. There is a need to ground proof the results through soil tests in the croplands, lawns and drain-fields in order to reduce any uncertainties associated with the simulation results.

The 95 percent confidence interval (arithmetic mean ± 2 SD) of the nitrate-N concentrations, estimated for sector 4-8 by addition of the background nitrate-N concentration of 0.4 mg N L^{-1} to the concentrations estimated from NLEAP simulated annual leaching losses and ground water flows modeled by Reuben and Sorensen (2012), was 1.6 – 2.2 mg N L^{-1} (Table 12). Table 12 also shows that the measured nitrate-N concentrations from wells 4 and 8 had a 95 percent confidence interval of 1.2 – 9.0 mg N L^{-1}. The two confidence intervals overlap implying that there was no statistical difference between the measured and estimated concentrations. The similarity among the estimated and measured nitrate-N concentrations suggest that NLEAP-GIS 4.2 may be a proper tool for estimating ground water nitrate-N concentrations for the water table aquifer when the nitrate-N background concentration and ground water flows are known.

Table 12. Comparisons of observed ground water nitrate-N concentrations with annual estimates using the NLEAP-GIS 4.2 simulated leaching loss of 9,021 kg N and modeled ground water flows for 1 May through 14 November 2011

Description	Mean	Upper limit	Lower limit
Measured concentrations (mg N L^{-1}) [c]	5.1	9	1.2
Estimated concentrations (mg N L^{-1})[b]	1.9	2.2	1.6
Ground water flow (m^3)[a]	6,162,684	7,224,931	5,100,437

[a]: Ground water flows for 1 May through 14 November 2011, obtained from a nearly log-normal spatial distribution of sector 4-8 flows obtained from a GIS database file developed by Reuben and Sorensen (2012); [b]: The sum of the background nitrate-N concentration of 0.4 mg N L^{-1} and the confidence interval of estimated concentrations from [a] and the NLEAP simulated leaching loss of 9,021 kg N. The simulated leaching losses were from 802 fields (croplands, lawns, and drain-fields) for 1 May through 14 November 2011. The total area for the 802 fields was 265 ha; [c]: Statistics obtained from measured N concentrations from wells 4 and 8. Seven grab samplings for 19 April through 14 November 2011 (Table 18, Appendix G) were used.

The estimated denitrification rate from sewage effluent in sector 4-8 was 0.13 mg N L^{-1}. The estimate was based on the simulated average annual denitrification losses for an annual sewage effluent application of 211,000 m^3. The estimated denitrification rate can be compared with results from a tracer study conducted by Korom (1991) on denitrification of septic effluent in Heber Valley, Utah, located 90 km SSE of Ogden Valley, which showed that denitrification rates in that valley would reach 0.74 mg N L^{-1}. So, it is conceivable that denitrification rates of the septic system effluent in sector 4-8 may have been greater than that estimated by the NLEAP program.

The overall contribution of lawns to nitrate-N loading of ground water in the sector was 30 percent lower than the croplands despite that the mean leaching rate from the lawns (180 kg ha^{-1} yr^{-1}) was more than double as much as the croplands (70 kg ha^{-1} yr^{-1}). Again, the high leaching rates from the lawns are overshadowed by the smaller (4-fold lower) total land area occupied by the lawns relative to cropland. The results imply that both lawn fertilizer applications and sewage effluent applications pose a threat to ground water quality. The threat may be pronounced in areas where lawns or drain-fields occupy a larger fraction of the area. It appears that sector 4-8 in Ogden Valley may load more nitrate to the reservoir from lawn and cropland leaching than drain-fields since the latter occupy a smaller area than the former two.

Simulation of nitrate pool and losses from lawns at 50 percent lower fertilizer application rate (i.e. 125 kg N ha^{-1}) than the recommended, lowered the nitrate-N residual, leaching, N_2 emissions, N_2O emissions, and volatilization while changes in runoff losses were negligible (Table 13). The overall nitrate-N leaching (and estimate of ground water concentration) declined by 15 percent and the overall residual soil N by 25 percent. The results underscore the significance of reducing the fertilizer application rates to lawns in order to lower nitrate-N leaching (and other) losses. Table 13 shows that a 50 percent reduction in irrigation application rate relatively decreased leaching rates by 2-fold lower than the 50 percent fertilizer application reduction. The reduction in irrigation application rates resulted in 31 percent higher residual N. The residual N and leaching results suggest the need for integration of irrigation water management, fertilizer application management and other best management practices in controlling N leaching

Table 13. Comparison of impact of lowering lawn irrigation amount by 50 percent with lowering fertilizer amount by 50 percent in sector 4-8, Ogden Valley, Utah

Input	Percent change in pathway or residual[a]					
	Leaching	Residual	Denitri-fication N_2 Emission	Denitri-fication N_2O Emission	Runoff	NH_3 Volatil-ization
Irrigation	-18 ± 6	31 ± 44	21 ± 4	37 ± 53	-19 ± 2	16 ± 112
Fertilizer	-36 ± 8	-40 ± 11	-41 ± 17	-39 ± 7	-0.02 ± 2	-50 ± 1

[a]: A negative change implies reduction in magnitude for the pool or pathway.

from lawns in sector 4-8. A comparison of leaching rates from the current fertilizer rate and the recommended irrigation application rate with a scenario comprising the current fertilizer application rate and the rate at which average Utah homeowners water their lawns showed that the latter had 9 percent higher leaching rates than the former.

The study has shown the need to implement best management practices to reduce the impact of lawns and croplands on ground water quality. For example, reducing the amounts of fertilizers applied to the lawns and practicing precision fertilizer application based on crop need, residual soil nitrate-N, soil drainage properties and organic matter content (NRCS, 2012b; Shumway et al., 2012). Precision irrigation water applications to the lawns and cropland based on crop water need and soil drainage properties in addition to following the water applications recommended by the UDWR (2012) is likely to be effective in reducing nitrate loading. Some soils may need the addition of organic matter to improve their water holding capacities (Stamatiadis et al., 1999) as long as the application is done based on the organic matter needs of the respective soils. Another best management practice that has a potential to reduce nitrate-N leaching losses from the cropland and lawns is split fertilizer applications. A sample simulation conducted in this study on split fertilizer applications (data not included) to the cropland showed a potential decline in leaching losses. The cost-effectiveness of the proposed and other best management practices that could be employed in Ogden Valley and other places in the world needs to be explored.

The study has also shown that NLEAP-GIS 4.2 may be a proper tool for estimating soil nitrate-N residual and ground water nitrate-N concentrations. This was evidenced from the similarity in measured residual N and residual N estimated from the NLEAP simulated results. Measured nitrate-N concentrations from wells 4 and 8 had a confidence interval that intersected with the confidence interval for the concentrations estimated from the background nitrate-N concentration, NLEAP simulated N leaching and ground water flows modeled by Reuben and Sorensen (2012).

Conclusion

The study has demonstrated that NLEAP-GIS 4.2 can be used to estimate soil residual nitrate + nitrite N. This was demonstrated through the similarity between the estimated and measured soil residual nitrate-N whose 95 percent confidence intervals overlapped. NLEAP-GIS 4.2 can also be used to estimate nitrate + nitrite N concentrations of ground water in the water aquifer as demonstrated from the overlapping confidence intervals of estimated and measured nitrate-nitrite N concentrations. A need exists for ground proofing the results through soil tests in the croplands, lawns and drain-fields in order to reduce any uncertainties associated with the simulated results. Ground proofing was beyond the study's ability due to time and financial resource constraints.

The study has also demonstrated the need for nutrient pollution control programs to incorporate best management practices for lawns especially in areas where lawns occupy a large fraction of the land. The impact of leaching from lawns in areas with smaller fractions of land occupied by lawns may be small due to the ground water dilution effect. Some of the best management practices that could be employed on the lawns and cropland include precision fertilizer and water applications and enhancement of the soil organic matter content to levels that would enhance the water holding capacities of coarse textured soils. The NLEAP-GIS 4.2 program currently doesn't have a provision for turf and lawn mowing. The study incorporated weekly harvests in the nitrate + nitrite N simulations to mimic weekly mowing. It is uncertain how this approach may have affected the results but it is believed that the effect was minimal. It is recommended that future modifications of the NLEAP-GIS 4.2 program incorporate lawn mowing to facilitate nitrate + nitrite pathway and pool simulations for lawns.

Drain-field leaching rates were higher than those from cropland but their combined nitrate + nitrite N leaching amount was superseded by the much larger area of cropland. The findings were not unexpected and they do not negate but emphasize the importance of proper management of wastewater discharge to ground water because the high leaching rates are a clear indicator of the threat that sewage effluent poses to ground water especially in areas with high septic system densities. The results will help Huntsville Town Council decide whether or not need exists to either construct a central sewer system or a waste water treatment system that would remove most of the nutrients. Such a decision would be vital as the town population grows because the increase in population would imply increase in septic system density which would consequently increase nutrient loading to ground water.

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CHAPTER 6

SUMMARY, CONCLUSIONS, ENGINEERING SIGNIFICANCE AND RECOMMENDATIONS

Abstract

Nutrient loading and unloading for Pineview Reservoir were studied and the role of ground water in the estimates characterized. The study evaluated nitrogen and phosphorus transport towards Pineview Reservoir, through streams and ground water in the most immediate water table aquifer that surrounds the reservoir. The reservoir's chlorophyll a concentrations were also studied. Ground water contributed 23 percent of the total reservoir nitrogen loadings despite its flow contributions being 2 percent of the total inflows. Increase in chlorophyll a and total dissolved phosphorus concentrations preceded low dissolved oxygen concentrations implying that internal nutrient loading provided additional phosphorus needed for phytoplankton growth. Ground water spatial variability and nutrient transport to the reservoir were also characterized. High frequency monitoring of ground water elevations, grab sampling for nutrients such as nitrogen and phosphorus, and modeling the flows and nutrient transport in GIS, were conducted. Large spatial variability in the flows, nutrient concentrations, and nutrient transport were observed and attributed to differences in hydraulic properties of the aquifer, snowmelt flushing events, irrigation, and other non-point sources. The study also conducted NLEAP-GIS 4.2 simulations for nitrogen leaching from lawns, cropland, and septic drain

fields. Despite observing higher leaching rates from lawns and drain fields, the cropland had the largest total nitrogen leaching losses than the former two due to large crop area.

Summary and conclusions

This dissertation expanded the information available about sources and transport of nitrogen and phosphorus through the shallow, unconfined (water table) aquifer towards surface water bodies such as reservoirs and lakes. Chapters 3, 4, and 5 provide information that can be used by water resource managers, environmental specialists, water users, community planners and other stakeholders in making informed decisions on utilization and conservation of ground water in Ogden Valley, Utah, and elsewhere, in order to control pollution and avoid over exploitation of ground and surface water.

Pineview Reservoir nutrient loading, unloading and the role of ground water in the estimates

The objective of this chapter was to evaluate the transport of nitrogen and phosphorus towards Pineview Reservoir in Ogden Valley, Utah, through streams and ground water in the water table aquifer. Ground and surface water flow rates and nutrient contents were monitored to determine their proportionate contributions towards the reservoir inflows. It was established that the aquifer's ground water contributions towards the reservoir total inflows were 2 percent flow, 2 percent total dissolved phosphorus, 13 percent soluble reactive phosphorus and 22 percent nitrogen. The ground water loadings of dissolved nitrogen and soluble reactive phosphorus were 35 percent to 40 percent lower than those estimated in the TMDL study in 2002. The study also examined nutrient and chlorophyll *a* concentrations in the reservoir. Internal nutrient loading provided the additional bioavailable phosphorus needed to initiate phytoplankton blooms.

The limitation of this study was that no monitoring wells existed on the southern and western banks of the reservoir due to resource constraints. The median flows and concentrations represented by the five wells studied were assumed to represent these shoreline areas. The ground water table aquifer properties are known to be very heterogeneous and this assumption undoubtedly introduced error into the estimates.

Ground water flow spatial variability and nutrient transport to a reservoir in an irrigated mountain valley

This chapter reported GIS-based simulations that were conducted to examine and characterize the spatial variability of ground water flow parameters and nutrient transport in the Ogden Valley, Utah, a mountain valley dominated by irrigated agriculture. Nine monitoring wells (five studied in Chapter 3, and four additional ones) were studied under this objective. The spatial distribution of ground water flows and nutrient loadings were simulated for 15 November 2010 through 14 November 2011 from daily ground water and reservoir elevations, and grab sampled ground water nutrient concentrations. The grab sample nutrient concentrations were linearly interpolated to estimate daily concentrations that were used in the simulations. A multiple regression equation developed from reservoir and well data elevation data was used to interpolate missing ground water flows and nutrient loadings for the additional four wells for 15 November 2010 through 31 April 2011.

Large spatial variations in ground water flows and nutrient loadings towards the reservoir were observed. The spatial variations in flows were attributed to flow parameter (hydraulic head and conductivity) variations while snowmelt and irrigation flushing events accounted for variability in nutrient loadings. Much spatial variation was observed with the largest confidence interval of on 6 May and the smallest (447-1,814 $m^3\ d^{-1}$) on 27 June 2011. The wide 67 percent confidence interval for the flows (1,518 – 5,077 $m^3\ d^{-1}$) was observed in spring and attributed to rapid increases in reservoir elevations due to spring runoff emanating from snowmelt in the mountains and spatially variable changes in water table elevation. These fluctuations are believed to have caused large variations in the hydraulic gradients due to differences in cell locations relative to the reservoir shoreline. The variations in nutrient loadings appeared to be influenced by agriculture and domestic non-point nutrient sources.

The influences of the large hydraulic conductivity range (of 0.86 to 22 m d^{-1}) and possible existence of preferential flow paths on the spatial distribution of the nutrient loadings were major sources of uncertainty in the loading estimates. The uncertainty is believed to have been minimized by aggregation of the low and high flows in the computation of the loadings.

Estimated ground water nitrate loadings from lawns, irrigated cropland and onsite wastewater

The objective of this work was to determine the nutrient leaching contributions from lawns, irrigated cropland and onsite wastewater drain-fields in a portion of Ogden Valley with relatively high concentrations of nitrate-nitrogen in the water table aquifer and with relatively high hydraulic conductivity. Nitrogen pathways and pools were simulated in NLEAP-GIS 4.2 for lawns, irrigated cropland and drain-fields in the western portion of the South Fork of Ogden River in Ogden Valley watershed. The study was motivated by results from the preceding studies which showed that: 1) 22 percent of the total nitrate-nitrogen loadings to Pineview Reservoir emanated from ground water in the water table aquifer; 2) the study area had two monitoring wells whose hydraulic conductivities bracketed the conductivity range for the entire water table aquifer; 3) the means of the nitrate-nitrogen concentrations in the two wells were statistically different; and 4) agricultural fertilizers and onsite wastewater were identified as probable key sources of ground water nutrient loading.

The study showed that drain-fields and lawns could have annual leaching rates that were, respectively, 2.6- and 1.1-fold higher than the croplands. Total leaching losses of N from the lawns and croplands were, respectively, 50- and 70-fold higher than from the drain-fields. The lawns and drain-fields had lower total leaching losses than the cropland because their respective total areas were substantially smaller than that of the cropland.

Simulation results showed that a 50 percent reduction in the recommended lawn fertilizer application rate could result in a 36 percent decline in the leaching rate. Leaching rates from lawns under the recommended fertilizer and irrigation application rates (of 245 kg ha^{-1} yr^{-1} and 13 mm per irrigation event, respectively) were 9 percent lower than those under irrigation application rates of 25 mm per event (practiced by average Utah homeowners). The results imply that a 45 percent reduction in lawn leaching losses would result if average Utah homeowners reduced lawn fertilizer application rates by 50 percent and irrigated their lawns at the recommended application rate. Simulation results showed that decreasing both irrigation and fertilizer application rates to 50 percent of the recommended application rates would lower the leaching rates by 46 percent. This would not be a viable option because it could supply inadequate water thereby wilting the turf. The results show that reduction of lawn fertilizer application by utmost 50 percent of the recommended rate and increasing irrigation application efficiency by implementing precision irrigation techniques would be a realistic and effective management practice for controlling the impact of lawns on ground water quality. The 50 percent reduction in fertilizer application would be expected to lower the quality of lawns. The effect would be small since the reduction in fertilizer amount is expected to be offset by an increase in soil residual nitrate + nitrite nitrogen due to reduction in the amount of deep percolating water emanating from improved irrigation water management.

The study showed that NLEAP-GIS 4.2 can be used to estimate soil residual nitrate. This was evidenced from the similarity between the NLEAP-GIS 4.2 estimated soil residual nitrate-N of 4.0 mg N kg^{-1}, and the 95 percent confidence interval (0 – 8.7 mg N kg^{-1}) of the measured soil residual nitrate-N values at well 8 and 9 (located less than 1.4 km from well8, outside the study area, sector 4-8). The study also showed that ground water nitrate-N concentrations for the water table aquifer can be estimated from NLEAP-GIS 4.2 leaching simulations and the background concentration of nitrate-N. This was portrayed by the intersection of the 95 percent confidence intervals for the measured nitrate-N concentrations from wells 4 and 8 (1.2 – 9.0 mg N L^{-1}), and the estimated nitrate concentrations based on the background concentration and nitrate leaching simulation results for sector 4-8 (1.6 – 2.2 mg N L^{-1}).

The study was limited by the absence of lawn mowing options in the NLEAP-GIS 4.2 Package. The effect of this limitation was minimized by representing mowing with weekly harvests of turf grass. Another source of uncertainty was on the incomplete data on the number, size and location of drain-fields and lawns. The uncertainty is believed to have been minimized since the median drain-field size and effluent application rate from the available data were used to represent the missing drain-field unit data. Data on lawn location and size were obtained from recent Google imagery and is believed to be reasonably accurate.

Engineering significance

The major contribution of the research is to provide water managers, environmental quality specialists, water users, and other stakeholders a more detailed understanding of the unconfined aquifer ground water flow and its role in nutrient loading to Pineview Reservoir and similar water bodies elsewhere. Water engineers and managers are faced with the challenge of deciding how to allocate ground water among competing uses in order to supplement surface water supplies. Environmental engineers, scientists and managers are concerned about how different management practices impact the quality of water and how the impact would be mitigated or eradicated. Water users are faced with the challenge of accessing water with qualities adequate for its uses, in the right amount, at the right time, and at a reasonable cost.

Knowledge of ground water quantity and quality would help water managers make informed decisions about allocating and/or treating the resource. The managers will be able to determine the appropriate amount of water to allocate to water users at a particular time of the year. They will also be able to determine whether or not the water needs to be treated and for what water quality attributes. Environmental engineers and specialists would use the ground water quality knowledge to develop site-specific best management practices that would be employed to control ground water pollution and its subsequent impact on surface water bodies such as lakes and reservoirs. Surface water pollution creates environmental concerns such as fish kills, skin disorders and/or death in humans who are exposed to cyanotoxins. The environmental quality specialists would also use the ground water quality knowledge to plan and implement water pollution remediation strategies if need be. Sources of ground water pollution include nutrient loading from agricultural land, septic systems and lawns. It is vital to know the proportionate contributions of each source towards water pollution of a given water body in order to make site-specific decisions on the best management practices that need to be applied.

Currently there are no studies available to explain the contribution of ground water from the shallow unconfined (water table) aquifer towards the quantity and quality of Pineview Reservoir in Ogden Valley, Utah. No study in the past has explained the spatial distribution of ground water flow parameters and nutrient loading from the shallow unconfined aquifer towards Pineview Reservoir. The proportionate contributions of irrigated agriculture, lawns and septic system effluent drain-fields towards the unconfined aquifer ground water nutrient loading in Ogden Valley have not been thoroughly explored. No previous study in the valley has documented the nitrogen pool and pathways associated with irrigated crop land, lawns and septic systems except for general estimates of leaching losses from crop land and septic systems (Lowe and Wallace, 1999; Lowe and Miner, 1990; Tetra Tech, 2002). The current study helps fill the knowledge gap on the pool and pathways of nitrate-nitrogen in the valley.

The work has built a platform on which decisions about ground water use and nutrient loading control in Ogden Valley, Utah, can be based by providing a better understanding of the flows and proportionate nutrient contributions by source. Some of the ways in which the research work results will help decision makers is through determination of the best management practices needed to control ground water nitrate loading from lawns, croplands and drain-fields. The management practices may include reduction of the fertilizer application rates to lawns and precision irrigation based on crop or lawn water needs and soil drainage properties. Coarse grained soils may need controlled addition of organic matter in order to improve their water holding capacities thereby reducing leaching losses. The results will also help Huntsville Town Council to decide whether or not to construct a central sewer system to control nutrient loadings from sewage effluent. Their decision would be based on the cost-effectiveness of constructing the sewer system based on the nitrate loadings estimated from the study.

Future research in Ogden Valley and elsewhere may use the methodologies and results from this study to assess both surface and ground water quality impacts emanating from various sources. The work would also contribute towards development of integrated watershed management plans aimed at protecting ground water in the water table aquifer from contamination and avoiding over exploitation of the resource.

Recommendations

The following recommendations have been made based on the approaches, methods, and results demonstrated in this dissertation.

1. There is a need to implement best management practices on both croplands and lawns in Ogden Valley, Utah, and similar inhabited mountain valleys dominated by irrigated agriculture. Applicable best management practices include precision fertilizer and irrigation water applications.
2. A field study on soil nitrogen residual variations in the croplands, lawns and drain-fields needs to be conducted in Ogden Valley in order to ground proof the simulated results. This was beyond the current study due to time and resource constraints.
3. Management of waste water disposal should be implemented including conducting regular checks on the performance of the onsite wastewater treatment systems to ensure that any system failures are corrected in order to reduce ground water pollution.

4. Future modifications of the NLEAP-GIS 4.2 program need to incorporate lawn and septic system drain-field management to facilitate nitrate + nitrite pool and loss simulations for since the study observed high leaching rates from the lawns and drain-fields.
5. Limit the number of additional septic system permits in Ogden Valley and elsewhere in order to protect ground water from more pollution. Future installation of a central sewer system in Ogden Valley needs to be considered if the number of people living in the area is allowed to rise to levels that may need a large increase in septic system density.
6. There is a need for stakeholders in Ogden Valley and elsewhere to develop and implement ground water monitoring plans in order to be able to evaluate the impact of implementing best management practices.
7. The spatial distribution of ground water flow parameters and nutrient transport need to be considered whenever estimates of ground water flow and/or nutrient loading are made. It is common practice to assume that the aquifer is homogeneous yet most, if not all aquifers, including the one in Ogden Valley, are heterogeneous.
8. There is a need for more studies on phosphorus transport through ground water. Some wells monitored in this study reported unexpectedly high total dissolved phosphorus concentrations signifying the possibility that phosphorus transport in ground water may not be as low as has been believed.

9. Soils with high leaching rates need to be amended through addition of required amounts of organic matter based on soil survey results. The soil amendment would help reduce leaching rates since organic matter improves the water holding capacity of the soil. The amendment could be done to croplands and new lawns through addition of compost, animal manure or incorporation of crop residues.
10. Nutrient control programs need to always incorporate ground water pollution control since ground water quality impacts surface water as evidenced from this study.

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APPENDICES

Appendix A.   Permission-to-Reprint Journal Article

Journal Publisher-Permission

August 25, 2012

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Transactions of the ASABE
American Society of Agricultural and Biological Engineers
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August 25, 2012

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Signed Brady Worwood

Date 8/29/2012

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Signed Lindsey D. Carrigan

Date 8/10/12

Appendix B. Ground water monitoring well construction

Ground water monitoring wells were constructed in Ogden Valley, Utah's Pineview Reservoir bank following guidelines from the State of Utah Water Well Handbook (UDWR, 2008). Figure 33 shows the locations of the wells overlaid on a surface contour map (USGS, 2009). The wells were constructed in three phases; 1) wells 1 through 5 were constructed from 10 through 12 February 2010; 2) wells 6 and 7 were constructed on 10 and 11 November 2010; and wells 8 and 9 were constructed on 11 April 2011. All the wells fully penetrated the shallow unconfined aquifer. Figure 3 shows the general specifications for the monitoring wells.

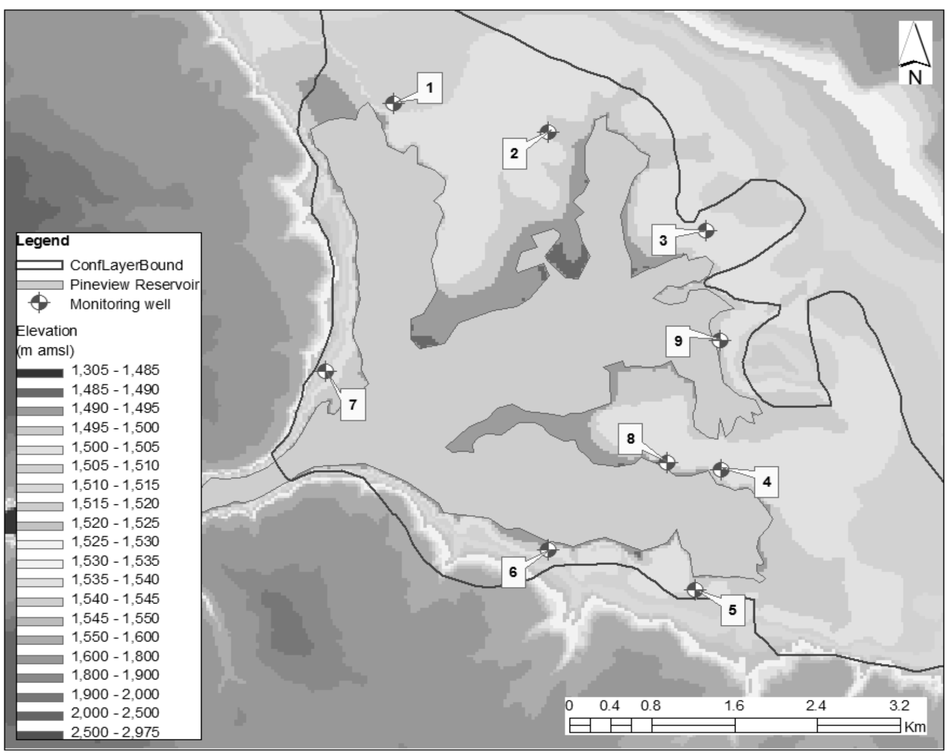

Figure 33. Locations of ground water monitoring wells in Ogden Valley, Utah. Wells are labeled 1 through 9

Identification of the locations was done using Arc GIS, Google Earth map and a hand-held Global Positioning System (GPS) device. The locations were based on the following criteria:

1. Ease of access: the wells were located within the road right of way of Weber County, Utah Department of Transportation (UDOT), or Huntsville City;
2. Proximity to the reservoir: all the wells were constructed within 610 m of the reservoir in order to ensure that the flows observed in the wells are a true representation of the water entering the reservoir through the shallow unconfined aquifer;
3. Proximity to wells whose logs were accessible: the monitoring wells were constructed within a distance of 610 m from existing deep wells in order to be

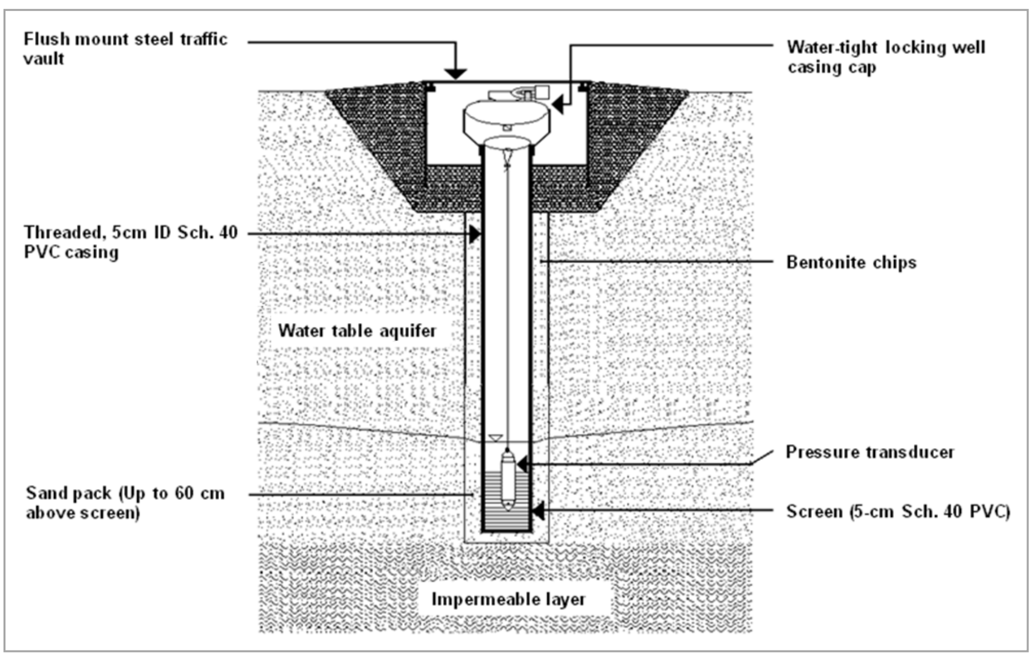

Figure 34. General specifications for ground water monitoring wells. Modified after the State of Utah Water Well Handbook (UDWR, 2008)

4. able to utilize the well logs in estimating depth to the upper limit of the confining layer;
5. Possible locations of flow lines based on direction of surface elevation contours (Fig. 2) from the National Elevation Dataset (USGS, 2009). The underlying assumption was that ground water flow is parallel to the direction of the surface slope (Tetra Tech, 2002); and
6. Presence of the confining layer beneath the surface: the wells were constructed within the soil and aquifer formation which overlays the confining layer (Figs. 1, 2 and 3) reported by Snyder and Lowe (1998).

Construction of the wells was done after obtaining permissions from the Weber County, Utah Department of Transportation, Huntsville City, Utah Blue Stakes and Utah Division of Water Rights. More accurate GPS coordinates and well-cap elevations above mean sea level were determined from topographic surveys.

Appendix C. Nutrient Limitation Study

A nutrient bioassay was conducted in 2008 fall season to determine the nutrient(s) that limited primary production in Pineview Reservoir. The bioassay was conducted on water samples from a mid-reservoir location whose water was assumed to be well mixed. The reservoir water samples were filtered through 153 µm Nitex Screen to remove macro-zooplanktons. The filtered water was thoroughly mixed and distributed into 12 acid washed 150-ml Erlenmeyer flasks in aliquots of 120 ml. There were separate treatments for controls, nitrogen, phosphorus, and nitrogen and phosphorus. Each treatment was replicated thrice. Respective nitrogen and phosphorus additions were at rates of 3.5 mg NH_4NO_3-N L^{-1} and 0.50 mg $NaHPO_4$-P L^{-1}, representing a molar ratio of 7:1 needed for balanced growth (Dodds, 2002). No nitrogen or phosphorus was added to the controls.

The flasks were incubated for 20 days at 19 ± 2 °C with photosynthetically active radiation of 150 µ moles s^{-1} m^{-2} and a photoperiod of 18 hours light and 6 hours darkness. The flasks were set at random, gently swirled and relocated every day to ensure sample mixing and minimize bias due to differences in light exposure. Four sample extractions for chlorophyll *a* analysis were done during the incubation period at an average interval of five days. An aliquot of 20 ml was withdrawn from each flask and passed through a 0.7 µm filter. The filters were frozen and chlorophyll *a* extractions done after day 20 of the bioassay set up. Chlorophyll *a* concentrations were determined using the Turner 10-AU fluorometer and the Welschmeyer method (APHA, 1995; Welschmeyer, 1994).

The chlorophyll *a* concentrations were compared following a natural log-transformed response ratio (RR_x) approach reported in the literature (Elser et al., 2007). The response ratios were computed by dividing each chlorophyll *a* concentration, from assays with nutrient addition, by the chlorophyll a concentration of the corresponding control treatment, and taking the natural logarithm of the quotient (Elser et al., 2007). Two-factor analysis of variance (ANOVA) with replication was conducted on the natural log-transformed response ratios. The analysis comprised 12 (i.e. 3 replicates x 4 chlorophyll *a* extractions) RR_x values per treatment.

ANOVA results (Table 14) showed that the RR_x means of the assays into which both nitrogen and phosphorus were added were significantly higher ($p < 0.0001$, $\alpha = 0.05$) than the assays with single nutrient additions. Assays with only phosphorus additions had a weak significant difference from those with nitrogen only. The analysis results imply that nitrogen and phosphorus co-limited phytoplankton production in Pineview Reservoir the fall of 2008. Further research needs to be conducted to establish whether or not the co-limitation occurs every year.

Table 14. A summary of analysis of variance (ANOVA) results for comparing natural log-transformed phytoplankton response ratios (RR_x)

Parameters	df	Mean squares	F	*P*-value
RR_N vs RR_P	1	2.13	6.17	0.024
RR_N vs RR_{NP}	1	45.2	151	1.47E-09
RR_P vs RR_{NP}	1	27.7	81.6	1.11E-07

Appendix D. Ground water total dissolved phosphorus (TDP) concentrations

Table 15. Measured ground water TDP concentrations (μg P L^{-1}) for 22 February 2010 through 14 November 2011 in Ogden Valley, Utah

Date	Well Number								
	1	2	3	4	5	6	7	8	9
22/02/2010	6	17	6	188	459	N.A.	N.A.	N.A.	N.A.
02/03/2010	12	729	15	226	314	N.A.	N.A.	N.A.	N.A.
05/04/2010	8	110	23	993	327	N.A.	N.A.	N.A.	N.A.
19/04/2010	45	8	25	267	305	N.A.	N.A.	N.A.	N.A.
04/05/2010	110	79	27	398	320	N.A.	N.A.	N.A.	N.A.
25/05/2010	294	54	8	276	332	N.A.	N.A.	N.A.	N.A.
08/06/2010	18	88	7	480	316	N.A.	N.A.	N.A.	N.A.
22/06/2010	14	25	24	420	340	N.A.	N.A.	N.A.	N.A.
20/07/2010	11	10	20	474	374	N.A.	N.A.	N.A.	N.A.
03/08/2010	18	17	30	418	342	N.A.	N.A.	N.A.	N.A.
25/08/2010	17	10	28	270	334	N.A.	N.A.	N.A.	N.A.
07/09/2010	21	23	16	284	336	N.A.	N.A.	N.A.	N.A.
28/09/2010	67	25	28	288	324	N.A.	N.A.	N.A.	N.A.
05/10/2010	27	N.D.	43	286	320	N.A.	N.A.	N.A.	N.A.
12/10/2010	932	204	43	548	330	N.A.	N.A.	N.A.	N.A.
09/11/2010	211	33	35	310	326	N.A.	N.A.	N.A.	N.A.
07/12/2010	72	65	27	218	273	N.D.	90	N.A.	N.A.
13/01/2011	63	30	33	238	281	N.D.	7	N.A.	N.A.
08/02/2011	69	17	29	253	306	N.D.	7	N.A.	N.A.
22/03/2011	10	6	27	291	280	N.D.	4	N.A.	N.A.
19/04/2011	17	15	31	282	236	157	17	102	502
03/05/2011	11	14	32	238	247	150	12	105	437
07/06/2011	14	18	29	255	287	160	16	64	424
22/08/2011	17	18	34	253	303	342	45	116	521
19/09/2011	5	15	25	226	305	67	34	108	824
17/10/2011	6	15	42	245	305	113[a]	N.D.	947	890
14/11/2011	17	42	31	443	318	158	19	90	1265

N.D. = not determined; N.A. = not applicable: the well was not constructed yet; [a] = the value was linearly interpolated because the observed concentration (2,143 μg P L^{-1}) was considered an extreme value.

Appendix E. Ground water soluble reactive phosphorus (SRP) concentrations

Table 16. Measured ground water SRP concentrations (μg P L^{-1}) for 22 February 2010 through 14 November 2011 in Ogden Valley, Utah

Date	Well Number								
	1	2	3	4	5	6	7	8	9
22/02/2010	2.3	3.3	2.0	190	101	N.A.	N.A.	N.A.	N.A.
02/03/2010	2.3	3.3	5.3	218	231	N.A.	N.A.	N.A.	N.A.
05/04/2010	2.1	2.6	22	205	304	N.A.	N.A.	N.A.	N.A.
19/04/2010	1.6	2.1	23	225	299	N.A.	N.A.	N.A.	N.A.
04/05/2010	2.1	4.5	22	229	317	N.A.	N.A.	N.A.	N.A.
25/05/2010	2.9	9.2	5.6	239	159	N.A.	N.A.	N.A.	N.A.
08/06/2010	9.2	4.2	3.3	219	302	N.A.	N.A.	N.A.	N.A.
22/06/2010	7.0	2.9	2.9	221	322	N.A.	N.A.	N.A.	N.A.
20/07/2010	6.0	2.4	15	215	336	N.A.	N.A.	N.A.	N.A.
03/08/2010	5.6	2.4	21	220	337	N.A.	N.A.	N.A.	N.A.
25/08/2010	9.2	3.3	22	215	299	N.A.	N.A.	N.A.	N.A.
07/09/2010	7.9	2.4	6.5	218	302	N.A.	N.A.	N.A.	N.A.
28/09/2010	6.0	2.4	23	219	276	N.A.	N.A.	N.A.	N.A.
05/10/2010	7.9	N.D.	22	225	313	N.A.	N.A.	N.A.	N.A.
12/10/2010	5.1	2.0	17	200	270	N.A.	N.A.	N.A.	N.A.
09/11/2010	8.8	4.2	24	218	293	N.A.	N.A.	N.A.	N.A.
07/12/2010	7.4	3.8	20	211	289	N.D.	2.9	N.A.	N.A.
13/01/2011	8.0	5.3	29	215	264	N.D.	5.6	N.A.	N.A.
08/02/2011	8.6	6.2	23	215	291	N.D.	5.0	N.A.	N.A.
22/03/2011	8.3	5.6	25	221	280	N.D.	6.2	N.A.	N.A.
19/04/2011	8.0	6.5	25	215	221	93	8.9	59	165
03/05/2011	8.9	6.2	24	207	241	100	8.3	88	191
07/06/2011	11	6.8	26	212	267	98	9.4	52	286
22/08/2011	9.4	4.7	24	224	269	115	7.7	93	444
19/09/2011	9.4	4.7	22	200	288	60	7.1	108	493
17/10/2011	10	5.9	24	211	279	118	7.4	86	493
14/11/2011	7.1	6.5	26	205	143	99	4.7	84	459

N.D. = not determined; N.A. = not applicable: the well was not constructed yet.

Appendix F. Ground water dissolved organic carbon (DOC) concentrations

Table 17. Measured ground water Dissolved Organic Carbon concentrations for 22 February 2010 through 14 November 2011 in Ogden Valley, Utah

Date	Well Number								
	1	2	3	4	5	6	7	8	9
05/04/2010	6.9	4.9	6.7	8.7	9.8	N.A.	N.A.	N.A.	N.A.
19/04/2010	6.3	4.1	6.6	10	12	N.A.	N.A.	N.A.	N.A.
04/05/2010	2.6	3.1	5.5	8.8	10	N.A.	N.A.	N.A.	N.A.
08/06/2010	4.6	5.0	4.6	11	11	N.A.	N.A.	N.A.	N.A.
22/06/2010	5.4	4.5	5.7	8.5	9.1	N.A.	N.A.	N.A.	N.A.
20/07/2010	3.2	2.4	4.3	5.9	6.7	N.A.	N.A.	N.A.	N.A.
03/08/2010	2.1	2.4	3.5	5.1	5.3	N.A.	N.A.	N.A.	N.A.
07/09/2010	3.7	3.0	5.4	5.3	5.9	N.A.	N.A.	N.A.	N.A.
28/09/2010	4.1	1.4	2.6	4.1	4.6	N.A.	N.A.	N.A.	N.A.
12/10/2010	4.0	2.0	3.1	4.0	3.1	N.A.	N.A.	N.A.	N.A.
09/11/2010	3.1	2.1	3.2	5.2	5.6	N.A.	N.A.	N.A.	N.A.
07/12/2010	6.0	3.0	5.2	4.8	5.8	N.D.	4.1	N.A.	N.A.
13/01/2011	3.9	2.7	4.0	5.7	6.1	N.D.	2.9	N.A.	N.A.
08/02/2011	4.3	2.5	4.5	6.6	6.8	N.D.	2.9	N.A.	N.A.
22/03/2011	6.4	3.2	4.6	6.3	6.8	N.D.	3.2	N.A.	N.A.
19/04/2011	5.7	2.8	4.3	9.8	6.9	4.7	3.4	6.4	9.9
03/05/2011	5.9	2.9	4.6	8.0	7.3	5.3	3.7	5.7	11
07/06/2011	5.6	2.2	4.3	6.7	5.1	4.5	3.0	4.2	8.5
22/08/2011	2.7	1.6	2.5	6.7	6.8	5.0	2.3	3.1	7.1
19/09/2011	2.3	2.3	3.9	8.3	2.7	3.4	2.0	3.6	8.5
17/10/2011	2.0	0.8	2.0	4.5	5.3	2.5	0.0	3.4	6.2
14/11/2011	2.7	2.4	6.1	7.1	4.3	5.0	1.4	7.6	11

N.D. = not determined; N.A. = not applicable: the well was not constructed yet.

Appendix G. Ground water nitrate + nitrite-nitrogen (NO_3+NO_2-N) concentrations

Table 18. Measured ground water NO_3 + NO_2 - N concentrations for 22 February 2010 through 14 November 2011 in Ogden Valley, Utah

Date	Well Number								
	1	2	3	4	5	6	7	8	9
05/04/2010	2.5	2.8	2.7	4.8	3.5	N.A.	N.A.	N.A.	N.A.
19/04/2010	8.4	2.9	2.7	5.0	3.9	N.A.	N.A.	N.A.	N.A.
04/05/2010	4.5	2.6	2.4	4.2	3.5	N.A.	N.A.	N.A.	N.A.
08/06/2010	4.0	2.9	0.1	5.6	1.3	N.A.	N.A.	N.A.	N.A.
22/06/2010	8.9	3.0	0.2	5.6	4.2	N.A.	N.A.	N.A.	N.A.
20/07/2010	6.7	3.7	1.9	5.3	4.4	N.A.	N.A.	N.A.	N.A.
03/08/2010	2.8	3.5	2.9	5.2	4.8	N.A.	N.A.	N.A.	N.A.
05/10/2010	3.7	N.D.	3.0	4.4	3.6	N.A.	N.A.	N.A.	N.A.
12/10/2010	5.4	4.2	2.8	4.0	3.0	N.A.	N.A.	N.A.	N.A.
09/11/2010	5.8	4.3	2.8	0.0	4.0	N.A.	N.A.	N.A.	N.A.
07/12/2010	5.4	1.4	1.0	1.4	1.2	N.D.	0.2	N.A.	N.A.
13/01/2011	10	4.1	2.5	4.9	3.0	N.D.	0.1	N.A.	N.A.
08/02/2011	15	4.7	3.6	6.6	4.1	N.D.	0.1	N.A.	N.A.
22/03/2011	47	3.6	2.6	4.9	4.3	N.D.	1.2	N.A.	N.A.
19/04/2011	16	3.4	2.6	5.2	3.9	0.2	1.6	3.8	12
03/05/2011	28	3.9	3.0	6.0	4.6	0.7	1.4	4.9	13
07/06/2011	8.6	3.9	3.3	5.3	1.7	0.6	0.8	0.1	5.9
22/08/2011	5.5	5.3	2.9	6.3	8.8	0.4	0.4	4.2	3.6
19/09/2011	3.6	5.1	3.2	7.0	4.8	0.4	0.8	4.2	2.3
17/10/2011	4.5	5.0	2.5	7.2	4.6	0.5	0.4	4.8	2.0
14/11/2011	12	5.1	2.5	6.9	1.2	0.3	0.4	4.2	2.1

N.D. = not determined; N.A. = not applicable: the well was not constructed yet.

Appendix H. Ground water total dissolved iron concentrations

Table 19. Measured ground water Total Dissolved Iron concentrations for 22 February 2010 through 14 November 2011 in Ogden Valley, Utah

Date	Well Number								
	1	2	3	4	5	6	7	8	9
02/03/2010	3.272	0.003	0.003	0.003	0.010	N.A.	N.A.	N.A.	N.A.
05/04/2010	4.616	0.043	0.309	0.012	0.022	N.A.	N.A.	N.A.	N.A.
19/04/2010	0.434	0.014	0.027	0.020	0.012	N.A.	N.A.	N.A.	N.A.
04/05/2010	0.147	0.117	0.025	0.014	0.033	N.A.	N.A.	N.A.	N.A.
25/05/2010	0.245	0.035	0.023	0.018	0.037	N.A.	N.A.	N.A.	N.A.
08/06/2010	0.006	0.004	0.008	0.007	0.008	N.A.	N.A.	N.A.	N.A.
22/06/2010	0.036	0.026	0.014	0.092	0.006	N.A.	N.A.	N.A.	N.A.
20/07/2010	0.034	0.033	0.021	0.046	0.027	N.A.	N.A.	N.A.	N.A.
03/08/2010	0.011	0.011	0.011	0.013	0.012	N.A.	N.A.	N.A.	N.A.
25/08/2010	0.003	0.006	0.004	0.005	0.006	N.A.	N.A.	N.A.	N.A.
07/09/2010	0.005	0.006	0.006	0.005	0.004	N.A.	N.A.	N.A.	N.A.
28/09/2010	0.024	0.002	0.005	0.004	0.007	N.A.	N.A.	N.A.	N.A.
12/10/2010	0.024	0.002	0.005	0.004	0.007	N.A.	N.A.	N.A.	N.A.
09/11/2010	0.005	0.002	0.002	0.004	0.013	N.A.	N.A.	N.A.	N.A.
07/12/2010	0.005	0.002	0.005	0.004	0.002	N.D.	0.063	N.A.	N.A.
13/01/2011	0.009	0.006	0.006	0.007	0.007	N.D.	0.139	N.A.	N.A.
08/02/2011	0.011	0.006	0.003	0.007	0.003	N.D.	0.066	N.A.	N.A.
22/03/2011	0.007	0.002	0.003	0.004	0.007	N.D.	0.005	N.A.	N.A.
19/04/2011	0.006	0.002	0.002	0.048	0.002	0.010	0.008	0.007	0.013
03/05/2011	0.018	0.013	0.014	0.010	0.011	0.008	0.026	0.015	0.019
07/06/2011	0.019	0.022	0.014	0.019	0.020	0.012	0.033	0.023	0.033
22/08/2011	0.003	0.005	0.005	0.003	0.002	0.002	0.004	0.034	0.006
19/09/2011	0.018	0.039	0.015	0.026	0.013	0.013	0.033	0.013	0.029
14/11/2011	0.045	0.026	0.016	0.015	0.017	0.043	0.155	0.007	0.017

N.D. = not determined; N.A. = not applicable: the well was not constructed yet.

Appendix I. Water table elevations measured during grab sampling

Table 20. Water table elevations measured on grab sampling days between 3 May 2010 and 15 November 2011 in Ogden Valley, Utah

Well Number	1	2	3	4	5	6	7	8	9
Well Cap Elevation (m amsl)	1507	1502	1503	1501	1505	1501	1507	1499	1500
Well Depth (m)	8.5	5.8	12.8	6.4	7.5	7.6	12.2	10.6	7.5
04/05/2010	7.6	4.6	6.1	4.9	5.4	N.A.	N.A.	N.A.	N.A.
06/05/2010	7.7	4.6	6.1	4.9	5.4	N.A.	N.A.	N.A.	N.A.
25/05/2010	7.6	4.6	6.0	4.8	5.5	N.A.	N.A.	N.A.	N.A.
01/06/2010	5.6	3.9	5.9	4.8	5.5	N.A.	N.A.	N.A.	N.A.
08/06/2010	5.6	3.9	5.9	4.8	N.D	N.A.	N.A.	N.A.	N.A.
16/06/2010	5.6	3.5	5.9	4.8	5.5	N.A.	N.A.	N.A.	N.A.
22/06/2010	5.4	3.5	5.8	4.8	5.5	N.A.	N.A.	N.A.	N.A.
20/07/2010	5.3	3.3	5.6	4.8	5.4	N.A.	N.A.	N.A.	N.A.
03/08/2010	5.5	3.4	5.5	4.8	5.4	N.A.	N.A.	N.A.	N.A.
25/08/2010	5.7	3.5	5.5	4.7	5.4	N.A.	N.A.	N.A.	N.A.
07/09/2010	6.3	3.4	5.6	4.7	5.4	N.A.	N.A.	N.A.	N.A.
28/09/2010	6.8	3.9	5.6	4.8	5.4	N.A.	N.A.	N.A.	N.A.
12/10/2010	7.0	4.2	5.6	4.8	5.4	N.A.	N.A.	N.A.	N.A.
09/11/2010	7.1	4.4	5.7	4.9	5.4	N.A.	N.A.	N.A.	N.A.
07/12/2010	7.2	4.7	5.9	4.9	5.4	N.A.	8.6	N.A.	N.A.
15/12/2010	7.3	4.6	5.9	4.9	5.4	N.A.	8.2	N.A.	N.A.
13/01/2011	7.3	4.3	5.8	4.8	5.4	N.A.	6.9	N.A.	N.A.
14/01/2011	7.3	4.3	5.7	4.8	5.4	N.A.	N.D.	N.A.	N.A.
08/02/2011	7.2	4.1	5.7	4.8	5.4	6.7	5.6	N.A.	N.A.
22/03/2011	5.9	3.4	5.4	4.5	5.2	N.D.	N.D.	N.A.	N.A.
19/04/2011	5.9	3.2	5.1	4.4	4.9	5.0	1.1	5.6	3.7
03/05/2011	6.1	3.4	5.2	4.4	4.8	5.2	2.5	5.5	3.9
07/06/2011	6.4	3.6	5.4	4.6	4.8	5.4	3.5	5.1	4.0
29/06/2011	5.5	3.4	5.3	4.6	4.9	4.3	4.8	4.3	4.8
22/08/2011	5.1	3.2	5.2	4.7	5.0	4.7	6.9	4.7	4.0
19/09/2011	5.7	3.4	5.4	4.7	5.1	5.0	7.6	5.2	4.8
17/10/2011	6.5	3.5	5.5	4.8	5.0	5.3	7.6	5.5	5.5
14/11/2011	7.0	4.0	5.7	4.9	4.1	5.0	7.7	5.6	5.6

N.D. = not determined; N.A. = not applicable: the well was not constructed yet.

Appendix J. A multiple regression equation for extrapolating ground water flow

The following multiple regression equation was used to extrapolate median ground water flows through the water table aquifer to Pineview Reservoir for 15 November 2010 through 30 April 2011 (the period during which water table elevation data for wells 8 and 9 were not available).

$$G = -255.56\,R + 201.8\,W_1 + 123.80\,W_2 + 108.95\,W_3 + 85.88\,W_5 + 136.97\,W_7 - 603{,}324$$

where G is the extrapolated ground water flow ($m^3\ d^{-1}$); R the reservoir elevation (m amsl); W_1, W_2, W_3, W_5, and W_7 are observed ground water elevations (m amsl) for wells 1, 2, 3, 5, and 7 during the extrapolation period.

The equation was empirically derived from modeled median ground water flows and observed reservoir and ground water elevations for wells 1 through 9 from1 May 2011 through 14 November 2011. Assumptions for use of this equation were as follows:

1. Assumptions 2, 4, and 5 in Appendix B;
2. Ground water flows through the water table aquifer to the reservoir were linearly related to ground water and reservoir elevations; and
3. Existence of steady state conditions on each day for which the flow was computed.

Appendix K. Soil sample core analysis

Table 21. Analysis results for soil sample cores collected on 11 April 2011 when drilling wells 8 and 9, respectively

Appendix K. Soil sample core analyses data

Table 21. Analysis results for soil sample cores collected on 11 April 2011 when drilling wells 8 and 9, respectively

Well No.	Depth (m)	pH	Olsen $NaHCO_3$	2 N KCl Extractable		Walkly-Black	Hydrometer			
			PO_4-P (mg kg^{-1})	NH_4-N (mg kg^{-1})	NO_3+NO_2-N (mg kg^{-1})	Organic Matter	Per-cent Sand	Per-cent Silt	Per-cent Clay	Texture
8	2.74-3.2	8.3	4.5	<1.25	<1.25	0.0	93	3.5	3.3	Sand
8	4.27-4.72	8.3	4.7	<1.25	<1.25	0.0	94	3.5	2.8	Sand
8	5.79-6.25	8.4	3.9	<1.25	<1.25	0.0	91	6.0	2.8	Sand
8	6.40-7.01	8.2	3.3	<1.25	<1.25	0.1	89	7.5	3.3	Sand
8	7.01-7.32	8.3	6.1	<1.25	<1.25	0.1	88	8.8	2.8	Sand
8	7.32-7.62	8.2	2.9	<1.25	<1.25	0.2	78	18	3.8	Loamy Sand
8	7.62-8.08	8.3	2.2	<1.25	<1.25	0.2	81	16	3.3	Loamy Sand
8	8.23-8.69	8.3	2.7	<1.25	1.255	0.2	81	17	2.8	Loamy Sand
8	8.84-9.07	8.0	7.0	<1.25	<1.25	0.4	21	68	11	Silt Loam
8	9.07-9.30	8.2	3.0	<1.25	<1.25	0.2	56	39	5.6	Sandy Loam
8	9.45-9.91	8.2	1.3	<1.25	<1.25	0.2	74	22	3.8	Sandy Loam/Loamy Sand
8	10.06-10.52	7.8	1.5	<1.25	<1.25	0.6	59	36	4.8	Sandy Loam
8	10.67-11.13	7.7	1.6	<1.25	<1.25	0.6	32	59	8.5	Silt Loam
8	11.28-11.51	7.5	1.6	<1.25	<1.25	1.1	58	33	8.5	Sandy Loam
8	11.51-11.73	7.7	0.8	<1.25	<1.25	0.5	44	48	7.5	Loam
8	11.89-12.65	7.6	2.8	1.795	<1.25	0.6	14	76	10	Silt Loam

Table 21 cont'd

Well No.	Depth (m)	pH	Olsen $NaHCO_3$	2 N KCl Extractable		Walkly-Black	Hydrometer			
			Phosphate (mg kg^{-1})	Ammonium N (mg kg^{-1})	Nitrate N (mg kg^{-1})	Organic Matter	Percent Sand	Percent Silt	Percent Clay	Texture
9	2.74-3.20	6.3	25	<1.25	17.1	0.5	44	38	18	Loam
9	4.88-5.03	6.4	24	<1.25	4.66	0.2	44	39	17	Loam
9	5.18-5.33	6.4	26	<1.25	3.86	0.3	55	30	15	Sandy Loam
9	5.49-5.72	6.7	24	<1.25	2.23	0.1	64	24	12	Sandy Loam
9	5.72-5.94	6.6	23	<1.25	2.00	0.2	79	7.7	13	Sandy Loam
9	6.10-6.25	6.7	26	<1.25	1.93	0.1	75	12	13	Sandy Loam
9	6.25-6.40	6.9	33	<1.25	1.86	0.3	54	23	23	Sandy Clay Loam
9	6.40-6.55	6.9	22	<1.25	2.06	0.2	66	20	14	Sandy Loam
9	6.71-7.16	6.8	17	<1.25	1.97	0.1	82	5.4	13	Sandy Loam
9	7.32-7.77	7.0	35	<1.25	4.70	0.7	0.0	40	60	Silty Clay

CURRICULUM VITAE

Thomas Nyanda Reuben
March 2013

CAREER OBJECTIVE:

To work at a position aimed at fostering development without compromising environmental quality whose requirements include leadership experience, objective oriented management, good interpersonal skills, and good written and spoken English. Special areas of interest: water resources engineering, environmental engineering, Geographical Information Systems, soil and water conservation.

EDUCATION:

PhD in Irrigation Engineering, Utah State University, USA (3/2013).| Dissertation: Nutrient contribution of the shallow, unconfined aquifer to Pineview Reservoir. MS in Irrigation Engineering, Utah State University, USA (12/2002). |Thesis: Trapezoidal flume calibration and development of correction algorithms for Parshall flumes in a computer program. BS in Agriculture, University of Malawi, Malawi (11/1997) Thesis: Calibration of cut-throat flumes.

EXPERIENCE:

RESEARCH ASSISTANT, Utah Water Research Laboratory (12/2012-04/2013)

<u>Ground water research</u>: Collected and analyzed ground water samples for nitrogen, phosphorus, dissolved organic carbon, and total dissolved iron.

<u>Soil sample core analysis</u>: Conducted sequential extraction of nitrogen and phosphorus from soil sample cores

GRADUATE RESEARCH ASSISTANT, Utah Water Research Laboratory (08/2008-12/2012).

Ground and surface water research: Designed and supervised construction of ground water monitoring wells; collected and analyzed stream, reservoir, and ground water samples; analyzed data; compiled reports and presented them at professional gatherings; and published research findings.

Instrumentation: Data loggers for measuring stream flow and monitoring ground water elevations, spectrophotometer, fluorometer, dissolved organic carbon discrete analyzers, flow meters, and dissolved oxygen, pH and conductivity sensors. Software used: HOBOware, Logger Net, GIS, Weather Link, Observations Data Model (ODM) tools, and NLEAP-GIS 4.2.

Irrigation system design and installation: Designed and installed a subsurface drip irrigation system at the Huntsman Onsite Wastewater Treatment Training and Demonstration Site, Utah State University (06/2007).

Image processing: processed satellite and air-borne images at the Remote Sensing Services Laboratory. Instrumentation: Radiometers and imaging cameras. Software used: GIS and ERDAS Imagine (2007-2008).

UNIVERSITY LECTURER IN HYDRAULICS AND HYDROLOGY, University of Malawi, Bunda College of Agriculture, Lilongwe, Malawi (2003-2007).

Teaching: Taught Irrigation System Design, Irrigation Water Management, Irrigation Scheme Management, Rural Water Supply, Mathematics and Physics to undergraduate and graduate students.

Student Research Supervision: Supervised undergraduate students on research work related to irrigation and water supply.

Consultancy: Conducted training of trainers workshops for district assembly staff in eight districts. The workshops were on flood moderation and mitigation of the impacts of floods.

Administrative Role: Acted as the Head of Engineering Department when the Head was away. Engineering Department representative at these college committees: 1) Administration and Academic Staff Welfare; and 2) Admissions and Awards.

IRRIGATION EXTENSION OFFICER, Ministry of Agriculture and Irrigation, Malawi (1999-2001).

Leadership: supervised three Assistant Irrigation Officers under three Rural Development Projects (RDPs) and provided technical support to the Assistant Irrigation Officers.

Operational Plan Development and Budgeting: Developed annual operational plans and budgets for irrigation extension activities for three RDPs and participated in operational planning and budgeting for Lilongwe Agricultural Development Division, an umbrella body for five RDPs.

Participatory Development: Conducted participatory rural appraisals aimed at planning irrigation development innovations with the beneficiaries.

RESEARCH TRAINING, Utah State University (2012)

Grant Writing Workshop: Attended a grant writing workshop organized by the Office of Research and Graduate Students. Facilitator: Dr. Stephen W. Russell, co-founder and managing member of Grant Writers' Seminars and Workshops LLC.

Research Scholar Orientation: Attended a research scholar orientation workshop organized by the Office of Research and Graduate Students. Facilitators: Dr. Jeff Broadbent (Research and Graduate School Associate Vice President and Dean), Dr. Tim Gilbertson (Biology Professor), and Dr. Russell Price (Federal Compliance Manager).

PUBLICATION

Peer-Reviewed Journal Article: T.N. Reuben, B.K. Worwood, L.D. Carrigan, and D.L. Sorensen, *Pineview reservoir nutrient loading, unloading, and the role of ground water in the estimates*, Transactions of the ASABE, Vol. 54, No 6, 2011.

Conference Proceedings: T.N. Reuben, L. DeBoer, B. Worwood, and D.L. Sorensen, *Resolving Pineview Reservoir Anoxia*, TMDL 2010: Watershed Management to Improve Water Quality Proceedings, 14-17 November 2010 Hyatt Regency Baltimore on the Inner Harbor, Baltimore, Maryland USA 711P0710cd

PRESENTATIONS AT PROFESSIONAL GATHERINGS

T.N. Reuben, "Ground water contribution to Pineview Reservoir nutrient loading", Water Environment Association of Utah Midyear Conference, West Valley, Utah, 2011

T.N. Reuben and L.D. Carrigan, “Pineview nonpoint source nutrient loading”, J. Paul Riley Graduate Student Research Competition, American Water Resources Association (Utah Chapter), University of Utah, 2011

T.N. Reuben, L. DeBoer, B. Worwood, and D.L. Sorensen, “Resolving Pineview Reservoir’s anoxia”, TMDL 2010 Watershed Management to Improve Water Quality Conference (organized by the ASABE), Baltimore, Maryland, 2010

T.N. Reuben and D.L. Sorensen, “Onsite wastewater, irrigation return flows, and the Pineview Reservoir TMDL”, 9th Conference of the Utah Onsite Wastewater Association, University of Utah, 2009

AWARDS AND HONORS

Third Place Best Presenter Award (Engineering Section), Utah State University Graduate Research Symposium, 2011

Golden Key International Honor Society Best Student Award, 2007

SKILLS

Language: English (fluent)

Good interpersonal skills: team work and team building in multicultural environments

Computer skills: ERDAS Imagine, Arc GIS, Python, NLEAP-GIS 4.2, QUAL2K, Excel, Word, Access, Power Point, MODFLOW, Visual Basic.Net, VBA, ACA, SIRMOD III, SURFER 8, LEVELGRAM, TAUDEM 5.0, SPSS, SAS, R, LaTeX, CATCH3D, AutoCAD, MATLAB.