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DEPARTMENT OF
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Moses Lake
Total Maximum Daily Load
Groundwater Study

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DEPARTMENT OF
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Moses Lake
Total Maximum Daily Load
Groundwater Study

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by
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Environmental Assessment Program
Olympia, Washington 98504-7710

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Abstract

Moses Lake has historically exhibited eutrophic or hypereutrophic conditions, and is on the 1996 federal Clean Water Act 303(d) list as an impaired waterbody. Phosphorus has been identified as the limiting nutrient for the lake. Based on characteristic uses of the lake, an in-lake total phosphorus concentration target of 0.050 mg/L has been proposed to manage water quality concerns. In order to develop an allocation strategy for phosphorus loading to the lake, a Total Maximum Daily Load (TMDL) study was conducted by the Washington State Department of Ecology (Ecology) Environmental Assessment Program, at the request of Ecology's Eastern Regional Office.

To better characterize the concentration and potential source of nutrients in groundwater directly discharging to the lake, 12 lake-bed piezometers were installed in the littoral zone of the lake along portions of the shoreline judged from regional data to be receiving groundwater discharge. Water quality samples were collected from the piezometers, as well as from one near-shore domestic well and two surface springs, during May, July, and October of 2001. Samples were analyzed for orthophosphate as P (OP), total dissolved phosphorus, nitrate+nitrite-N, ammonia-N, total persulfate nitrogen, chloride, total dissolved solids, and dissolved iron and manganese.

The majority of stations (75%) exhibited groundwater OP concentrations above the 0.050 mg/L surface water target criteria. Concentrations of OP in groundwater generally increased from north to south, paralleling increases in concentrations of parameters that indicate anthropogenic (human-caused) impact to water quality. A statistically significant relationship was established between OP concentration and the relative percentage of urban development upgradient of each station. These findings suggest that urban releases of wastewater to the aquifer are the primary source of phosphorus entering the lake via groundwater discharge. Loading calculations predict an annual OP mass flux to the lake from approximately 400 to 40,000 kg_{op} per year via groundwater discharge, with a value from 10,000 to 20,000 kg_{op} per year considered the best estimate of field conditions.

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Executive Summary

To support the development of a TMDL for phosphorus for Moses Lake, Washington, the Washington State Department of Ecology (Ecology) Environmental Assessment (EA) Program conducted a study of the water quality condition of groundwater directly discharging to the lake. The purpose of the study was to characterize the nutrient content of the groundwater entering the lake, and evaluate the possible sources for the nutrients observed. The specific findings regarding groundwater phosphorus contributions to the lake are incorporated into a more comprehensive analysis of lake loading reported separately by Carroll (2003).

To conduct the study, a network of 12 lake-bed piezometers, two surface springs, and one domestic well were sampled for phosphorus and nitrogen, as well as a variety of indicator parameters. Sampling stations were distributed along portions of the shoreline identified as regional aquifer discharge faces. Water quality samples were collected during May, July and October of 2001.

The primary findings of the study are:

- The available evidence indicates that groundwater discharge to the lake occurs primarily along the northwestern and eastern shorelines, with limited additional discharge to the southeastern shoreline. Lake water recharges the surficial aquifer along the southwestern and far southern shorelines. This indicates that from a groundwater perspective, Moses Lake is a flow-through lake.
- The majority of the groundwater that interacts with the lake moves through the unconfined, high permeability gravels, cobbles and boulders of the Pleistocene-age Missoula Flood deposits, with limited direct contribution from the basalt strata. Groundwater discharge to the lake from the adjacent aquifer system is probably concentrated in the near-shore areas of the lake bottom.
- The dominant form of dissolved phosphorus in groundwater in the Moses Lake area is orthophosphate (OP). The area background condition for groundwater OP in the surficial aquifer system is less than 0.05 mg/L as P, the proposed in-lake TMDL target criteria for total phosphorus. Groundwater concentrations of OP measured in the study area greater than 0.05 mg/L are not considered to originate from a natural mineralogic source, but are likely the result of anthropogenic (human-caused) loading of phosphorus to the aquifer. Higher phosphorus concentrations in groundwater at the northernmost sampling station are probably derived from groundwater transport through a limited area of wetland soils, which supply phosphorus through decomposition of organic material.
- Groundwater concentrations of OP and total dissolved phosphorus generally increase from north to south adjacent to the lake, and parallel increases in concentration of parameters indicating anthropogenic impact on water quality. The majority of the sample stations (75%) showed OP concentrations above 0.05 mg/L as P.

- The occurrence and dissolved concentration of phosphorus in the groundwater discharging to Moses Lake is controlled by a combination of a) variations in the natural attenuation capacity of the aquifer, b) variations in the local geochemical conditions, and c) the availability and proximity of upgradient anthropogenic sources. The coarse deposits of the surficial aquifer system appear to have a limited natural phosphorus attenuation capacity, suggesting that the remaining two factors are the most important controls on phosphate occurrence in groundwater discharging to the lake. Specifically, locations where reducing conditions coincide with sources in close proximity to the lake show the highest dissolved phosphorus concentrations, while areas where oxic conditions coincide with low density distant sources show the lowest values. The data suggest that the reducing conditions that favor the presence of dissolved phosphate are frequently the result of depletion of oxygen by anthropogenic loading of organic-rich wastewater.
- Increases in OP concentration in groundwater were positively correlated with the relative area percent of urban development upgradient of the sample station. No systematic relationship was established between groundwater OP and the relative area of agricultural land use upgradient of each station. In view of the data collected during the study, this evaluation suggests that the primary source of the groundwater phosphorus moving to the lake is urban wastewater.
- Wastewater impacting local groundwater quality is probably derived from a combination of sources, including leachate from septic system drain fields, leakage from municipal waste lines, and direct discharge or infiltration of partially treated municipal wastewater. Other land uses and sources, including the downward migration of phosphorus fertilizer from irrigated agricultural fields, likely contribute to raising the ambient condition of OP in the study area groundwater, but are not considered the primary source of the elevated OP observed in the aquifer at the southern end of the lake.
- The results of a loading analysis predict that on an annual basis, a probable range from 10,000 to 20,000 kg of dissolved phosphorus (as OP) enter the lake by groundwater discharge.
- Long-term loading of phosphorus-rich water to an aquifer system may create a reservoir of sorbed phosphorus that can impact groundwater quality for many years, even after the cessation of loading. If such a reservoir is present in the study area subsurface, it is possible that transport of phosphorus to the lake by groundwater discharge may not be a controllable load in the short-term.

Ecology’s Eastern Regional Office also requested a determination of the source of the elevated phosphorus concentrations observed in the Rocky Ford Springs discharge north of Moses Lake. To support this request, Ecology’s EA Program reviewed existing information, and conducted a limited water quality field sampling effort of the springs and several area lakes considered possible sources for the spring discharge. The primary findings of that effort are:

- The hydrogeologic and geochemical evidence collected confirm that Soap Lake is not in hydraulic communication with Rocky Ford Springs, indicating the lake is not the source of the phosphorus in the spring discharge.

- The existing evidence regarding area groundwater flow patterns indicate that the spring discharge is in direct hydraulic communication with shallow groundwater present in a narrow channel of high transmissivity gravel deposits northeast of the springs.
- The ionic composition of the spring water best matches groundwater with a comparatively short residence time in unconsolidated deposits. In contrast, the sample collected from Brook Lake shows an enrichment in sodium (common to waters that have experienced extended contact with basalts) that is not observed in the spring water. These data favor the interpretation that the origin for the bulk of the spring water is the irrigation recharge to the unconsolidated deposits present downgradient of Brook Lake. Additional groundwater sampling and study would be required to confirm if this recharge is also the source for the phosphorus observed at the springs.

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Introduction

Excess nutrients present in Moses Lake have historically led to eutrophic or hypereutrophic conditions during the summer months, resulting in persistent algal blooms that can inhibit the public's use of the lake. As a consequence, the lake was listed as an impaired waterbody on the 1996 Clean Water Act 303(d) list. Carroll et al. (2000) conducted a data review, and concluded that phosphorus is currently the limiting nutrient for the lake. On the basis of characteristic uses of the lake (fishing, jet skiing, swimming, boating) these authors proposed an in-lake total phosphorus (TP) concentration target of 0.050 mg/L to manage water quality concerns. The authors also concluded that gaps in the existing data for the lake prevent the development of an effective management approach.

In response to these findings, the Eastern Regional Office of the Washington Department of Ecology (Ecology) asked Ecology's Environmental Assessment (EA) Program to conduct a comprehensive field study of water quality conditions for the lake during the 2001 water year. The field study was intended to provide an up-to-date and complete assessment of water quality conditions and nutrient loads in the lake's watershed. The ultimate goals of this effort are the development of a water quality model, and an allocation strategy for nutrient loading to the lake. This allocation strategy will be formalized through the development of a Total Maximum Daily Load (TMDL) for phosphorus.

As an integral part of the EA Program's Moses Lake TMDL project, this study focused on improving the understanding of the nutrient content of the direct *groundwater* inflow to the lake, particularly with respect to phosphorus. While groundwater has been identified as a significant contributor to the overall annual nutrient load to the lake (Carroll et al., 2000; Jones, 1988; Carlson, 1983), only a limited number of samples of the groundwater discharging directly into the lake have been collected and analyzed. Characterization of the quality of the groundwater entering Moses Lake will assist in the development of an up-to-date, lake-wide nutrient budget. The specific findings reported here regarding groundwater phosphorus contributions to the lake are incorporated into a more comprehensive analysis of lake loading documented separately by Carroll (2003).

Study Purpose and Goals

The primary goal of this study was to characterize the nutrient content of groundwater that directly discharges into Moses Lake from the study area aquifer system, and evaluate the possible sources for the nutrients observed. The data developed from this study may be used in the construction of a water quality model and nutrient budget for the lake.

Ecology's Eastern Regional Office also expressed an interest in determining the source of the phosphorus concentrations found in the spring water discharging at the head of Rocky Ford Creek, which ultimately drains to Moses Lake (Figure 1)(Carroll et al., 2000; Cusimano and Ward, 1998). Some suggest that Soap Lake, a mineral-rich lake located approximately five miles north of the Rocky Ford Springs, is the source of a significant proportion of the dissolved phosphorus detected in the spring water. An extensive field study of these questions was beyond the resources of Ecology's EA Program to investigate during this study. However, existing information was assembled and reviewed to determine if a particular source area could be identified or ruled out. To support this review, a limited field sampling effort to characterize the hydrochemical profile of the spring water, and compare that profile to the hydrochemistry of several suspected upgradient source lakes also was performed. The results of this evaluation are described in Appendix A.

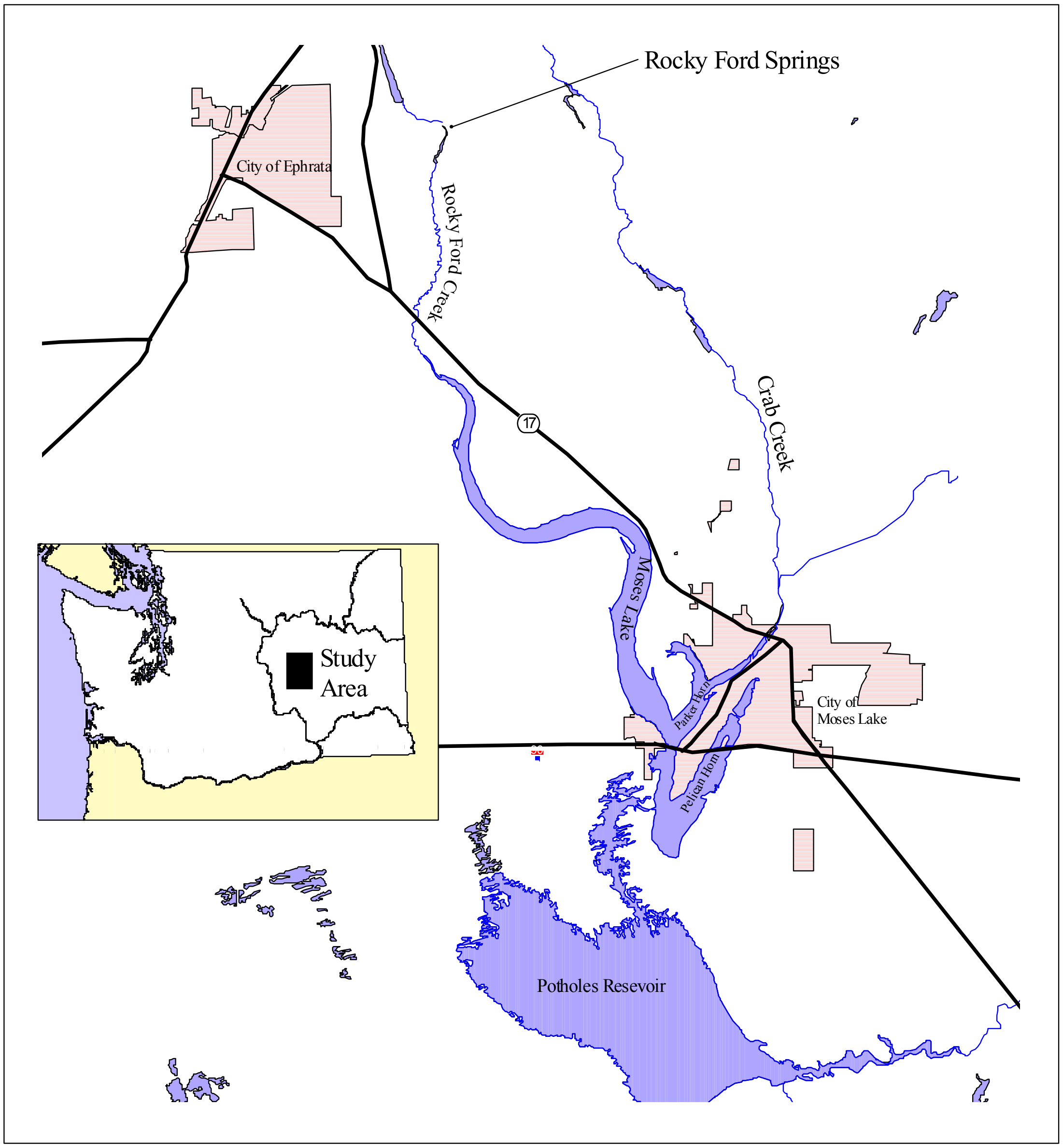

Figure 1 -
Location Map
Moses Lake, WA

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Physical Setting

Study Area Description

Moses Lake is located in east-central Washington State, in Grant County (Figure 1). The lake, which drains to the Potholes Reservoir, lies in the northeastern portion of the Quincy Basin, a sub-province of the Central Columbia Plateau. The lake and surrounding area is located within the boundaries of the Columbia Basin Irrigation Project, an irrigation water management program overseen by the U.S. Bureau of Reclamation (USBR). The lake is used as an important component in the storage and distribution of irrigation water to the agricultural fields of the lower Columbia Basin. As a result, the hydrodynamics of the Moses Lake watershed, including the distribution and movement of groundwater, is strongly influenced by the Columbia Basin Irrigation Project.

Moses Lake is over 20 miles in total length, approximately 11 square miles in total area, and has a mean depth of 18.5 feet (Bain, 1987a). The lake has three major arms; the main arm (also called the Rocky Ford Arm) is located to the north, and Parker and Pelican Horns are to the south, separated by an area known as the lower peninsula (Figure 2). Lewis Horn is a smaller embayment located on the northern side of Parker Horn. Two main surface tributaries drain to the lake: Rocky Ford Creek to the main arm, and Crab Creek into Parker Horn.

Surface discharge from the lake is controlled by two USBR-operated dams located at the southern end of the lake. The lake surface elevation is manipulated by the USBR throughout the year for irrigation management. Just prior to the irrigation season (normally in mid-March) the lake level is set to an elevation of approximately 1046-1047 feet above mean sea level (AMSL), and remains there throughout the summer (Churchill, 2001). At the end of the irrigation season (typically late October) the lake level is lowered to approximately 1041 feet AMSL to create storage capacity for winter/early spring runoff, and to protect and allow maintenance of shoreline structures.

Figure 3 shows the generalized land use in the study area during the late 1980s and early 1990s, as interpreted from remote sensing imagery and other sources (USGS, 1999). The city of Moses Lake is located adjacent to the southeastern portion of the lake, occupying the lands between and surrounding Parker and Pelican Horns (Figure 2). Urban and suburban development is predominant along the southern shorelines of the lake. Extensive irrigated cropland is present to the west, southeast, east and northeast of the lake. Dry range and shrubland is the predominant land use adjacent to the northern shorelines of the lake, interspersed with low density rural development and more limited irrigated agricultural land.

The Grant County Municipal Airport, formerly known as the Larson Air Force Base, is located north of the city. The airport and surrounding properties have been the subject of multiple environmental field investigations over the past ten or more years due to the presence of chlorinated solvents in the underlying groundwater.

Numerous private residences are situated immediately adjacent to the lake shoreline, with particularly dense development along the peninsula between Parker and Pelican Horns, and along the northern shoreline of upper Parker Horn. In total, more than 27,000 people live around the lake, with the majority concentrated in and around the city (Carroll et al., 2000). Rapid development has occurred over the last 15 years in several unincorporated areas beyond the city boundaries, most notably in the Cascade Valley area, and along the southeastern shoreline of Pelican Horn (O'Brien, 2002) (Figure 2). The lake is extensively used for recreational purposes (e.g. boating, fishing, jet skiing, swimming), both by residents and out-of-town recreationists.

Study Area Geology and Hydrogeology

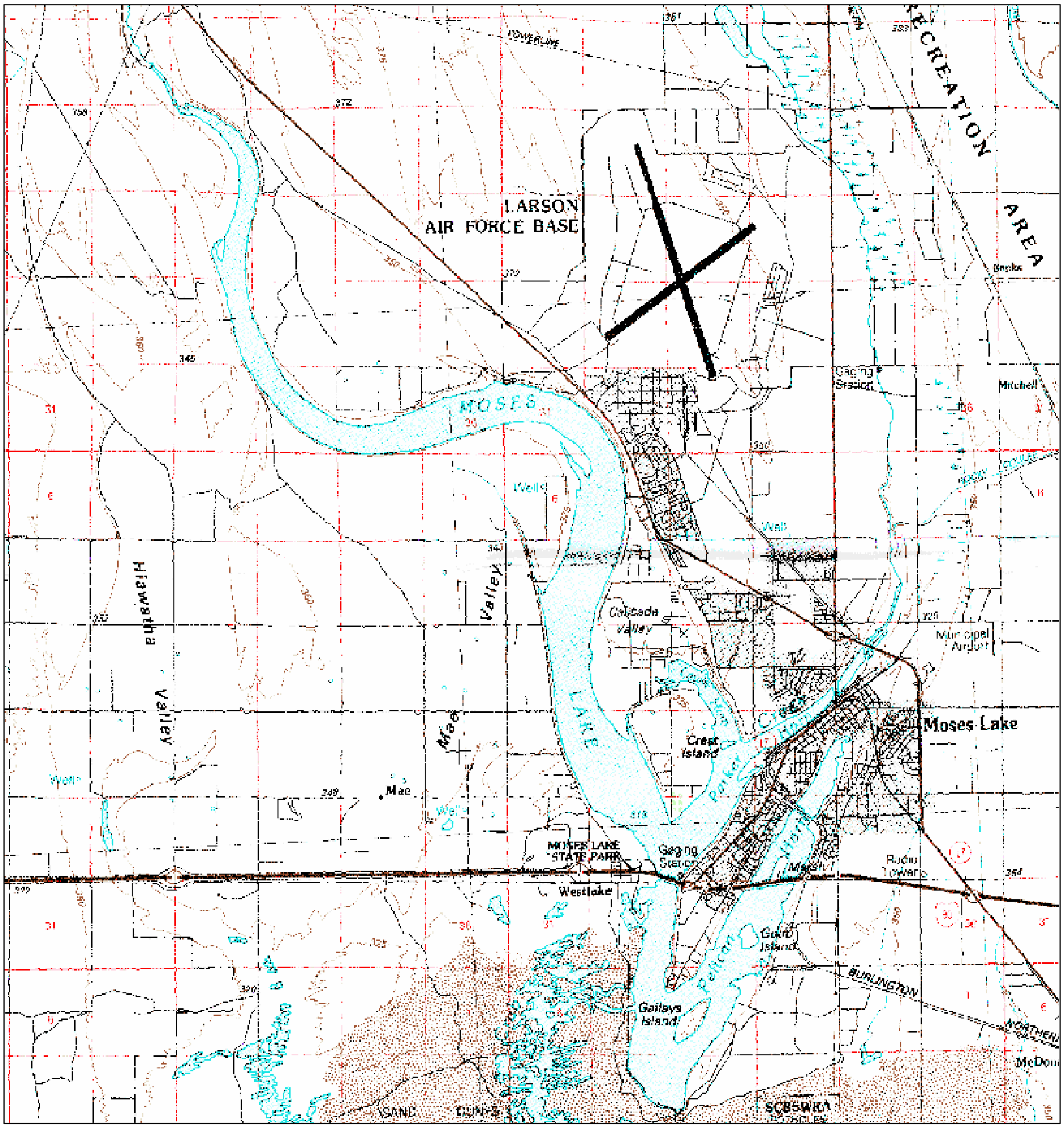
Geology and Soils

Moses Lake is situated within the Quincy Basin, a structural sub-basin of the central Columbia Plateau. The subsurface stratigraphy of the Moses Lake area is comprised of a thick series of broadly folded, Miocene-age flood basalt lava flows and interbedded sediments (collectively known as the Columbia River Basalt Group – CRBG) overlain by unconsolidated deposits of late Miocene to recent age (collectively named the overburden or suprabasalt deposits)(Mundorff et al., 1952; Walters and Grolier, 1960; Grolier and Foxworthy, 1961; Grolier and Bingham, 1971; Grolier and Bingham, 1978; GAI, 1991; USACE, 2001a; Whiteman et al., 1994; Bauer and Hansen, 2000; Gulick, 1990). Figure 4 presents a map of the surficial geology of the study area; Figure 5 presents a generalized northeast-southwest cross-section showing the relative distribution of the major study area stratigraphic units.

In the Moses Lake area the uppermost layers of the CRBG are fractured bedrock of the Wanapum Basalt formation. The most recent basalts underlying most of Moses Lake are mapped as belonging to the Roza Member (Grolier and Bingham, 1971). The basalt system is composed of a series of individual flow units. A typical CRBG basalt flow is characterized by an uppermost fractured and vesicular flow top, a dense columnar and entablature jointed flow interior, and a glassy, rubbly, or pillowed flow bottom. The combination of a flow top and overlying flow bottom from two adjacent flows is called an interflow zone, and is normally significantly more porous than the basalt interior. Regionally, the top surface of the Wanapum Basalt is known to slope towards the southwest beneath the lake, although local depressions or rises on the surface have been reported (Figure 5).

Throughout much of the study area the basalts are directly overlain by fine-grained deposits of the late Miocene to Pliocene-age Ringold Formation. In the Moses Lake area Ringold sediments are comprised of lacustrine clay, silt, and fine sand. Additional deposits of fine-grained, tuffaceous, eolian sand and silt, basaltic gravel lenses, and inter-layered or capping caliche also are present.

Subsurface investigations associated with the former Larson air base indicate that the Ringold sediments pinch out to the east of the lake approximately 1 mile west of the Crab Creek drainage. The Ringold deposits generally thicken to the west, and have been interpreted by previous investigators to separate the lake from the underlying basalt units throughout much of the area



Base map source: USGS Digital Topographic Map, 1: 100,000

Figure 2 -
Study Area Map

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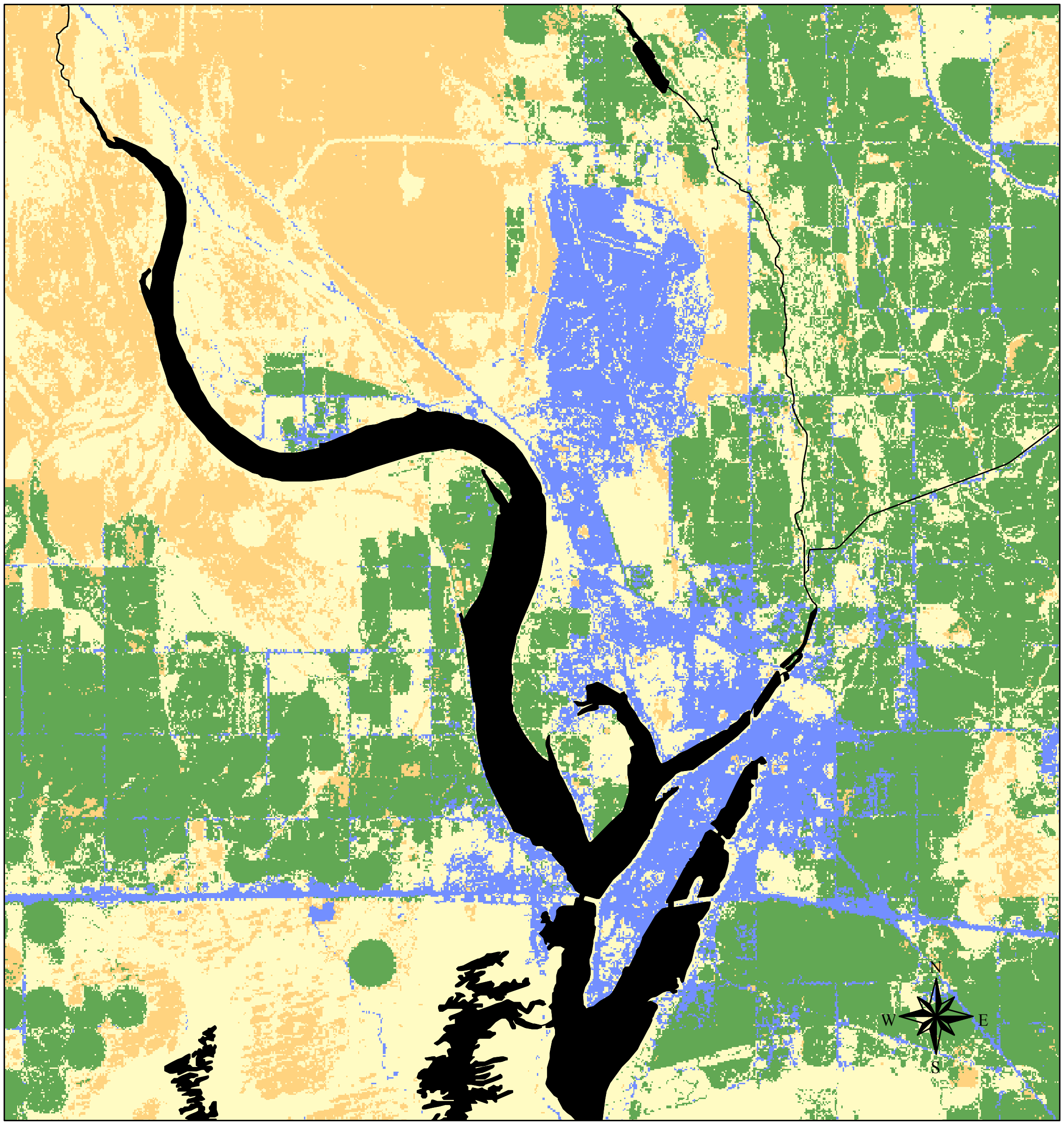


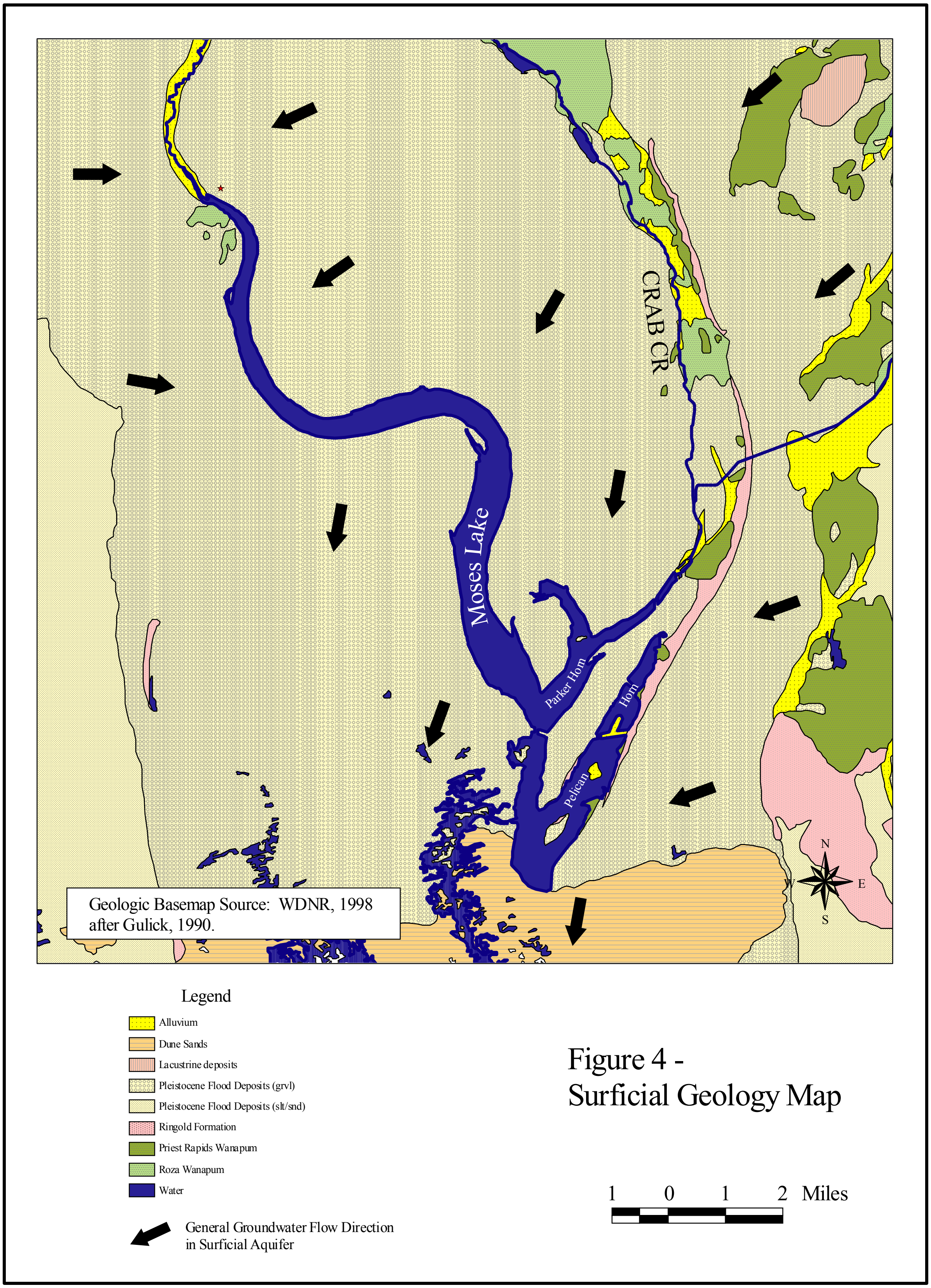
Map Source: USGS, 1999

Legend - Generalized Land Use

- ■ Water
- ■ Developed Lands
- ■ Shrubland/Undeveloped Lands
- ■ Grassland/Grazing
- ■ Agricultural Land

Figure 3 -
Generalized Land Use Map

1 0 1 2 Miles



Geologic Basemap Source: WDNR, 1998 after Gulick, 1990.

Figure 4 - Surficial Geology Map

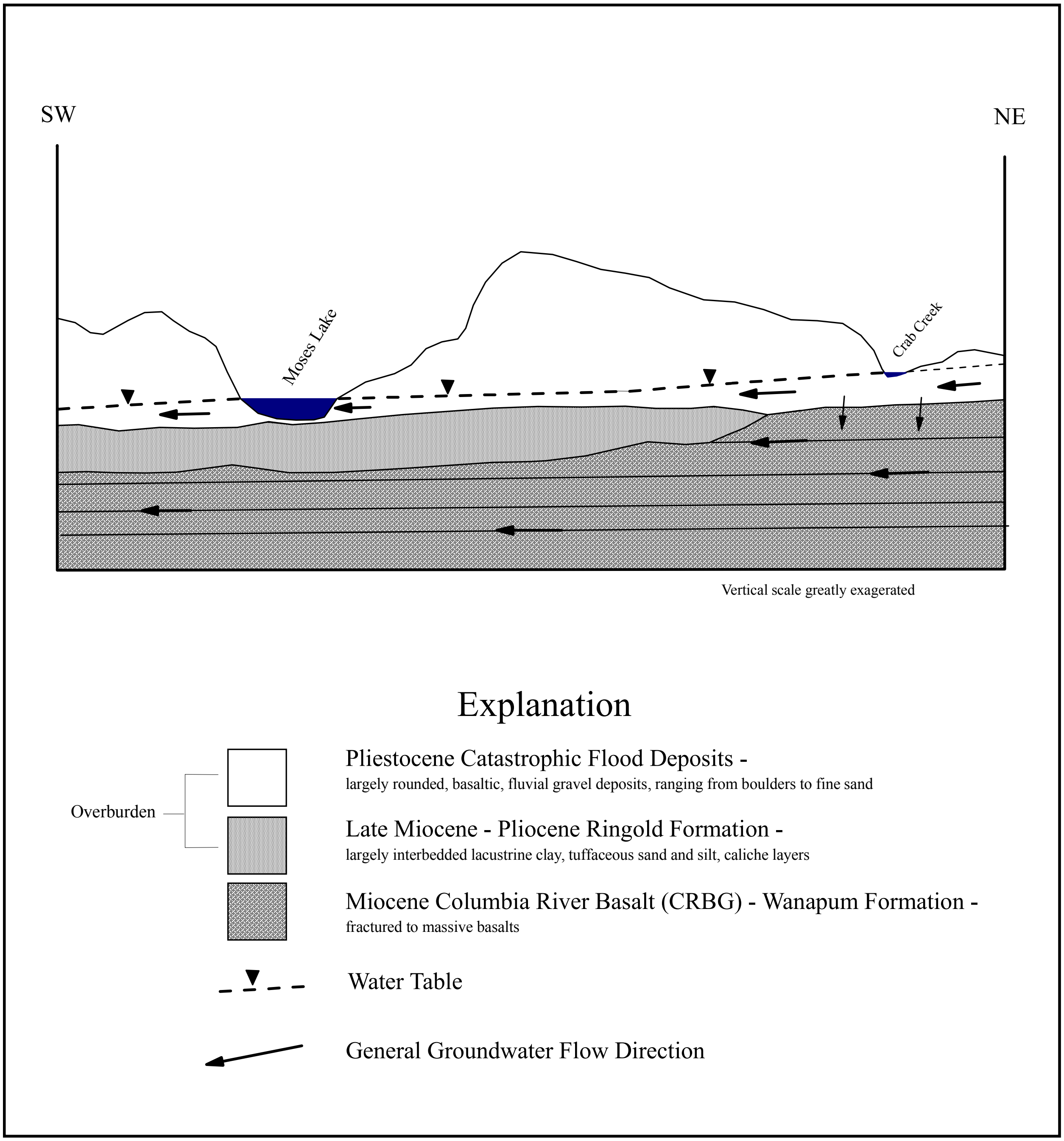


Figure 5 - General Stratigraphy and Conceptual Cross-Section Model of Groundwater Flow

between the airport and the city. Up to 50 feet of fine-grained Ringold deposits have been reported underlying the eastern shoreline of the main arm of the lake (Figure 5)(GAI, 1991; USACE, 2001a; Grolier and Bingham, 1971; Grolier and Foxworthy, 1961).

The geometry and distribution of the Ringold sediments is less well understood for the northern half of the lake. A review of the available well logs from the northern half of the study area indicates that the majority of wells drilled to the basalts did encounter a fine-grained clay or silt unit deposited on the upper basalt surface (interpreted as Ringold sediments). Where encountered, the upper surface of the Ringold sediments normally occurs near or below the lake surface elevation (Figure 5). However, several investigators have reported that the top of the Ringold lies above the lake surface in the area of the big bend of the main arm, southwest of the airport. Previous investigators also have suggested the presence of erosional windows or coarser-grained facies changes in localized areas of the Ringold deposits (Grolier and Foxworthy, 1961; Grolier and Bingham, 1971; GAI, 1991; USACE, 2001a).

Overlying the Ringold sediments are a sequence of Pleistocene-age flood deposits that mantle the ground surface around the majority of the lake (Figure 4). These glaciofluvial deposits (alternatively named the Missoula Flood Deposits, the Hanford Formation, or the Pleistocene gravels by various authors), are largely comprised of massive to well-stratified boulder to granule-sized basaltic gravel, with lesser deposits of sand, silt, and non-basaltic gravel. Caliche fragments and coatings of caliche on gravel surfaces are present in the shallower portions of this unit. The coarse sediments, which often display an open-work texture, were deposited as a result of repeated, high-energy catastrophic floods that occurred with the rapid release of water from glacial-age Lake Missoula in Montana.

Reconnaissance of the shoreline and exposed littoral zone during the course of this project indicated that the typical particle size of the material in the near shore area of the lake is cobble to boulder size. The flood deposit unit extends beyond Crab Creek to the east, although exposures of the basalt are seen through erosional cuts in the creek channel. East of the pinch out of the Ringold Formation, the flood deposits directly overlie the basalts. The flood deposits thicken to the west, and reach an estimated maximum thickness in the vicinity of the eastern shoreline of the lake of approximately 140 feet (Grolier and Foxworthy, 1961; Grolier and Bingham, 1971; GAI, 1991; USACE, 2001a; Gulick, 1990).

Moses Lake occupies the sinuous, lowermost channels of a large-scale meandering drainage system cut into these deposits by floodwaters. Localized Quaternary deposits of eolian, lacustrine, and alluvial sediments have subsequently accumulated within low-lying portions of the study area. Moses Lake itself was formed due to the deposition of dune sands across this channel system, resulting in a natural barrier to flow.

The topography surrounding the lake is defined by a series of relatively flat depositional terraces and cross-cutting, abandoned erosional channels. These terraces step steeply down to the lake shoreline in several areas where cut banks were eroded by floodwaters, exposing thick sequences of the flood gravels along the lake shoreline (Grolier and Foxworthy, 1961).

The area inland of the southeastern shoreline of Pelican Horn exhibits a significant change in the sequence and vertical position of strata adjacent to the lake. In this area, the flood deposits only thinly veneer the older formations, and a thick sequence of underlying lacustrine clays and eolian sands of the Ringold Formation are exposed on bluffs down to the shoreline of the lake (Figure 4)(Grolier and Foxworthy, 1961; Grolier and Bingham, 1971; Gulick, 1990). Upland of the shoreline, the basalts that immediately underlie the Ringold Formation deposits belong to the Priest Rapids Member, which in turn overlie basalts of the Roza Member. The uppermost surface of the basalts upgradient of the lake in this area is approximately 75 feet above the lake surface. The Priest Rapids pinches out towards the shoreline, bringing the Ringold sediments in direct contact with the Roza member basalts beneath the lake.

Surface soils in the Moses Lake area are largely from the Ephrata and Malaga series. These soils are typically characterized by very deep profiles of well-drained to excessively well-drained material formed on glacial flood deposits. The grain size profile with depth is normally characterized by a shallow-horizon gravelly sandy loam (Ephrata) or cobbly sandy loam (Malaga) grading to a deep-horizon extremely gravelly and cobbly coarse sand (USDA, 1984; Bain, 1990).

Soil permeabilities are moderately rapid within the upper horizons, and very rapid in the lowermost portion of the soil profile, reflecting the coarse-grained nature of the underlying parent deposits. Wind-born deposits of loess may be incorporated into the upper soil profile, and calcium carbonate coatings on particles are often present. The percentages of natural organic material and clay particles are typically low in the deeper portions of the soil horizon.

The soils encountered in the emergent wetland at the northernmost sampling station included in this study (near the mouth of Rocky Ford Creek), are distinct from soils encountered elsewhere in the study area. The soil profile at this location is composed of very poorly drained black peat and muck, interpreted to be equivalent to the Saltese Muck soil type mapped in this area by the Soil Conservation Service (USDA, 1984). As discussed later in this report, wetland soils can be a significant natural source of phosphorus.

Groundwater Hydrology

The groundwater hydrology of the Columbia Basin is defined by a complex multi-aquifer system comprised of the CRBG formations and overburden deposits (Whiteman et al., 1994; Bauer and Hansen, 2000). The focus of the current study is on the shallow portions of the flow system that most likely interact with the waters of Moses Lake.

Many regional studies of the Columbia Basin have mapped the overburden sediments and the uppermost flow unit of the CRBGs as a single aquifer. However, groundwater studies in the area of the lake indicate that, where present, finer-grained deposits of the Ringold formation act as an aquitard, hydraulically separating groundwater in the flood deposits from groundwater in the uppermost basalt flows (GAI, 1991; USACE, 2001a; USACE, 2001b; Sinclair, 1999).

The position and distribution of the Ringold sediments with respect to the lake bed geometry suggest that the majority of groundwater that interacts with Moses Lake moves through the unconfined, high permeability flood deposits, with limited direct interaction from the basalt system (Figure 5). Groundwater interacting with the lake along the southeastern shoreline of Pelican Horn (as well as in the area of the big bend), is presumably transported through the finer-grained Ringold deposits present above and adjacent to the lake.

There are no data available to determine if the basalts are in direct contact with the lake in local areas of the lake bottom. Limited shoreline surface exposures of basalt are observed at the northern end of the main arm, as well as the northern end of Pelican Horn, suggesting the basalts directly underlie and hydraulically interact with the lake in these areas.

Descriptions of the hydraulic characteristics of the aquifer materials of the Moses Lake area are largely drawn from the environmental investigations associated with the former Larson air base east of the lake. These investigations have included the installation and monitoring of observation wells, field tests of hydraulic parameters, and ultimately, the development of a three-dimensional numerical groundwater flow model for this area (USACE, 2001a; USACE, 2001b).

The horizontal hydraulic conductivity of the flood deposits in the vicinity of the base reportedly range from 2,800 to 28,000 ft/day, with average seepage velocities of 1,100 ft/day. These are extraordinary values, reflecting the coarse nature and open-framework texture of the deposits. The Ringold sediments in this area have a significantly different hydraulic character, with hydraulic conductivities ranging between 0.02 and 1.7 ft/day, and an estimated average seepage velocity of 0.01 ft/day. Hydraulic gradients in these units range from 0.002 to 0.008 (USACE, 2001a). Model-calibrated horizontal hydraulic conductivities for the uppermost basalt units reportedly range from 15 to 120 ft/day, with vertical conductivities estimated an order of magnitude lower (USACE, 2001b).

Groundwater flow within the basalt system primarily occurs as lateral flow within interflow zones, with limited lateral or vertical movement through the flow interiors. The lower permeability values for the basalts suggest an additional limit to the contribution of groundwater to the lake from deeper portions of the basalt system.

Depth to groundwater over the study area is a function of topography. In low relief areas adjacent to the lake shoreline (e.g. along the peninsula between Parker and Pelican Horns) depth to groundwater is routinely less than 20 feet. Depth to the water table on high bluffs above the lake surface may be 100 feet or more. Due to the extremely coarse character of the flood deposits, infiltration rates through the study area vadose zone are considered to be very rapid, with limited attenuation capacity for pollutants.

To the east, where the Ringold is absent, groundwater in the basalt aquifer is present under unconfined conditions, and interacts directly with the flood deposits. Moving west towards the lake, the groundwater in the basalt is generally confined beneath the Ringold sediments. The vertical hydraulic gradient between the flood deposits and the basalt system is neutral or downward, suggesting the flood deposits locally recharge the basalts (USACE, 2001a). Heavy groundwater withdrawals from both systems (dominantly from the basalt units) have been noted to influence the vertical gradient between the aquifers, particularly in the area between the city and the airport.

Recharge to the local aquifer system originates from a combination of precipitation, infiltration of groundwater and surface water derived irrigation, and groundwater injection. The total annual recharge rate for the irrigated areas east of the lake was estimated for groundwater flow modeling purposes at approximately 22 inches/year; the recharge rate for undeveloped areas adjacent to the lake are significantly lower, typically less than 6 inches/year (USACE, 2001b; Bauer and Hansen, 2000).

Discharge from the aquifer system is primarily from water-supply withdrawals, discharge to local surface waterbodies (including baseflow or spring discharge to Crab and Rocky Ford Creeks), and direct discharge to the lake. The lake has historically been described as a regional discharge feature for shallow groundwater within the Columbia Basin. Published regional potentiometric head maps show contours within the uppermost portions of the basin aquifer system broadly converging towards the lake from the west, north, and east (Mundorff et al., 1952; Bauer et al., 1985; Bauer and Hansen, 2000; Lane, 1988).

Identifying specific areas of the lake where groundwater discharge occurs was important for the purposes of the current study. Characterization of local groundwater flow directions is complicated by a number of factors. These factors include: a) seasonal fluctuations in groundwater levels due to irrigation and aquifer pumping cycles, b) poorly constructed wells that enhance hydraulic communication between aquifers, c) complex heterogeneities in the subsurface geology, d) localized mounding of the water table surface, and e) the dynamic influences of lake surface elevation adjustments. As a result, determining which portions of the lake shoreline are receiving groundwater discharge is difficult to do in an exact manner.

Figures 4 and 5 illustrate the general groundwater flow directions for the aquifer system surrounding the lake, as interpreted from a variety of data sources (Mundorff et al., 1952; Walters and Grolier, 1960; Bauer et al., 1985; Lane, 1988; GAI, 1991; Sinclair, 1998; Sinclair, 1999; USACE, 2001a). Water level data from the U.S. Geological Survey's Ground Water Site Information (GWSI) database were mapped to evaluate the groundwater flow regime specifically in the study area surficial aquifer. The available data, largely collected in the mid-1980s during the Columbia Basin Regional Aquifer System Analysis study, indicate that Moses Lake is, on a regional scale, a flow-through lake. This means that groundwater enters into Moses Lake from the adjoining surficial aquifer system along certain portions of the lake shoreline, but that lake water also is lost to the aquifer in other areas. Groundwater in the deeper basalt system is interpreted to largely flow beneath the lake towards the southwest.

The existing data indicate that groundwater from the unconfined aquifer discharges to the lake primarily along the northwestern and eastern shorelines (Figure 4). An exception along the eastern shoreline may lie in the area of the big bend, where the flood deposits have been reported as unsaturated, and the uppermost surface of the Ringold Formation lies above the lake. Groundwater discharge in this area may be limited due to the lower permeability of the Ringold deposits (USACE, 2001a). Additional groundwater inflow also enters the lake along the southeastern shoreline of Pelican Horn, although the volume of inflow is probably limited by the hydraulic character of the Ringold sediments bounding the lake in this area. Conversely, the data indicate that the lake likely recharges the aquifer along the far southern and southwestern shorelines (Figure 4).

Local gradient reversals such as those caused by irrigation-derived recharge or septic tank releases may cause local scale movement of groundwater towards the lake in an area identified as a regional recharging shoreline. This suggests that some additional loading of nutrients from near-shore sources could occur along the western and southwestern shorelines of the lake.

Where it occurs, groundwater inflow to the lake is assumed to discharge predominantly within the littoral zone via upward and lateral seepage from the unconfined aquifer through the lake bed sediments. The volume of groundwater discharge most likely declines exponentially with distance from the shoreline (Lee, 1976; Winter, 1978; Wagner et al., 1983; McBride and Pfannkuch, 1975; Harvey et al., 2000; Lee et al., 1980). No published information is currently available regarding the character of the lake bottom sediments for Moses Lake. However, the prospect that fine deposits of the Ringold formation underlie large portions of the lake suggests that extensive discharge of groundwater into deeper areas away from shore is unlikely.

It is well established that groundwater seepage to a lake can be a highly localized phenomenon, strongly influenced by small-scale heterogeneities in the subsurface (Lee, 2000). In addition to large changes in seepage rate over short distances, there can be significant temporal fluctuations in discharge. Local irrigation and withdrawal cycles, seasonal leakage from canals and wasteways, and manipulation of the lake surface elevation are probably all locally important influences on groundwater inflow rates.

Background Information

Phosphorus Fate and Transport Controls in Groundwater

In many aquifers the presence of dissolved phosphorus in groundwater is limited due to its strong affinity to bond with sediment particles. Phosphorus is readily sorbed to clay minerals, iron, manganese, and aluminum hydroxides, and calcium carbonate coatings present on the outer surfaces of sediment particles (all of which are common constituents in geologic environments).

Sorption reactions transfer phosphorus from solution to the solid phase, lowering the dissolved concentration in groundwater. Phosphorus is preferentially sorbed by finer-grained sediments, partly due to the larger overall surface area (and therefore greater number of sorption sites) on fine particles in contrast to coarse deposits (Zanini et al., 1998; Zilkey et al., 2001). The higher flow velocities of groundwater through coarser-grained deposits also limit the contact time available for sorption reactions to occur.

Phosphorus also can combine with a variety of common cations (iron, manganese, aluminum, and calcium) to precipitate secondary phosphate minerals. Similar to sorption processes, precipitation reactions remove dissolved phosphorus from solution, lowering the groundwater concentration. The concentration of phosphorus in groundwater in contact with sediments containing phosphate minerals is determined by the mineral solubility. Phosphate mineral form and solubility are a function of the prevailing geochemical conditions in the aquifer, including the pH, redox potential, temperature, and ionic strength of the solution phase (Ptacek, 1998, Stollenwerk, 2002; Zanini et al., 1998).

Phosphorus sorption and precipitation reactions in an aquifer are both controlled in part by the prevailing groundwater/soil pH. Sorption capacities for phosphate are lower at neutral or slightly alkaline pH conditions (Robertson et al., 1998; Stollenwerk, 1996; Walter et al., 1995, Stollenwerk, 2002). Zanini et al., 1998 concluded that phosphate concentrations in groundwater are likely to be highest in coarse-grained calcareous sediments, and lowest in fine-grained non-calcareous deposits. This is due in part to the pH buffering by calcium carbonate that prevents the development of acidic conditions that favor phosphate sorption and precipitation (Robertson, 2003).

Sorption and precipitation reactions and capacities for immobilizing phosphorus also are influenced by the prevailing redox conditions. For example, under reducing conditions the metal hydroxides (e.g. iron and manganese) that often serve as sorption sites for phosphorus are dissolved. Dissolution of hydroxide coatings and particles limits the number of available sorption sites, and results in the release of accompanying phosphorus to groundwater. This release results in an increase in the dissolved phosphorus concentration in the aquifer. If there is a change from reducing to oxidizing conditions, dissolved iron and manganese precipitate out of solution, resulting in the rapid sorption and immobilization of the phosphorus (Robertson et al., 1998; Walter et al., 1995; Zilkey et al, 2001; Zanini et al., 1998; Ptacek, 1998; Vanek, 1991; Carlyle and Hill, 2001).

Orthophosphate (OP) is the dominant species of dissolved phosphorus in groundwater (Hem, 1989). Elevated concentrations of dissolved phosphorus in groundwater can be due to an exhaustion of the attenuative capacity of an aquifer by anthropogenic (human-caused) loading, geochemical conditions favoring desorption or dissolution of phosphorus (also frequently caused by anthropogenic impacts), or both.

The creation of a reservoir of sorbed (or precipitated) phosphorus in the subsurface beneath areas that have received a long-term load from an anthropogenic source (e.g. a septic system) may cause the chronic release of the nutrient to groundwater even after the cessation of loading. The long-term concentration of phosphorus in groundwater under this scenario may largely be determined by the phosphate mineral solubility. A sharp increase in the dissolved concentration also may occur if there is a change to the geochemical conditions that favor phosphate immobilization (e.g. a change in pH)(Walter et al., 1995; Robertson and Harman, 1999).

Due to its geochemical characteristics, phosphorus transport distances are often assumed to be relatively limited in the subsurface. However, an increasing number of case studies have demonstrated that phosphorus can, under the right conditions, be carried long distances in groundwater, and can directly impact the trophic status of surface waterbodies distant from a source (Kelly et al., 1999; Robertson et al., 1998; Robertson and Harmon, 1999; Harmon et al., 1996; Zilkey et al., 2001; Ptacek, 1998; Carlyle and Hill, 2001). Walter et al. (1995) noted transport of dissolved OP in a wastewater plume at concentrations above 0.050 mg/L over 3000 feet from the source area in a sand and gravel aquifer, demonstrating the capacity for significant lateral transport of phosphorus in groundwater. These prior studies have shown that the mobilization of phosphorus is often the result of a change in redox or pH conditions in the aquifer by the subsurface release of organic-rich wastewater.

Natural Condition of Phosphorus in Area Groundwater

It has been suggested that the elevated phosphorus concentrations observed in the groundwater of the Moses Lake area naturally originate from dissolution of surface or subsurface geologic deposits containing phosphate minerals. As discussed above, under normal aquifer conditions, the concentration of phosphorus present in the dissolved phase is limited, even when a mineralogic phosphate source is present.

Limited occurrence of apatite, a common, low-solubility phosphate accessory mineral in basalt, has been confirmed in Ringold sediments in the central Columbia Basin (Gaylord, 2002; Horton, 2001). There are no references to the presence of significant phosphate mineral deposits in the Ringold or flood sediments in the published geologic study reports for the Moses Lake area, although detailed mineralogical descriptions of these formations are limited in the literature (Grolier and Foxworthy, 1961; Walters and Grolier, 1960; Grolier and Bingham, 1971).

The available regional groundwater quality data do not indicate an area-wide pattern of elevated phosphate in groundwater, as might be expected if there were significant mineralogic dissolution of phosphorus in the aquifer system. For example, Jones and Wagner (1995) reported a median OP concentration of 0.020 mg/L as P for samples collected between 1942 and 1992 from 81 wells located throughout the central Columbia Plateau. While their data showed no clear trend in OP concentration with depth as might be expected from a buried geologic source, the highest concentrations reported were from the shallowest wells, suggesting a surface or near surface source.

To better understand the historic condition of phosphorus in the shallow aquifer system in the vicinity of Moses Lake specifically, a search of the U.S. Geological Survey (USGS) National Water Information System (NWIS) water quality database was conducted. In the Moses Lake area, the average total dissolved phosphorus (TDP) and dissolved OP concentrations reported for wells sampled since 1980 that were less than or equal to 150 feet deep, were 0.037 and 0.035 mg/L as P, respectively (n=9). No values above 0.050 mg/L were reported for either parameter.

These data were compared to the values reported by Bain (1987a, 1987b), who conducted groundwater and spring sampling in the fertilized agricultural areas adjacent to Crab Creek to compare to samples collected from urbanized areas. Bain reported finding soluble reactive phosphorus (SRP – equivalent to OP in this report) values normally ≤0.05 mg/L. While it is possible that the slightly higher values reported in the Moses Lake area (versus the central Columbia Basin in general) reflect a greater localized natural mineral contribution, the evidence assembled during this study suggests anthropogenic impacts from upgradient sources play an important role in raising the ambient groundwater concentration.

There are case studies that have demonstrated that naturally-derived, groundwater-borne phosphorus can adversely impact surface water. A regional study that has similarities to the Moses Lake area was conducted by Kelly et al, (1999). These authors described a significant dissolved OP contribution from an aquifer system to the Tualatin River in Oregon. They concluded that the OP in the groundwater inflow to the river results from the dissolution of a natural phosphate mineral source present in basin sediments buried far below the river (see also Wilson et al., 1999).

The phosphate in the Tualatin River basin originates in the Neogene-aged, fine-grained sediments of the Hillsboro Formation (Wilson et al., 1999). The Hillsboro Formation is composed of a thick sequence of fluvial basin-fill sediments. This unit directly overlies basalt bedrock of the CRBG and is equivalent stratigraphically to the Ringold Formation underlying Moses Lake. The Hillsboro sediments are thought to originate from erosion of the phosphorus-rich Paleogene marine sedimentary units of the Oregon Coastal Range (Burns, 2002; Wilson et al., 1999). In contrast, the quartzo-feldspathic sediments of the Ringold Formation underlying Moses Lake are thought to originate largely from metasedimentary, granitic, and volcanic terrane located northeast of the Columbia basin. These settings are unlikely to provide a source of phosphorus-rich sediments like that in the Tualatin basin (Gaylord, 2002; Lindsey, 2002).

The high dissolved phosphate concentrations found throughout the groundwater system of the Tualatin River basin (up to 2.5 mg/L) are present because of the naturally reducing conditions that occur in the basin aquifer system. These conditions are thought to be caused by the decomposition of the extensive deposits of organic material found at the contact between the Hillsboro Formation and the overlying silts of the Missoula flood deposits (the Willamette Silt)(Kelly et al., 1999; Wilson et al., 1999). The reducing conditions have resulted in elevated concentrations of ammonia throughout the Tualatin aquifer system, with only limited presence of nitrate (typically less then 0.1 mg/L).

Review of existing literature and well logs from the Moses Lake area did not reveal evidence of widespread organic material between the Ringold deposits and the overlying catastrophic flood gravels (Grolier and Foxworthy, 1961; Walters and Grolier, 1960; Grolier and Bingham, 1971; GAI, 1991; USACE, 2001a). In addition, ammonia is only detected locally in the Moses Lake area, while the wide presence of nitrate (Williamson et al., 1998; Ryker and Frans, 2000) indicates a regionally oxidizing groundwater environment.

In contrast to the regional OP data assembled by Jones and Wagner (1995) for the central Columbia Basin, the Tualatin data show a clear increase in OP groundwater concentration with depth, the expected data distribution for a buried source (Kelly et al., 1999). Therefore, while there are stratigraphic parallels between the Tualatin basin and the Moses Lake area, the existing evidence does not indicate that the study area geologic setting acts as a significant source of phosphorus to groundwater.

Potential Anthropogenic Sources of Nutrients in Area Groundwater

Nitrogen

Characterization and source analysis of nitrogen concentrations in groundwater of the central Columbia Basin were areas of active investigation over the past decade. Ryker and Frans (2000) summarized results for a large-scale sampling effort of wells throughout the Grant, Franklin and Adams County area. For their study, groundwater was analyzed for nitrate plus nitrite as nitrogen, and reported as nitrate as N (nitrate-N, due to the fact that nitrite concentrations in groundwater of the basin are routinely below detection).

The median nitrate-N concentration from the Moses Lake sub-region of Grant County (the area lying immediately east of the lake) was reported as 3.8 mg/L for wells less than 200 feet deep. Seventeen percent (17%) of the shallow wells tested in the sub-region had nitrate-N concentrations above 10 mg/L, the value identified as the U.S. Environmental Protection Agency’s maximum contaminant level (MCL) in drinking water.

Groundwater nitrate-N concentrations in the central Columbia Basin above 3 mg/L are normally interpreted as an indication of anthropogenic impact (Ryker and Frans, 2000). Statistical studies of the nitrate data for the basin show that land use practices strongly correlate to the concentration and distribution of nitrate in the aquifer system. Specifically, irrigated agriculture (and the associated downward migration of nitrogen fertilizer) has been identified as the best explanatory factor correlated to elevated nitrate-N concentrations in basin groundwater (Frans, 2000). Nitrate-N concentrations above 10 mg/L also were positively correlated to the degree of urban development overlying the aquifer system.

Nitrogen-bearing fertilizer is commonly applied to irrigated land in the area surrounding Moses Lake. In 1990, Bain reported that nitrogen application rates averaged approximately 161 lbs/acre in the irrigated farmland areas east and northeast of the lake. Due to the high permeability of the surface soils and underlying sediments, Bain noted concerns regarding the potential for downward migration of nitrogen to the water table, followed by lateral transport to the lake (Bain, 1990; Bain, 1987). Bain concluded that deep percolation loss of nitrogen to groundwater from irrigated cropland represented a significant percentage of the groundwater nitrogen load to Moses Lake.

As discussed in the section below, infiltration of urban wastewater in the Moses Lake area also represents a significant potential source of nutrients to the shallow aquifer system. Urban wastewater typically contains elevated concentrations of organic nitrogen and ammonia, which are quickly converted to nitrate in most wastewater treatment systems. Dissolved nitrate can rapidly leach through the soil column, resulting in elevated concentrations in the underlying groundwater system.

Phosphorus

There are a variety of potential anthropogenic sources of phosphorus loading to groundwater, both on a local and regional scale. These sources include:

- releases from on-site septic systems
- infiltration from waste water treatment plant ponds and land application sites
- leakage from buried sewer pipes in sewered areas (particularly those at or below the water table)
- contaminated surface runoff draining to drywells
- lawn and garden fertilization (including sports fields, parks, and golf courses)
- pet waste
- use of phosphate detergents (particularly prior to 1994)
- infiltration from feed lots and dairies
- large scale application, and subsequent downward migration, of phosphorus fertilizer to agricultural crops

Household wastewater is managed by the municipal sewage utility for a large majority of the population within the city limits of Moses Lake. A limited number of neighborhoods and individual households located within the city limits continue to be served by on-site septic systems (O’Brien, 2002; Henning, 2002; Bain, 1986).

The municipal system collects and transfers wastewater from sewered areas of the city via buried lines, and conveys that water to the Central Operations Facility on the northwestern shore of Pelican Horn. After pretreatment, the water is pumped south to the Moses Lake Dunes wastewater treatment plant (WWTP) (Figure 6). Treated wastewater from the plant is discharged to a series of rapid infiltration basins under the terms of a State Waste Discharge permit issued by Ecology.

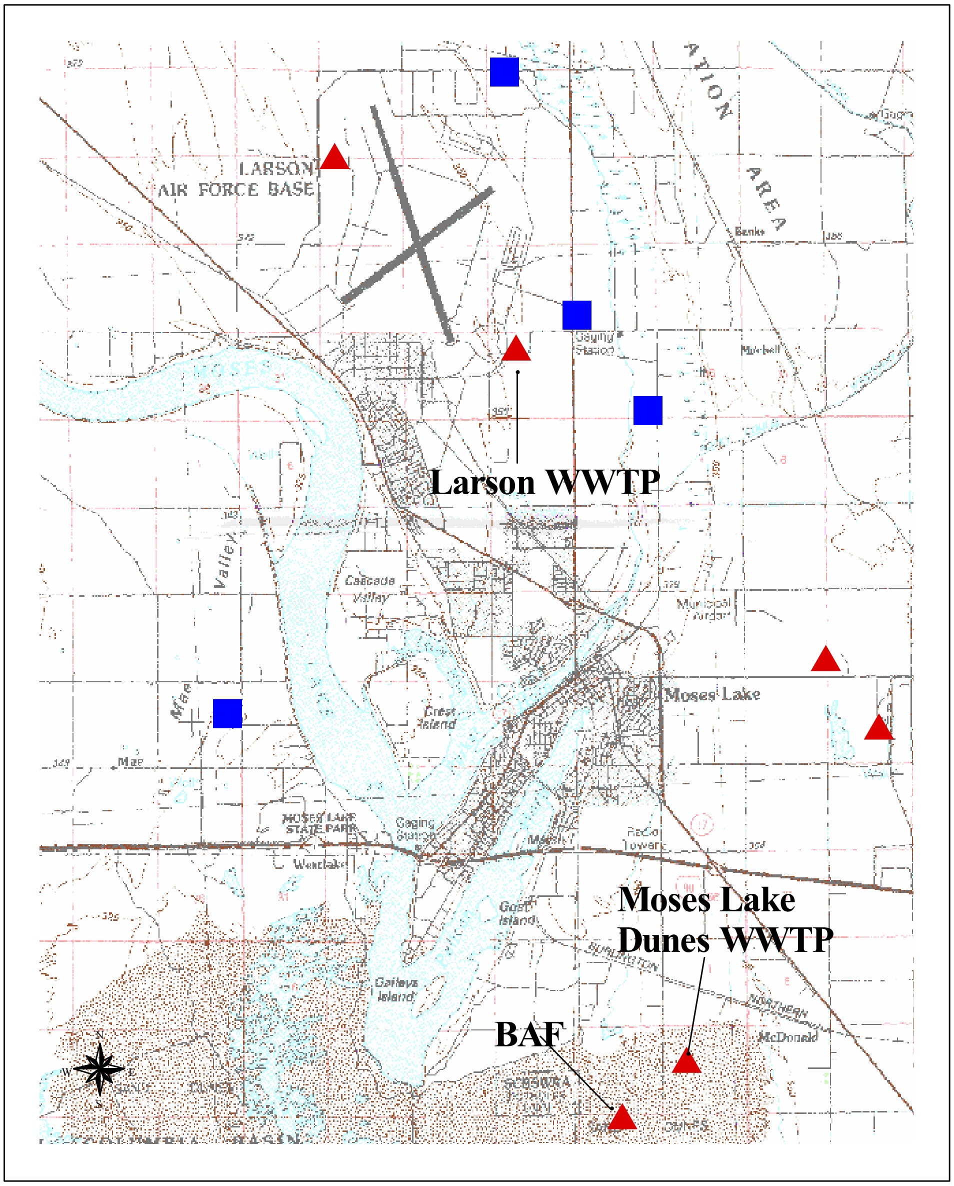


Figure 6 - Potential Point Sources of Phosphorus to Shallow Groundwater

Historic data of the effluent quality discharged from the facility indicate that the plant is a significant source of phosphorus load to the subsurface, and groundwater samples collected from monitoring wells beneath and downgradient of the facility have shown elevated concentrations of TP (0.03 – 2.15 mg/L)(Sinclair, 1999). However, the groundwater flow direction beneath the discharge site is predominantly towards the southwest, away from Moses Lake.

Wastewater from residences located in the unincorporated areas surrounding the city (representing approximately half of the area urban population) is predominantly released to small on-site septic systems (Ness, 2002; Carroll et al., 2000). Phosphate is routinely observed in septic system effluent at concentrations between 3 and 20 mg/L. While significant reductions in phosphate concentrations are commonly observed short distances below and downgradient of septic system drain fields due to sorption and precipitation reactions, groundwater concentrations often remain above levels necessary to impact adjacent surface water trophic conditions.

The attenuation of phosphorus normally observed in soils beneath waste discharge facilities can be inhibited under certain circumstances. One of the factors hampering phosphorus removal by soil beneath septic drain fields includes short effluent/soil contact times due to high soil permeability or high water table. A second important factor is the presence of reducing conditions, often brought about by the depletion of oxygen during biological breakdown of the organics present in septic effluent (Robertson and Harmon, 1999; Zanini et al., 1998; Walters et al., 1995; Ptacek, 1998; Robertson et al., 1998; Vanek, 1991; Harmon et al., 1996; Gilliom and Patmont, 1982; Wagner et al., 1983; Cogger, 2002).

Bain (1986) described a groundwater sampling effort to determine the effect of on-site septic systems on the quality of urban groundwater beneath the developed residential areas of Moses Lake. For his study, Bain installed and sampled nine shallow wells for TP. While Bain did not follow standard sampling protocols for obtaining representative groundwater quality data (e.g. no well purge or sample filtration was reported), the data he collected confirmed that phosphorus levels beneath un-sewered areas of the city were elevated significantly above concentrations he reported for rural areas in the Crab Creek drainage.

The values reported for urban area TP (0.80-10.5 mg/L, average 1.5 mg/L) were attributed by Bain to a combination of contaminated leachate from on-site septic systems, and large-scale discharge of treated effluent to ground. Since his samples were not filtered, Bain’s results are probably unrealistically high, but do serve as a relative indicator of phosphorus presence in the aquifer under or downgradient of areas served by septic systems.

Approximately 5000 residents living in the Larson area are served by the Larson WWTP, operated by the City of Moses Lake (O’Brien, 2002). The Larson plant, originally established in 1973, also operates under the terms of a State Waste Discharge permit issued by Ecology (Figure 6). Approximately 350,000 to 400,000 gallons of treated wastewater are released on a daily basis from this facility to the high permeability flood deposits via rapid infiltration ponds (USACE, 2001a).

Groundwater quality data were evaluated from water table monitoring wells installed adjacent to the rapid infiltration ponds at the Larson facility. The data are drawn from routine monitoring conducted at the site under the terms of the discharge permit. Between January 1999 and June 2001, the monthly average for TP in the groundwater monitoring well downgradient of the ponds was approximately 1.46 mg/L, with maximum values up to 3.5 mg/L. In contrast, the average TP value reported during the same period for the upgradient well was approximately 0.080 mg/L (Peterson, 2002). Onsite groundwater flow directions are largely to the south, although recent groundwater flow modeling studies predict water from the facility would ultimately move to the southwest to discharge along the eastern shoreline of the main arm, adjacent to the Cascade Valley area (USACE, 2001b)(Figure 6).

These data indicate that the Larson WWTP discharge is a probable long-term source of loading of phosphorus to the aquifer system upgradient of the lake. During his study of impacts of local urban wastewater sources on area groundwater, Bain concluded that the direct infiltration of treated wastewater into the high permeability soils below the Larson facility could potentially load as much as 4500 kg/yr of phosphorus to the lake (Bain, 1986). Bain's estimates were described as worst case due to the assumption that there was no additional retention of the phosphorus between the point of release to the vadose zone and the point of discharge at the lake.

In addition to the municipal treatment plants, Figure 6 presents the location of the facilities in the Moses Lake area that are permitted by Ecology to discharge-to-ground treated water that may contain phosphorus (typically via land application of treated effluent). One of these facilities, Basic American Foods (BAF), lies immediately southwest of the Moses Lake Dunes site. Process wastewater from the BAF facility is land-applied to cropland. Historic data indicate the BAF effluent is a significant source of phosphorus load to the subsurface, and groundwater beneath the land application fields is elevated in OP. However, similar to conditions observed at the Moses Lake Dunes site, the available data indicate the groundwater flow direction beneath the BAF land application fields is predominantly towards the southwest, away from the lake (Sinclair, 1998).

The locations of the known dairy operations within the study area also are presented on Figure 6. Manure management at these facilities may result in the movement of phosphorus and other nutrients to the subsurface. Due to the dynamic nature of the area groundwater flow regime, and the distance of these point sources from the lake, it is unknown if nutrients released by any of these facilities ultimately migrate to the lake.

The non-point application, and subsequent downward migration, of phosphorus fertilizer from agricultural lands is another significant potential source of nutrients to the regional aquifer system surrounding the lake. As discussed above, deep infiltration of irrigation water from fertilized cropland has had a significant effect on the dynamics and nutrient concentrations of the aquifer system of the Columbia Basin in general, as well as the Moses Lake area specifically. Large scale data sets are not available to determine if there are correlations between phosphorus in groundwater and irrigated agriculture in the basin. Such correlations would be complicated by the non-conservative transport of phosphorus in the groundwater environment, in contrast to nitrate.

Phosphorus-bearing fertilizer is commonly applied to irrigated land in the area surrounding Moses Lake. The high permeability nature of the surface soils and underlying sediments creates a potential for downward migration of phosphorus to the water table, followed by lateral transport to the lake. Grant County agricultural lands have historically had the highest annual phosphorus application rate of the counties in the central Columbia Plateau (Jones and Wagner, 1995). In 1990 Bain reported that annual phosphorus application rates averaged approximately 66 lbs/acre in the irrigated farmland areas east and northeast of the lake.

While a number of efforts have been implemented over the last decade to reduce the amount of nutrient loss to groundwater from irrigated fields, the high permeability and low attenuation capacity of the sediments in the greater Moses Lake area suggest irrigated agriculture may play a role in elevating the ambient condition of phosphate in groundwater above natural background. Irrigated agricultural land is present in close proximity to the lake along the western shore of the main arm, the southeastern shore of Pelican Horn, the eastern shore of the main arm by Cascade Valley, and the northern shore of the main arm east of the Larson area. Samples collected in these areas are the most likely to show an impact from agricultural land use, depending on geologic conditions.

Historic Estimates of Groundwater Loading to Moses Lake

Previous studies of Moses Lake have shown that groundwater is an important component of the lake's water and phosphorus budgets. However, the accuracy of the estimates of the relative contribution of groundwater remains in question. Estimates of the phosphorus contribution have historically been determined by calculating the product of the estimated volume of groundwater inflow, and the assumed average concentration of SRP in the aquifer (Patmont, 1980; Carlson, 1983; Jones, 1988).

Previous estimates of groundwater volume flux into Moses Lake were calculated using a water budget approach. Because Moses Lake is a flow-through lake from a groundwater standpoint, water budget calculations only provide an estimate of the *net* groundwater inflow to the lake, not the gross inflow value. Failure to account for the flow-through nature of the lake may introduce significant error into volume estimates. This error can be compounded when estimating nutrient flux. The phosphorus concentration of groundwater discharging to surface waterbodies is often an order of magnitude higher then that seen in the surface water itself, but the outflow concentration from a lake to an adjoining aquifer is commonly assumed to be the same as the lake concentration (for example Vanek, 1991; TPCHD, 1996).

Historic estimates of the groundwater inflow contribution to the total annual inflow to the lake range from 20 to 45%, primarily based on water balance calculations (Carlson, 1983; Carroll et al., 2000). Estimates of groundwater contributions to the total annual external phosphorus load have ranged between 5 and 30% (Carroll et al., 2000; Jones, 1988).

Previous estimates of groundwater inflow assumed that the majority of the discharge (so in turn, the majority of the subsurface phosphorus loading) occurs to Pelican Horn, particularly along the southeastern shoreline. For example, Jones (1988) concluded that as much as 80% of the groundwater inflow to the lake enters this arm. Jones reached this conclusion on the basis of two main lines of evidence: a sodium mass balance comparison between surface water and groundwater, and an examination of regional potentiometric head maps.

The groundwater component of Jones's mass balance calculations was based on the average of the sample results from one spring discharging to the lower southeastern shoreline of Pelican Horn, an area of comparatively undeveloped land use. Groundwater samples impacted by urban and residential land use typically exhibit sodium concentrations elevated above background. Considering the level of development adjacent to much of Pelican Horn, the spring sample may not have been an adequate representative of local groundwater discharge conditions for the mass balance calculations.

Jones' review of area head maps was based on the concept that the groundwater inflow to the lake is proportional to the hydraulic gradient. An important error in Jones' method was the assumption that the hydraulic conductivity of the aquifer system is the same around all portions of the lake's shoreline. As discussed earlier, while the surface exposures are geologically similar around the lake perimeter, the vertical position and hydrostratigraphy of the units in direct hydraulic communication with the southeast shoreline of Pelican Horn (Ringold) is different than to the north (flood deposits). An evaluation of the character of the sediments adjacent to this portion of the lake indicates that while the reported water table position may be higher then elsewhere (suggesting a higher gradient to the lake), the horizontal conductivity of the aquifer system is likely markedly lower in this area. Failure to account for the change in hydraulic character in this area could result in an overestimate of flux to Pelican Horn.

In contrast to Jones' assumptions, Carroll et al., 2000 reported that several earlier investigators had concluded that the principal source of groundwater inflow to the lake is along the eastern shoreline of the main arm. The geologic and hydrologic data discussed earlier support this interpretation.

The assumptions used for the concentration of the groundwater inflow for developing loading estimates have varied. For example, Patmont (1980) assumed an average SRP groundwater discharge concentration of 0.033 mg/L. Carlson (1983) assumed a value of 0.019 mg/L for nutrient budget modeling specifically for groundwater discharge to Pelican Horn, on the basis of water quality sample data from the Pelican Horn spring mentioned above. For his lake-wide calculations, Jones (1988) assumed a value of 0.051 mg/L for SRP for groundwater entering the lake (a value approximately equal to the proposed TMDL TP criteria). This assumption was based, again, on analysis results from the Pelican Horn spring station.

Methods

Monitoring Network

A network of 12 lake-bed piezometers was installed and sampled in the littoral zone adjacent to the lake shoreline to characterize the nutrient concentration range and distribution in direct groundwater discharge to the lake. Where possible, stations were distributed along portions of the shoreline that were identified by the study area hydrologic analysis as probable regional discharge faces.

To identify potential sampling locations for this study, a boat-based reconnaissance survey was conducted during March of 2001, while the lake was drawn down. Observations of the littoral zone sediment character were noted in order to identify locations where piezometers could be installed after the lake elevation was raised.

In addition to the piezometers, one shoreline seep, one shoreline spring pipe, and one domestic well were included in the study network to provide data in areas where piezometers could not be installed due to the large size of the littoral zone sediments. For the same reason, some areas are not well represented by the sampling network, most notably the far northeastern and lower eastern shoreline of the main arm, and the northwestern shoreline of the lower peninsula. Due to the difficulty of installing piezometers in cobble and boulder sized sediments, the piezometer locations selected during the survey are probably biased towards areas that are finer-grained than the majority of the shoreline.

Figure 7 shows the locations and station names of the sampling sites used for this study. Location information for each of the sampling stations is presented in Table B-1 in Appendix B, including construction information for the piezometers, and brief descriptions of land-use patterns upgradient of each station. At one station, the original piezometer (MLGW-04D – 5 feet deep) was supplemented during the second sampling round with an additional shallower piezometer (MLGW-04S – 3 feet deep) to allow investigation of changes in water chemistry with depth. Between the second and third sampling events, the piezometer at location MLGW-01 was destroyed by wildlife.

Piezometers were constructed and installed using methods modified from Lee and Harvey (1996), Harvey et al. (2000), and Welch and Lee (1989). To construct the piezometers, a steel drive point was securely attached to a 10 to 12 foot length of ¼" I.D. rigid, translucent polyethylene tubing via a barbed fitting. The lowest one foot of the tubing above the drive point was perforated to allow pore water entry. The perforated section of the tubing was screened using an inert polyester mesh filter fabric with 50-micron openings. The filter fabric was wrapped around the tubing, and secured in place using hot glue and orthodontics bands.

Piezometers were installed at the chosen locations in the lake bed between April 30-May 4, 2001, several weeks after the lake water level had been raised. To install the piezometers, a 5/8" I.D., ¾" O.D. hollow steel drive pipe was slipped over the tubing, and seated against the

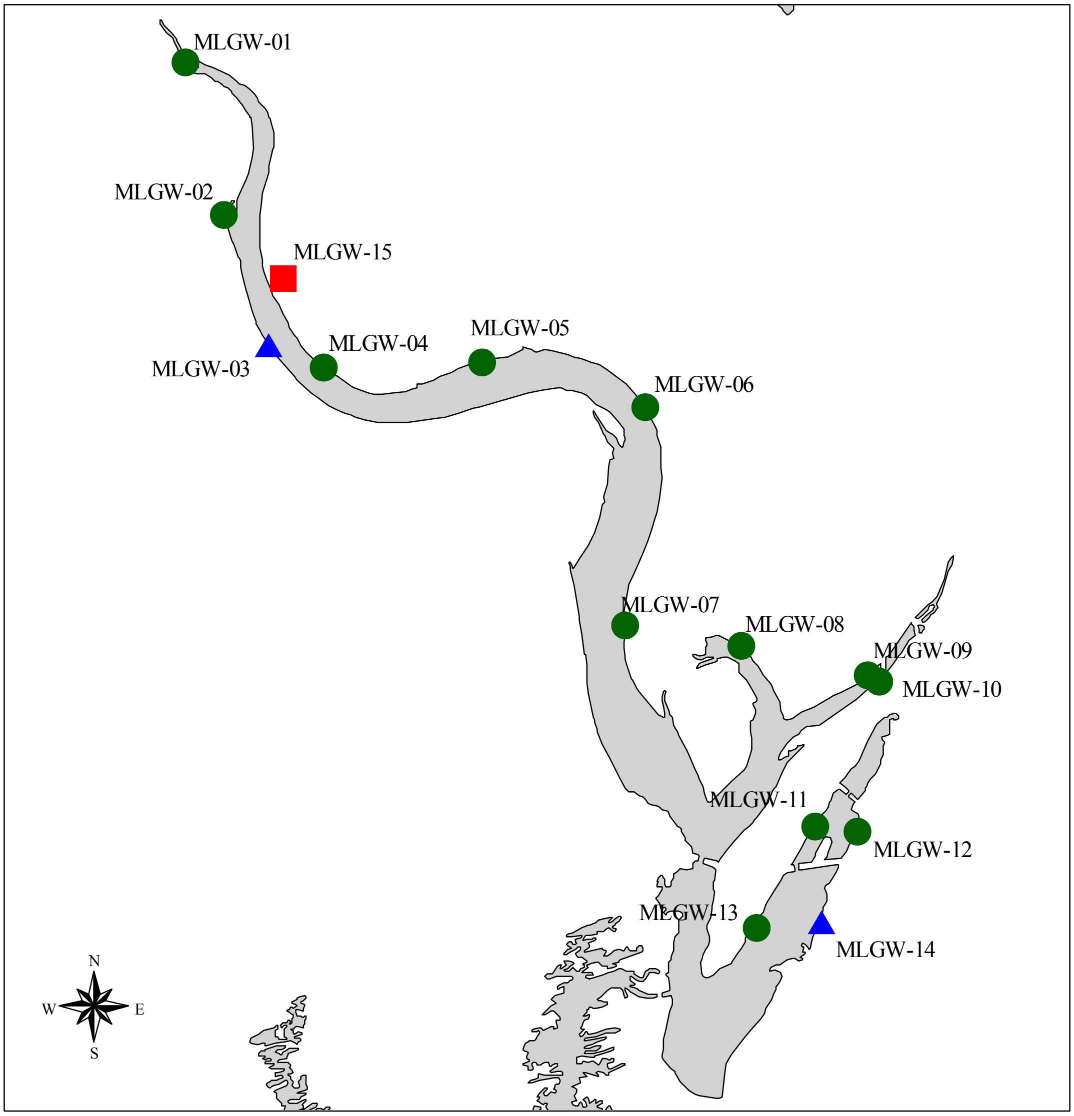


Figure 7 -
Sampling Station Location Map

shoulder of the drive point. The pipe and tubing assembly was then driven to a depth of 3 and 6 feet below the sediment/water interface, using a tripod and drop hammer. Piezometers were normally installed between 10 and 40 feet from the lake shoreline, in water up to 4.5 feet deep.

Once the piezometer was installed to the appropriate depth, the drive pipe was back-hammered and withdrawn, leaving the piezometer in place. As the drive pipe was withdrawn, the formation sediments collapsed against the tubing. The tubing extending above the sediment/water interface was cut to a suitable length for sampling, with several feet of tubing extending above the lake water surface. The sediment adjacent to the tubing was tamped to improve the seal at the sediment/water interface. Between site visits, the upper end of the piezometer was wrapped securely with filter fabric, and the tubing extending above the sediment/water interface was weighted and laid out of sight on the lake bottom.

Upon installation, each of the tubing piezometers was attached to a peristaltic pump, and developed using a surged pumping action. Development continued until the purge water was visually free of turbidity through several surge cycles. All piezometers were allowed to equilibrate after development for one week prior to sampling.

The domestic well sampled for this study (MLGW-15) lies approximately 900 feet from the upper northeastern shoreline of the lake. The well was drilled to a total depth of 140 feet below ground surface, and based on the well log is cased through 128 feet of unconsolidated sand, gravel, and cobbles. The remainder of the well is uncased and extends into fractured basalt bedrock. Unlike most wells in the area, no evidence of fine-grained Ringold deposits was recorded on the well log. The static water level at the time of construction was approximately 95 feet below ground surface. Accounting for the elevation difference between the land surface at the well and the lake surface, the water level in the well is interpreted to represent the water table adjacent to the lake.

The groundwater in the vicinity of the well is presumed to be in direct hydraulic connection with the lake. While the well does not provide as direct a measurement of the groundwater discharge to the lake as the piezometers, water quality data from the well are considered to be reasonably representative of conditions for the area. The potential sources of nutrient loading to the groundwater system between the well and the shoreline are limited (2-3 homes served by on-site septic systems), and the character of the materials reported on the well log suggests the attenuation capacity of the intervening aquifer is likely to be minimal.

Sampling and Analysis Procedures

The sampling network described above was sampled on three occasions (May 7-11, approximately 1 month after the rise in lake surface elevation; July 23-27; and October 1-5 of 2001) to characterize seasonal changes in groundwater quality. Lake freeze over, and the manipulation of the lake surface elevation by the USBR prevented additional sampling between late October and mid-April.

Prior to sampling each piezometer, the tubing was lifted to a vertical position, and the potential head of the pore-water at the screened interval was measured against the lake's water surface (after Welch and Lee, 1989)(Figure 8). This comparison provided an estimate of the relative direction and magnitude of the hydraulic potential between the aquifer material and the lake. If the water level in the piezometer is above the lake surface the vertical hydraulic gradient between the aquifer and the lake is upward and indicates groundwater is discharging to the lake. A water level in the piezometer below the lake indicates downward loss of lake water to the aquifer. A short section of tubing of the same diameter as the piezometer was used as a stilling well to eliminate the effect of wave action on the lake surface measurement. The stilling tube was sealed at the bottom, and a pin hole was made at the lower end to allow gradual water entry. A metric scale was used in the field to record the difference between the equilibrated water levels.

Once the head measurements were recorded, a peristaltic pump was used to collect lake water via a length of flexible medical-grade silicon tubing. The intake was placed immediately adjacent to the piezometer at the sediment/water interface. The discharge line from the pump was routed into a flow cell sealed from atmospheric conditions. The interior of the flow cell was metered with appropriate probes for measurement of field parameters (temperature, conductivity, pH, and dissolved oxygen). The field meters and probes were calibrated daily against known standards, as described in the project quality assurance plan (Pitz, 2001). Dissolved oxygen was not recorded for the May sampling round due to equipment failure.

After recording the lake water field parameters adjacent to the piezometer, the pump intake was connected directly to the top end of the piezometer, and purging was initiated. Purge pumping rates were kept at <0.5 L/min to reduce the chance of inducing downward annular leakage of lake water into the piezometer intake. Field parameters were monitored at five minute intervals during purging until all parameters had stabilized (changes of less than 10% between two consecutive measurements). The field measurements for each piezometer (and adjacent lake water) are presented by sample round in Table C-1 of Appendix C.

The piezometer field parameters typically matched lake water conditions at the beginning of purging, and then progressively diverged from the lake water. The conductivity of the groundwater was routinely higher than the lake at the end of the purging period, and the temperature, pH, and dissolved oxygen were routinely lower. Ongoing comparison of the measured lake values to those from the piezometer served as a field quality assurance check against annular leakage of lake water into the piezometer intake. Figure C-1 in Appendix C presents radar charts of the field parameter profile comparison between the lake water and the final piezometer measurements, for each station by sampling round. Upon the completion of purging, sample collection was initiated. Sample pumping rates were maintained at less then 0.5 L/min.

Piezometer sampling procedures for the May event proceeded as follows. A syringe and dedicated 0.45 micron syringe filter were triple rinsed with formation water pumped from the piezometer. The syringe was then filled, and the first 50-60 mL of filtrate was discarded prior to collecting a filtered OP sample. After collection of the OP sample, the remaining analytes were collected without filtration other than that provided by the filter fabric on the piezometer intake. Additional analytes for the May round included TP, nitrate+nitrite as N (reported as nitrate-N in

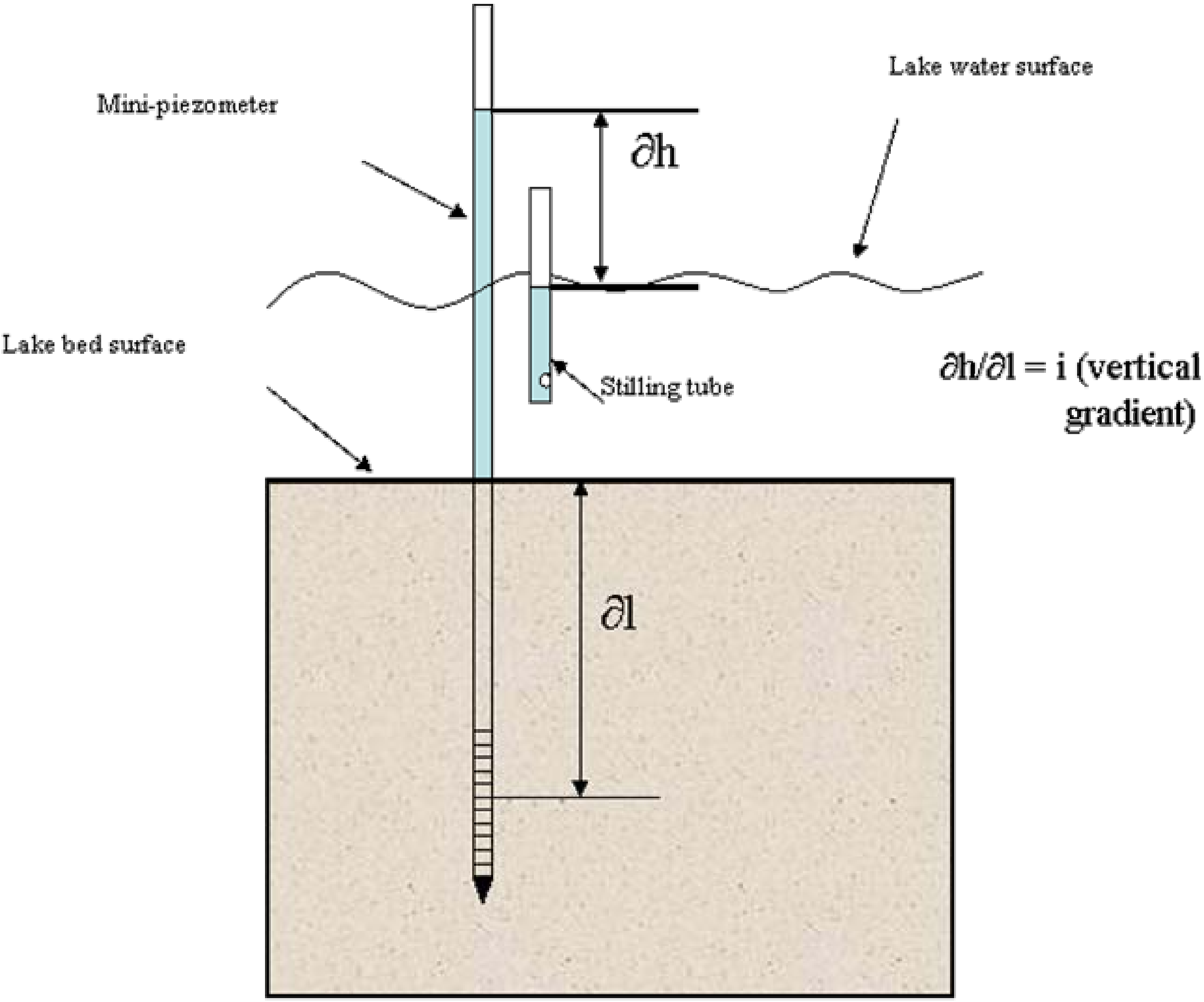


Figure 8 – Measurement Method to Estimate Vertical Hydraulic Gradient at Study Piezometers

this report; nitrite concentrations in groundwater are presumed to be negligible), total Kjeldahl nitrogen (TKN), total persulfate nitrogen (TPN), ammonia as N (ammonia-N), total dissolved solids (TDS), and chloride.

Nutrient samples other than OP were collected into containers pre-preserved with sulfuric acid (H_2SO_4). Sample containers were labeled and immediately stored in an ice-filled cooler for transport to Ecology's Manchester Laboratory for analysis. Table B-2 of Appendix B lists container types, holding times and preservation information for each of the analytes of interest.

Information provided by the lab indicated that seven of the 15 samples submitted during the May round for testing of nutrients other than OP were judged to be still too turbid for analysis without additional filtration by the lab. These seven samples (MLGW-01, -04D, -05, -06, -09, -10, and -11) were filtered by the lab prior to analysis using a 0.45 micron filter.

To prevent further turbidity problems, and to process samples consistently, *all* nutrient samples submitted to the lab during the July and October sampling rounds (including OP samples) were field filtered using dedicated in-line 0.45 micron capsule filters attached to the outlet end of the pump. Accordingly, all values for TP for the July and October rounds are reported as total dissolved phosphorus (TDP).

When using in-line filters, the filter was presoaked with formation water pumped from the piezometer, and the first 100ml of filtrate was discarded prior to filling the sample container. To reduce project costs, analysis for TKN was dropped after May due to the lack of detections. Total dissolved iron and total dissolved manganese analyses were added for the July and October rounds to provide additional insight into the phosphorus transport controls affecting the study area groundwater. Iron and manganese samples were field filtered as described above, and preserved by adding nitric acid (HNO_3) to the container immediately after collection.

Measurement of field parameters at the seep (MLGW-03) and spring (MLGW-14) stations required the collection of a grab sample, or diversion of flow into the flow cell chamber (Table C-1). Samples for analysis were collected directly into the appropriate container, or when applicable, into a rinsed syringe for filtration and subsequent transfer to the sample container.

Sampling of the domestic well (MLGW-15) followed a similar procedure to the piezometers. Field parameters were measured during purging by attaching a high volume flow cell to an outside faucet near the well via an adapter hose. Purge flow rates averaged approximately 2 gallons per minute (gpm). Purging continued until activation of the well pump, and stabilization of field parameters (Table C-1). After purging, samples were collected directly from the faucet, or when applicable, via a rinsed syringe for filtration and subsequent transfer to the appropriate sample container. Samples were collected prior to any water treatment equipment, and are considered reliable indicators of the groundwater quality adjacent to the well.

All samples were immediately placed on ice and were transferred to Manchester Laboratory for analysis. Test methods and detection limits for the field and laboratory analytes are presented in Table B-3 of Appendix B.

Quality Assurance

Field Quality Assurance

A primary concern for this project was the assurance that the piezometer water samples accurately represent the groundwater in the aquifer. As discussed above, a variety of steps were taken in the field to limit downward leakage of lake water along the piezometer annulus during sampling. This included maintaining piezometer pumping rates at low levels, and continuous comparison of the field parameter profile of the purge water versus the lake water. The depth of the piezometer intake below the sediment/water interface (a mean screen-midpoint depth of approximately 4 feet) further helped to isolate the intake from the lake.

The radar charts in Figure C-1 show distinct differences between the water quality conditions of the lake and that of the piezometers, indicating successful isolation of the piezometers from the lake. Since the vertical hydraulic gradient between the piezometers and the lake was consistently upward (groundwater discharge condition), annular leakage would have favored groundwater movement into the lake.

Because pumping may induce a temporary reversal of gradient in the vicinity of the piezometer intake, a dye test also was conducted for most of the piezometers as an additional measure of annular leakage of lake water (Table B-1). To conduct this test, several water soluble, biodegradable, fluorescent dye tablets were placed immediately adjacent to the piezometer tubing at the sediment/water interface. The water in the flow cell and sample containers was then continuously observed during purging and sampling for evidence of dye capture. No dye was observed in water from any of the tested piezometers.

These controls suggest that the piezometer samples collected are representative of groundwater quality conditions immediately beneath the lake.

Unless otherwise noted, the field sampling procedures and equipment use described were applied in a consistent manner at each sampling station. The use of consistent field procedures helped to minimize the degree of variability introduced into the project results.

Analytical Quality Assurance

A variety of analytical quality assurance test samples were collected and evaluated during this project. These tests, which included analysis of blanks, standards, and split duplicates, are described in Appendix D. Test results indicate that no significant bias was introduced into the project analytical results by the sampling or filtration equipment, containers, or preservatives used during the study.

All results reported by Manchester Laboratory were subject to an initial quality assurance review by the lab. On the basis of this review, most of the project results were considered acceptable by the laboratory for use without qualification. Exceptions identified by the laboratory are discussed in detail in Appendix D, and are designated by the appropriate qualifiers in the data tables.

During the author's subsequent data review, the reliability of the May TP results was questioned. Due to concerns regarding the tendency of phosphorus to sorb to soil particles (e.g. to surface coatings of iron and manganese hydroxides), entrainment of suspended particles in sample water is of particular concern when examining the TP results. It is commonly assumed that larger particles are not likely to be transported significant distances through an aquifer matrix under normal flow conditions. The turbidity in a number of the May samples reported by the laboratory suggests that some fine particles may have been artificially mobilized to the piezometer as a result of pumping, despite the presence of the piezometer filter fabric.

Suspended phosphorus-bearing particles or metal oxide complexes collected as part of a sample could potentially be subject to dissolution when exposed to the preservation acid, resulting in desorption of the phosphorus (Stollenwerk, 2002). Desorption of previously bound phosphorus to the dissolved phase would positively bias the sample results, providing an unreliable indicator of the mobile fraction of phosphorus in the aquifer. As a result, all TP results from the May round that required lab filtration were rejected by the author. Because of the similarity to the concentrations reported for later rounds, and the lack of turbidity, the TP results for May samples that did not require lab filtration were retained, but are flagged as possible overestimates (MLGW-02, -03, -07, -08, -12, -13, -14, and -15). The results for TP for the July and October rounds were field filtered, and are listed for accuracy as TDP.

The unfiltered and lab-filtered May concentrations reported for the nitrogen species also were compared closely to the values reported from later rounds when all samples were field filtered. As expected, little discernable difference was noted in the nitrate-N results, since nitrate is not preferentially bound to the solid phase. Accordingly, the May nitrate-N concentration data were retained for use, but are flagged appropriately. The comparison for TPN, and ammonia-N data suggested that the values for the more turbid, lab-filtered May samples did potentially exhibit false positive results. All results from May for these parameters (as well as TKN) for samples that required lab filtration were therefore rejected. The unfiltered sample values were retained for use, but are flagged as possible overestimates.

The author's qualification of the TP results for May (and the laboratory's qualification of the TDP values for October), raise concerns regarding the reliability of these data sets for characterization. Many authors have noted that OP is normally the dominant phosphorus species in groundwater; this was confirmed for the Moses Lake area by comparing the unqualified data from the July round (OP averaged 86% of TDP concentrations). The average percentage of TP as OP for the May sampling round (68%) is uncharacteristically low. The average percentage of TDP as OP for the October data (119%) clearly indicates that the TDP values underestimate the true condition. Because the July TDP values are unqualified concentrations, and because they report at reasonable levels in comparison to the OP data, they are considered the most reliable measure of the TDP condition in the Moses Lake area. The qualified May and October TP/TDP values are reported here, but any conclusions drawn about the distribution and concentration of TDP in the study area are based largely on the July results.

Blind field duplicates were collected during each sampling round and submitted to the laboratory for analysis to evaluate overall sampling and analytical precision. A detailed discussion of these results is presented in Appendix D. For the concentration ranges of interest, the precision of the project data was routinely below the target percent relative standard deviation (% RSD) identified by Pitz (2001). The mean % RSD for the key study parameters, OP and TDP, were 1.5% and 4.6%, respectively, well within the acceptable limits established for the project.

The quality assurance testing and analysis implemented for this study collectively indicate that, with the qualifications noted, the results presented are acceptable for the study purposes.

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Results

Hydraulic Gradient

Table E-1 in Appendix E presents the relative head measurements and vertical gradient estimates for each of the piezometers by sampling round. Figure 9 illustrates the direction and magnitude of the vertical gradient for each piezometer. A positive (upward) vertical hydraulic gradient between the aquifer and the lake was reported for the majority of the measurement events. A downward (negative) vertical gradient was observed on only one occasion during the study period, at station MLGW-07 in July. A significant decrease in the gradient was observed at station MLGW-10 between the May and July rounds, and a significant increase was observed at stations MLGW-07, -08, -09, and -10 between the July and October rounds. The gradient measurements were comparatively stable over time for the remaining stations.

Groundwater Quality Results

Laboratory results for the three sampling rounds are summarized in Table 1.

Figure 10 illustrates the geographic distribution of nitrate-N concentrations over the study area for each sampling round. Figure 11 shows the relationship between different nitrogen species. Nitrate-N is the dominant nitrogen form in the study area groundwater, representing an average median value of 84% of the TPN concentration over the three sampling rounds. Nitrate-N concentrations range from less than 1 to over 12 mg/L. Only one measurement (MLGW-08, October) was reported above the MCL value of 10 mg/L.

Nitrate-N and TPN concentrations generally increase from north to south along the lake, although specific stations (MLGW-01, -07, -11, -13) show very low concentrations for these parameters. Stations MLGW-03, -06, -08, -09, -10, -12, and -14 consistently show nitrate-N values above the concentration indicating a likely anthropogenic influence (3 mg/L). The reported nitrate-N and TPN concentration values were consistent between sampling rounds for the majority of the stations. A significant increase in nitrate-N and TPN concentration was observed at station MLGW-08 over the course of the study, with a particularly notable change between May and July. No TKN was detected at any of the stations during the May round; analysis for TKN was discontinued in later rounds.

Figure 11 indicates an inverse relationship between ammonia-N and nitrate-N concentrations, most notable at stations MLGW-01, -07, -11, and -13. Where detected, ammonia-N concentrations were reported at <1 mg/L. Ammonia-N was routinely reported as undetected at the remaining stations. Organic nitrogen concentrations, estimated by taking the difference between the TPN value minus the sum of the nitrate-N and ammonia-N concentrations, was interpreted as the dominant nitrogen form at stations MLGW-01, -07, -11, and -13, although absolute concentrations were estimated to be less than 2 mg/L.

Figure 12 illustrates the geographic distribution of OP concentrations over the study area by sampling round. Figure 13 presents graphs of OP concentrations reported during the study and

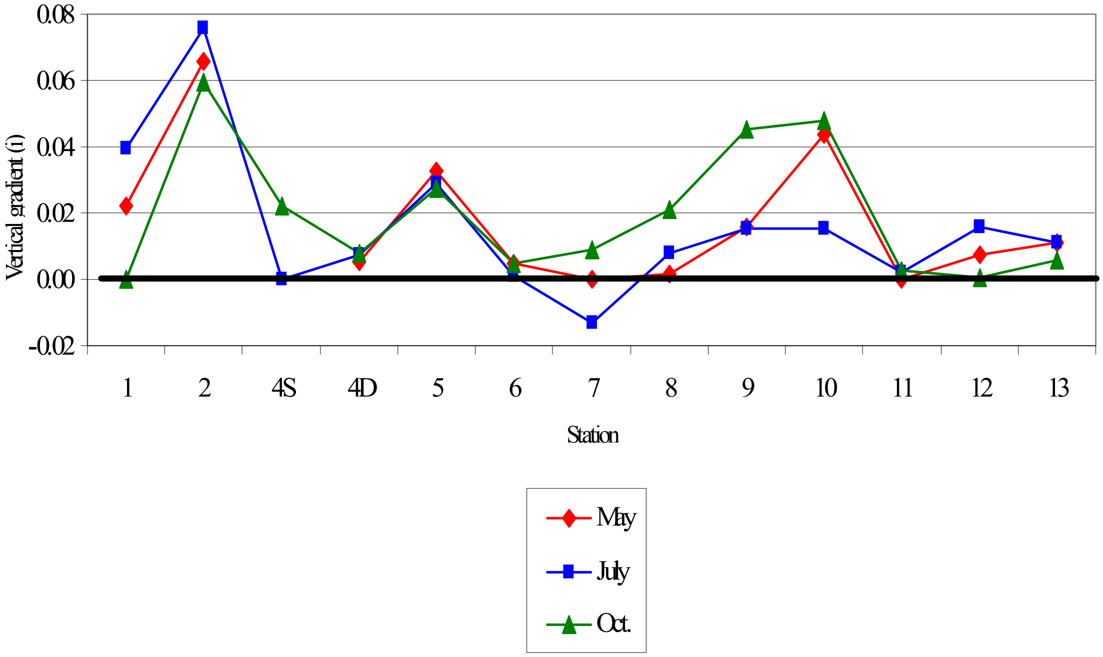


Figure 9 – Estimated Vertical Hydraulic Gradient at Study Piezometers

Table 1 - Groundwater Quality Results (mg/L)

	Total Dissolved Solids (TDS)			Chloride (Cl)			Dissolved Ammonia-N		
Station	May 2001	July 2001	Oct 2001	May 2001	July 2001	Oct 2001	May 2001	July 2001	Oct 2001
MLGW-01	1060	NM	NM	12.4	NM	NM	R	0.121	NM
MLGW-02	247	240	236 J	4.71	4.64	4.45	0.010 U,NF	0.010 U,J	0.010 U,J
MLGW-03	332	341	348 J	6.30	6.41	6.70	0.010 U,NF	0.010 U,J	0.010 U,J
MLGW-04S	NM	363	344 J	NM	7.05	7.92	NM	0.028	0.041 J
MLGW-04D	260	267	321 J	7.29	7.73	8.71	R	0.010 U,J	0.010 U,J
MLGW-05	356	360	355	7.21	7.16	7.22	R	0.010 U,J	0.010 U,J
MLGW-06	403	443	437	21.1	20.5	20.9	R	0.010 U,J	0.010 U,J
MLGW-07	245	353	392	4.85	4.33	8.71	0.010 U,NF	0.010 U,J	0.023 J
MLGW-08	275	502	603	5.57	26.5	28.7	0.010 U,NF	0.010 U,J	0.010 U,J
MLGW-09	NM	435	459	NM	15.6	17.2	R	0.010 U,J	0.010 U,J
MLGW-10	419	444	455	17.6	22.8	22.3	R	0.010 U,J	0.010 U,J
MLGW-11	1790	1590	1140	66.5	46.6	16.3	R	0.090	0.062 J
MLGW-12	535	580	717 J	42.4	45.6	NM	0.021 NF	0.010 U,J	0.010 U,J
MLGW-13	825	745	695	27.9	18.6	12.4	0.033 NF	0.010	0.038 J
MLGW-14	458	471	476	14.4	15.6	17.1	0.010 U,NF	0.010 U,J	0.010 U,J
MLGW-15	250	268	274	5.38	7.59	9.71	0.010 U,NF	0.010 U,J	0.010 U,J

NM - Not measured

R - Results rejected

U - Analyte not detected at or above the reporting limit

J - Result is an estimate

NF - Sample not filtered at 0.45 micron level; reported value may overestimate true dissolved concentration

Table 1 (page 2) - Groundwater Quality Results (mg/L)

	Dissolved Nitrate+Nitrite-N (Nitrate-N)			Total Dissolved Kjeldahl Nitrogen (TKN)			Total Dissolved Persulfate Nitrogen (TPN)		
Station	May 2001	July 2001	Oct 2001	May 2001	July 2001	Oct 2001	May 2001	July 2001	Oct 2001
MLGW-01	0.047 LF	0.060	NM	R	NM	NM	R	0.720	NM
MLGW-02	1.33 NF	1.31	1.30	0.500 U,NF	NM	NM	1.37	1.32	1.34 J
MLGW-03	5.43 NF	4.86	5.26	0.500 U,NF	NM	NM	5.62	6.02	6.64 J
MLGW-04S	NM	0.011	0.087	NM	NM	NM	NM	0.168	0.237 J
MLGW-04D	0.223 LF	0.104	0.566	R	NM	NM	R	0.182	0.658 J
MLGW-05	2.37 LF	2.06	2.13	R	NM	NM	R	2.14	2.22 J
MLGW-06	4.70 LF	4.08	4.06	R	NM	NM	R	4.45	4.87 J
MLGW-07	0.132 NF	0.010 U	0.112	0.500 U,NF	NM	NM	0.251	0.203	0.346 J
MLGW-08	0.101 NF	8.51	12.6	0.500 U,NF	NM	NM	0.171	9.60	14.6 J
MLGW-09	6.43 LF	4.99	6.02	R	NM	NM	R	6.13	7.39 J
MLGW-10	5.30 LF	5.43	5.25	0.500 U,NF	NM	NM	R	5.56	7.17 J
MLGW-11	0.010 U,LF	0.010 U	0.010 U	R	NM	NM	R	1.23	0.717 J
MLGW-12	9.41 NF	9.22	5.79	0.500 U,NF	NM	NM	11.4	10.7	5.88 J
MLGW-13	0.010 U,NF	0.010 U	0.010 U	0.500 U,NF	NM	NM	0.132	0.240	0.255 J
MLGW-14	9.56 NF	8.74	8.50	0.500 U,NF	NM	NM	10.6	9.70	9.99 J
MLGW-15	1.90 NF	2.34	2.96	0.500 U,NF	NM	NM	2.11	2.86	3.41 J

NM - Not measured

R - Results rejected

U - Analyte not detected at or above the reporting limit

J - Result is an estimate

NF - Sample not filtered at 0.45 micron level; reported value may overestimate true dissolved concentration

LF - Sample was lab filtered just prior to analysis; reported value may overestimate true dissolved concentration

Table 1 (page 3) - Groundwater Quality Results (mg/L)

	Dissolved Orthophosphate as P (OP)			Total Dissolved Phosphorus as P (TDP)		
Station	May 2001	July 2001	Oct 2001	May 2001(a)	July 2001	Oct 2001
MLGW-01	0.146	0.145 J	NM	R	0.216	NM
MLGW-02	0.044	0.046	0.0468	0.091 NF	0.060	0.034 J
MLGW-03	0.032	0.031	0.0374	0.045 NF	0.046	0.025 J
MLGW-04S	NM	0.182	0.188	NM	0.198	0.178 J
MLGW-04D	0.044	0.062	0.0558	R	0.069	0.045 J
MLGW-05	0.069	0.086	0.0782	R	0.094	0.065 J
MLGW-06	0.116 J	0.124	0.116	R	0.140	0.100 J
MLGW-07	0.106 J	0.168	0.157	0.133 NF	0.178	0.142 J
MLGW-08	0.097	0.086	0.0839	0.151 NF	0.099	0.072 J
MLGW-09	0.067 J	0.087	0.0728	R	0.100	0.061 J
MLGW-10	0.187 J	0.169	0.191	R	0.199	0.170 J
MLGW-11	1.05	1.15	1.42	R	1.40	1.33 J
MLGW-12	0.148	0.170	0.184	0.230 NF	0.181	0.173 J
MLGW-13	0.18	0.245	0.249	0.267 NF	0.229	0.233 J
MLGW-14	0.029	0.038	0.0321	0.042 NF	0.047	0.023 J
MLGW-15	0.043 J	0.058	0.0613	0.055 NF	0.073	0.051 J

NM - Not measured

R - Results rejected

U - Analyte not detected at or above the reporting limit

J - Result is an estimate

NF - Sample not filtered at 0.45 micron level; reported value may overestimate true dissolved concentration

(a) - May values represent total phosphorus (TP)

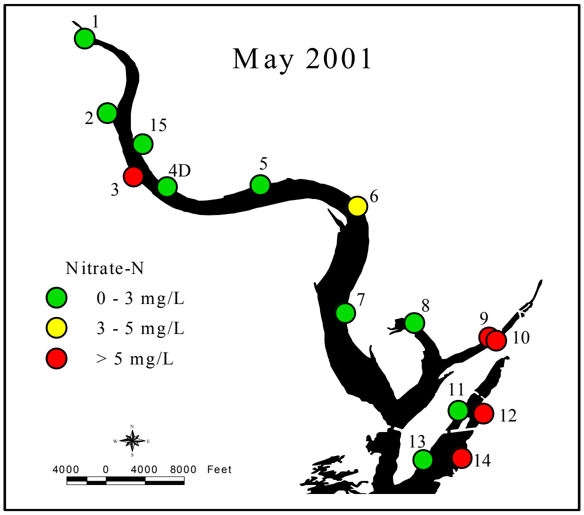
Table 1 (page 4) - Groundwater Quality Results (mg/L)

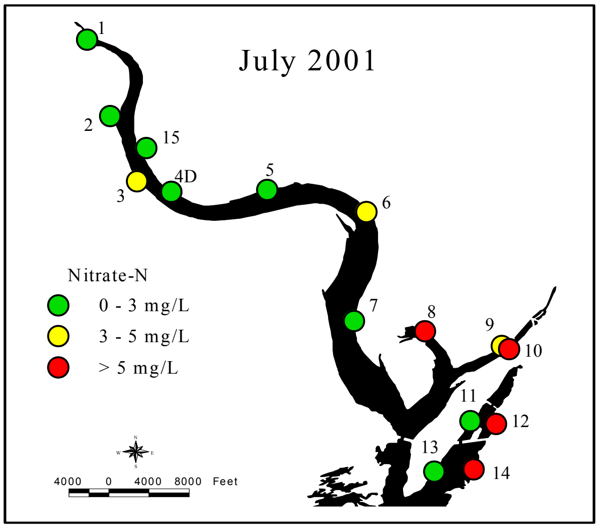
	Total Dissolved Iron (Fe)			Total Dissolved Manganese (Mn)		
Station	May 2001	July 2001	Oct 2001	May 2001	July 2001	Oct 2001
MLGW-01	NM	7.620	NM	NM	0.159	NM
MLGW-02	NM	0.020 U	0.020 U	NM	0.002 U	0.001 U
MLGW-03	NM	0.020 U	0.020 U	NM	0.002 U	0.001 U
MLGW-04S	NM	0.129	0.227	NM	0.240	0.233
MLGW-04D	NM	0.020 U	0.020 U	NM	0.010	0.0043
MLGW-05	NM	0.020 U	0.020 U	NM	0.002 U	0.0019
MLGW-06	NM	0.020 U	0.020 U	NM	0.004	0.0023
MLGW-07	NM	0.045	0.062	NM	0.047	0.0575
MLGW-08	NM	0.020 U	0.020 U	NM	0.002	0.0027
MLGW-09	NM	0.020 U	0.020 U	NM	0.012	0.0077
MLGW-10	NM	0.020 U	0.020 U	NM	0.002 U	0.001 U
MLGW-11	NM	0.089	0.057	NM	0.0287	0.0248
MLGW-12	NM	0.020 U	0.020 U	NM	0.002	0.0017
MLGW-13	NM	0.020 U	0.020 U	NM	0.0347	0.0207
MLGW-14	NM	0.020 U	0.020 U	NM	0.002 U	0.0013
MLGW-15	NM	0.020 U	0.020 U	NM	0.002 U	0.0010

NM - Not measured

U - Analyte not detected at or above the reporting limit

J - Result is an estimate





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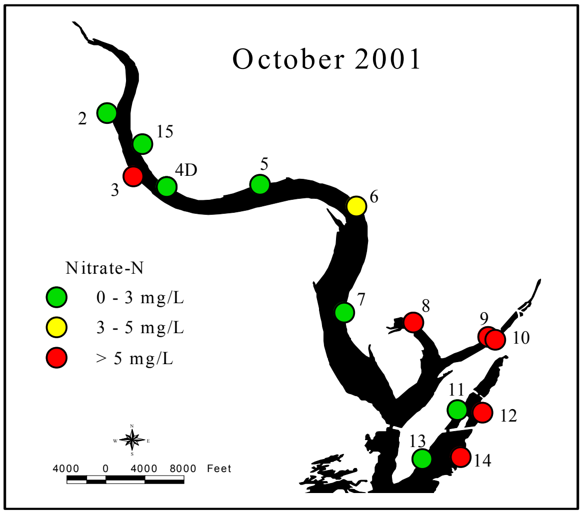

Figure 10 –
Nitrate-N Concentration
May, July, October 2001

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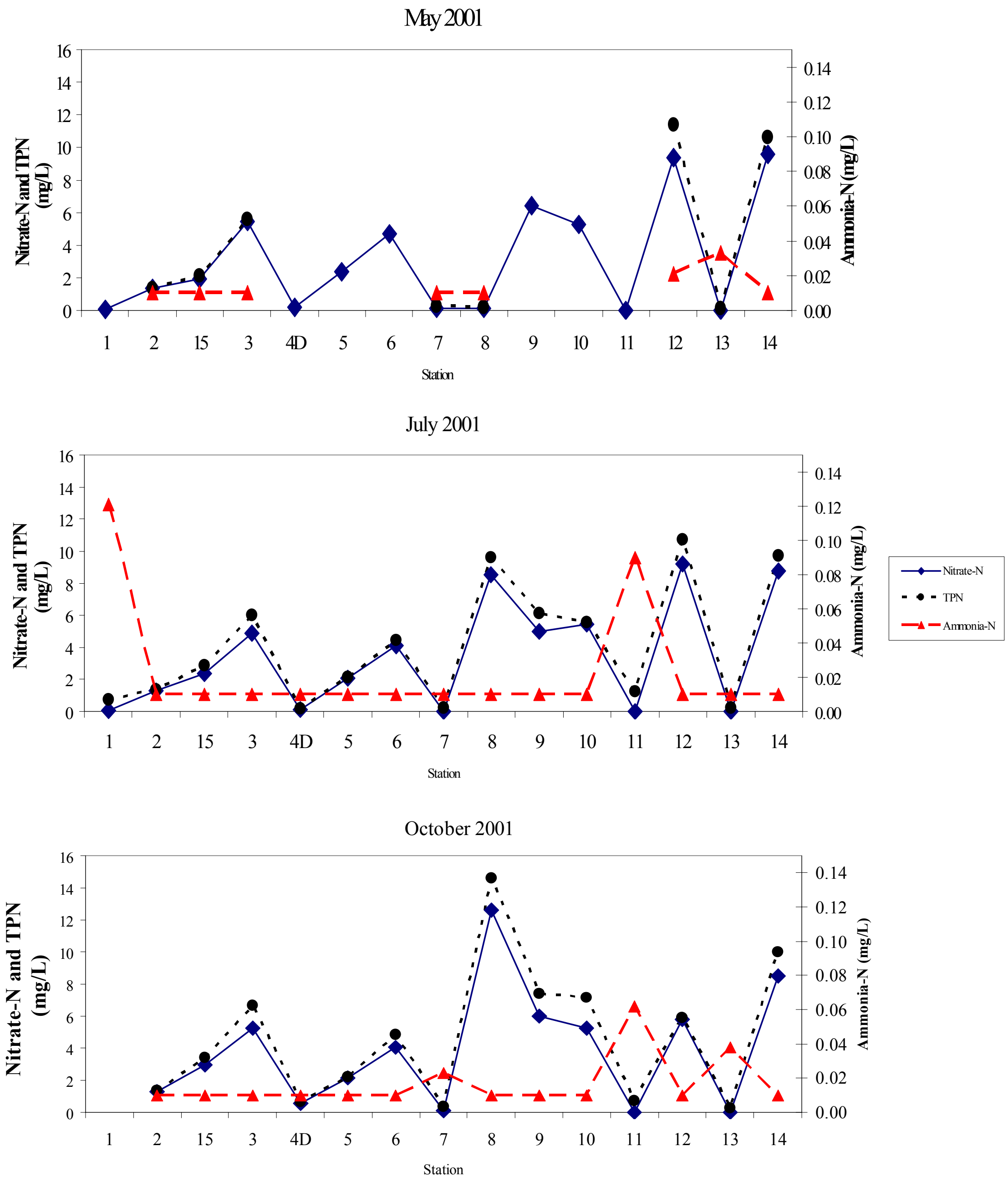


Note: Stations on the lower axis of the graphs are arranged in approximate geographic order from north to south

Figure 11 –
Nitrogen Concentrations - May, July, October 2001

compare these values to the proposed TMDL TP lake criteria of 0.050 mg/L as P. As expected, OP is the dominant form of phosphorus in the study area groundwater, and comprised on average 86% of the TDP during the July sampling round. The percentage of TDP as OP at station MLGW-01 (67%) was notably lower in July than other stations, suggesting a significant organic phosphorus content.

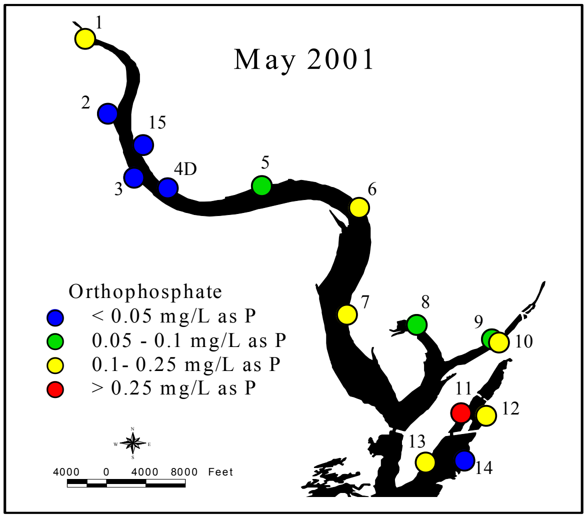
The concentrations reported for OP (0.031-1.42 mg/L as P) were essentially stable between sampling rounds for the majority of the stations, although a progressive increase in OP concentration was noted at station MLGW-11 between May and October. Orthophosphate concentrations generally increase from north to south, with the exception of MLGW-01 (the station located in the wetland), and the Pelican Horn spring at station MLGW-14. The majority of samples (75%) show OP concentrations above the 0.050 mg/L level. The OP concentrations at station MLGW-11 were reported by the laboratory at levels approximately an order of magnitude above the remaining stations (Figure 13).

Figures 14, 15, and 16 compare OP concentrations against the values reported for the key indicator parameters TDS, chloride, iron, manganese, and dissolved oxygen (note that the concentration axis for OP is presented on a logarithmic scale in these figures). Higher OP concentrations are closely associated with higher TDS and chloride, with an overall increase from north to south (specific conductance values show a similar pattern). With the exception of station MLGW-01, TDS and chloride concentrations from stations in undeveloped areas were normally less than 375 and 10 mg/L, respectively. Values for these parameters in the more developed areas to the south (generally 400-800 mg/L and 20-45 mg/L, respectively) were frequently well above those observed to the north.

The TDS and chloride values observed at MLGW-01 (>1000 mg/L and >12 mg/L) were notably higher than the other stations located in undeveloped areas. The OP concentration at MLGW-01 also was elevated significantly above the values seen at other undeveloped stations. The concentration for both TDS and chloride dropped significantly over the course of the study at station MLGW-11, while the OP values increased. Station MLGW-08 showed a significant increase in the chloride concentration between the May and July rounds; an increase also was reported for TDS during this period. Orthophosphate concentrations between May and July at station MLGW-08 showed no significant change.

The reported OP (and when detected iron, manganese, and ammonia-N) concentrations are inversely related to dissolved oxygen (Figures 15 and 16). With the exception of the wetland station MLGW-01, dissolved oxygen concentrations at the northernmost stations (MLGW-02, -15, and -03) were consistently greater than 6 mg/L. Dissolved oxygen concentrations at piezometers in developed areas toward the south end of the lake were routinely less than 5 mg/L. Suboxic conditions (<1 mg/L dissolved oxygen) were observed at stations MLGW-04D, -06, -07, -11, and -13. Increases in dissolved oxygen concentration were noted for a number of the stations between July and October.

Detectable concentrations of iron were observed at stations MLGW-01, -07, and -11, coincident with elevated OP values. Dissolved manganese was detected at low levels at most stations. The stations with the highest manganese concentrations (MLGW-01, -07, -11, -13) were also sites of elevated OP.



Number next
to station represents
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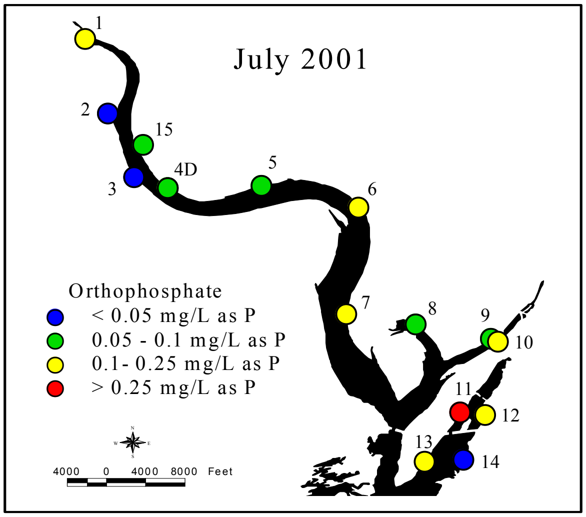

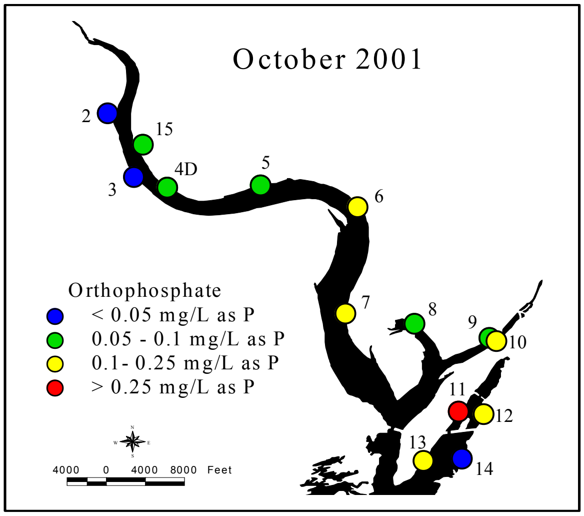

Figure 12 –
Orthophosphate
Concentration -
May, July, October 2001

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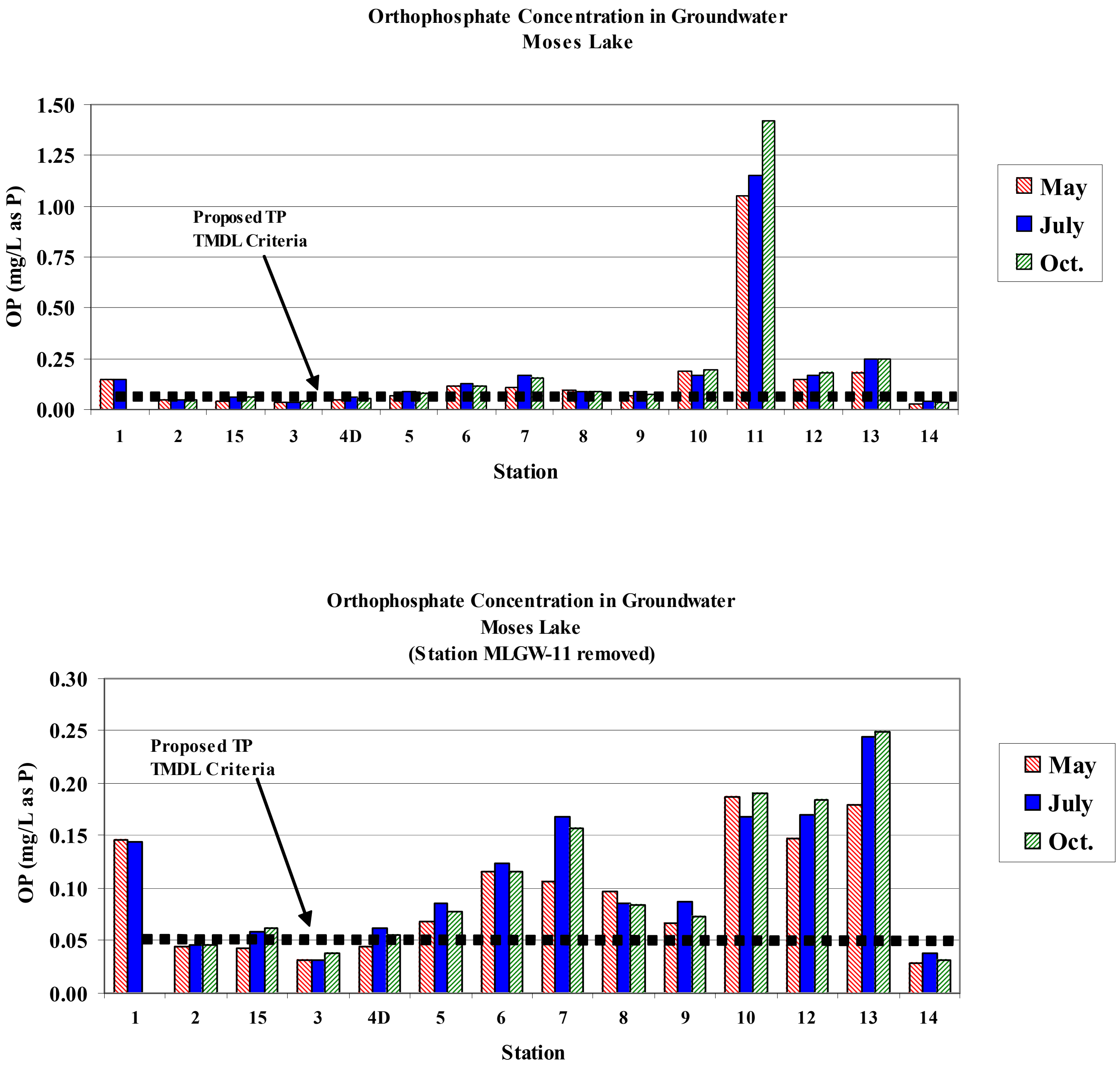
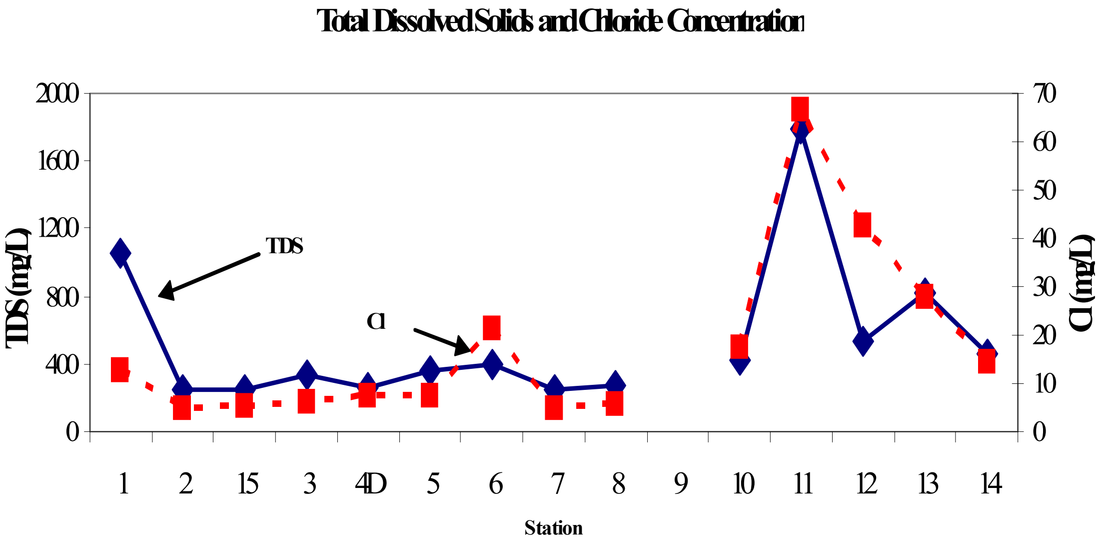


Note: Stations on the lower axis of the graphs are arranged in approximate geographic order from north to south

Figure 13 – Orthophosphate Concentration in Groundwater


Total Dissolved Solids and Chloride Concentration

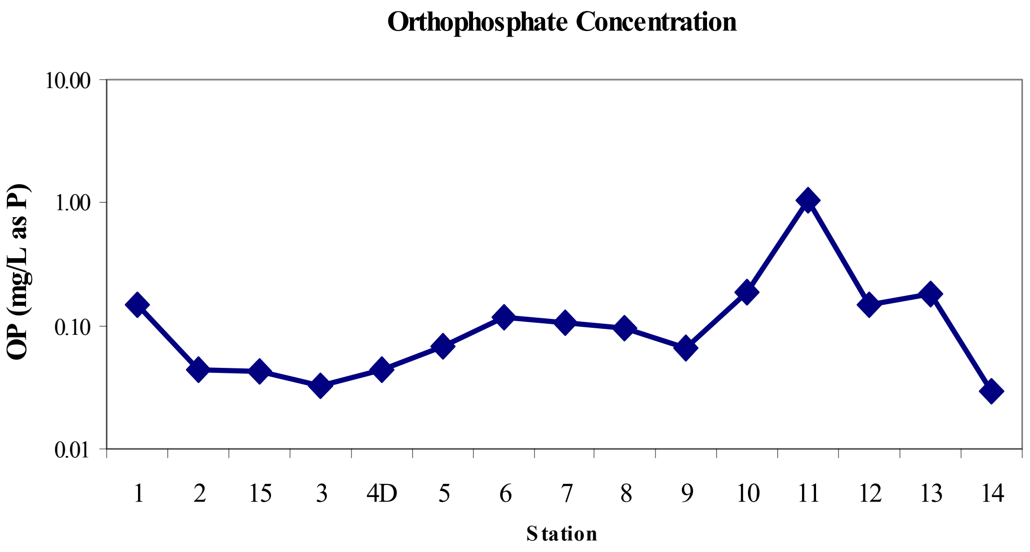
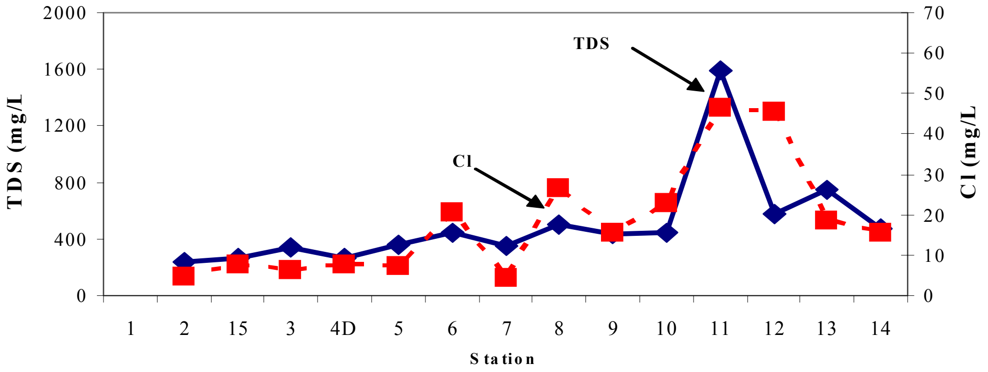
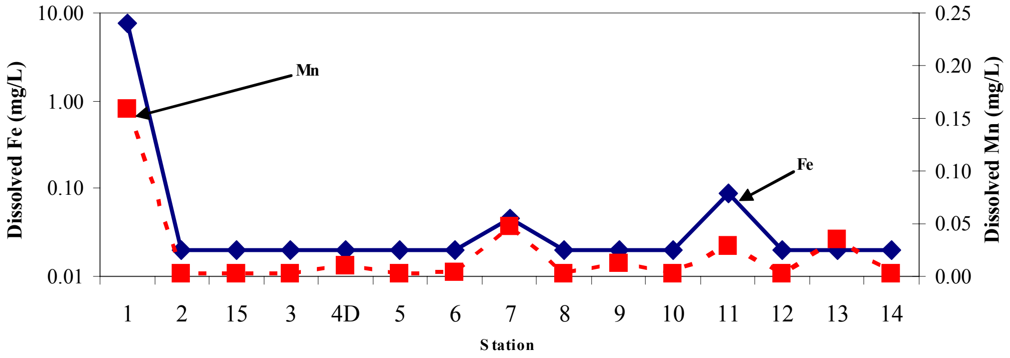

Orthophosphate Concentration

Figure 14 – TDS, Chloride, and Orthophosphate Concentration - May 2001

Total Dissolved Solids and Chloride Concentration



Dissolved Iron and Manganese Concentration



Orthophosphate and Dissolved Oxygen Concentration

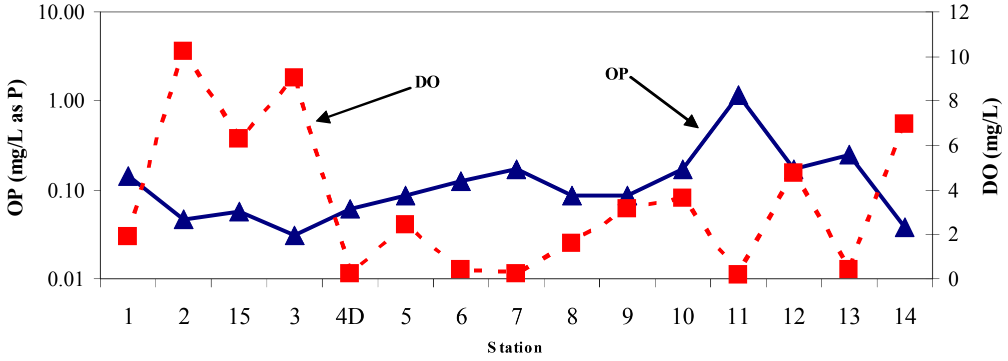
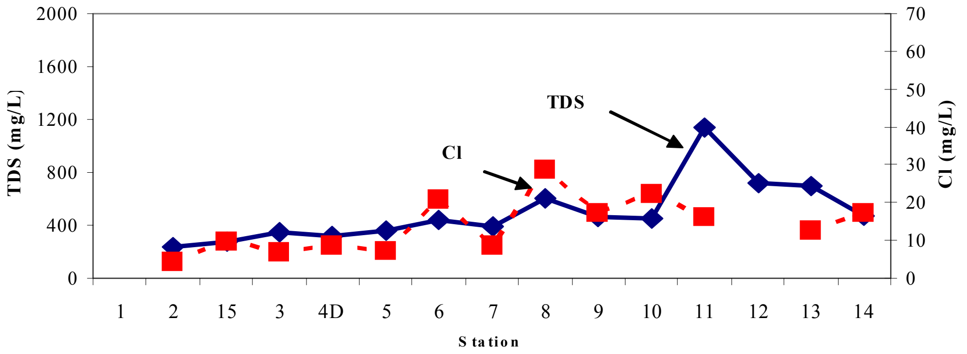
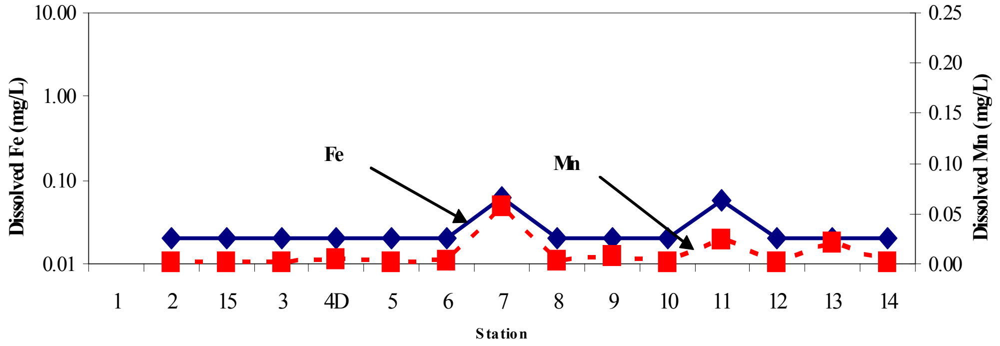


Figure 15 – TDS, Chloride, Iron, Manganese, Dissolved Oxygen, and Orthophosphate Concentration - July 2001

Total Dissolved Solids and Chloride Concentration



Dissolved Iron and Manganese Concentration



Orthophosphate and Dissolved Oxygen Concentration

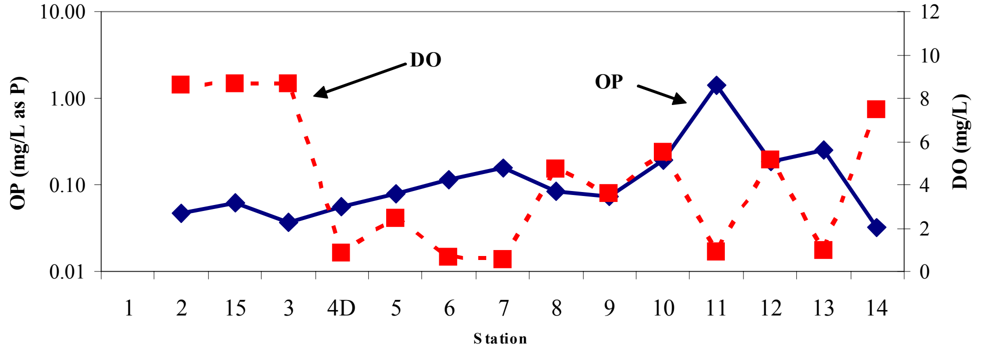


Figure 16 – TDS, Chloride, Iron, Manganese, Dissolved Oxygen, and Orthophosphate Concentrations - October 2001

Discussion

Hydraulic Conditions

The data presented in Table E-1 and Figure 9 indicate that the selection of station locations for the study was successful in providing representative water quality data from discharging areas. As seen on Figure 9, the majority of the hydraulic gradient measurements demonstrated an upward vertical gradient between the aquifer and the lake, indicating groundwater discharge to the lake along those portions of the shoreline. Gradient variations between stations are interpreted to be a function of variations in the hydraulic conductivity of the littoral zone sediments, and upgradient aquifer conditions. The decrease in gradient at MLGW-10 between May and July indicates a lowering of the upgradient water table, possibly by mid-summer groundwater withdrawals. The subsequent increase in gradient at stations MLGW-07, -08, -09, and -10 between July and October indicates a rise of the water table position, possibly by the recharge influence of upgradient irrigation near the end of the growing season, or rebound due to reduced groundwater withdrawals.

Groundwater Quality Conditions

Representativeness of the Data

As described above, a variety of steps were taken to ensure that the samples collected and analyzed for this study were representative of groundwater quality conditions as close to the point of discharge as possible. The evidence indicates that potential problems with the capture of lake water through annular leakage were avoided, and that all of the sampling stations are located in areas of groundwater inflow. The study design was developed to characterize groundwater quality near the end of the flow path through the aquifer, just prior to its discharge into the lake. The approach of sampling in the near vicinity of the surface water/groundwater interface provides the advantage of measuring the net effects of aquifer diffusion, dispersion, and attenuation on target parameter concentrations, while also improving the chances of accounting for near-shore sources of contamination.

As noted earlier, the piezometer locations are probably biased towards finer-grained settings. This bias may result in underestimates of the typical groundwater phosphorus concentrations present beneath the majority of the lake, due to the tendency of phosphorus to sorb more readily to finer-grained sediment particles. It also is possible that phosphorus-bearing particles larger than 0.45 micron in diameter are mobile in very coarse-grained areas of the aquifer, which are often the areas of preferential flow. If this is the case, then the measured OP and TDP values may actually represent lower-bound estimates of the groundwater phosphorus concentration moving towards the lake/aquifer interface.

Phosphorus concentrations reported for the seep (MLGW-03) and spring (MLGW-14) stations also likely underestimate the local inflow conditions adjacent to these sites since these samples were exposed to atmospheric conditions prior to sample collection. As discussed earlier, contact with the atmosphere drives redox-controlled sorption of phosphorus to surfaces of metal oxides, and can transfer phosphorus from the dissolved to the solid phase. Phosphorus previously dissolved in the aquifer would therefore be filtered out during sampling, biasing the reported values lower than the true condition. The fact that the OP concentrations for these two stations were the lowest of the stations tested supports this assumption.

These same geochemical processes can result in significant changes to the chemistry of groundwater as it moves through the interface zone between the aquifer and lake, particularly within the upper centimeters of the lake-bed sediments. In this zone, a transformation from reducing to oxidizing conditions can trigger sorption and precipitation reactions that immobilize previously dissolved phosphorus in the bed sediments (Robertson et al., 1998).

These reactions may decrease the phosphorus concentration of the water as it exits the aquifer and enters the lake to values below that measured by the piezometers. Consequently, the concentrations reported for this study may represent the maximum value of dissolved phosphorus expected to discharge to the lake from the underlying aquifer. However, previous researchers have noted that areas of high seepage velocity are less likely to create significant changes in phosphorus concentration at the interface (Rounds, 2001). The large overall grain size of the littoral zone sediments, in combination with the high groundwater velocities previously reported, suggest that reductions in phosphorus concentration in the final centimeters of the flow path may be of limited significance for this study area. In fact, areas of preferential groundwater flow and discharge are likely to be the least well-suited for attenuation of dissolved phase phosphorus.

In addition, vertical profile data collected at station MLGW-04 (MLGW-04S screen midpoint depth = 2.5 feet; and MLGW-04D screen midpoint depth = 4.5 feet) suggest that (at least at this location) reducing conditions actually increase closer to the sediment/water interface. The shallow piezometer at this location (MLGW-04S) displayed detectable levels of dissolved iron, while no iron was reported in the deeper sample (MLGW-04D)(Table 1). Concentrations of dissolved manganese were significantly higher in the shallower piezometer (>0.2 mg/L shallow vs. <0.005 mg/L deep), as was OP (>0.180 mg/L as P, shallow vs. <0.065 mg/L as P, deep). Ammonia-N was detected in the shallower pore water, but was undetected at depth. In contrast, concentrations of nitrate-N increased with depth. Collectively, these data show reducing conditions increase near the interface; indicating that the ability of redox sorption processes to attenuate phosphorus in groundwater discharging at the interface may be comparatively limited[1].

Area Background Groundwater Quality Conditions

Groundwater quality conditions at station MLGW-02 are considered the best measure of "area background", defined as the groundwater condition *least impacted by upgradient anthropogenic sources*. This station lies at the northern end of the lake, well away from the urban development to the south. The land use immediately upgradient of this station is primarily undeveloped rangeland and shrubland. Few, if any, residences, on-site septic systems, or known point

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[1] It is also important to remember that sorbed phosphorus accumulated in sediments at the interface can be remobilized to the water column if redox conditions change.

discharges are present upgradient of the station for a minimum of 3 miles, and livestock densities on the rangeland are low, although extensive irrigated agriculture is present further upgradient. The similarity in geologic setting and littoral zone character of this area relative to the remainder of the study area indicates that this site provides a reasonable upper-end concentration for geologically-derived phosphorus (and other target parameters) in the study area shallow aquifer.

Groundwater conditions at the background station exhibit comparatively low values for specific conductance (<335 uS/cm), chloride (<5 mg/L), TDS (<250 mg/L), ammonia-N (undetected at 0.010 mg/L), and iron and manganese (undetected at 0.020 mg/L). Dissolved oxygen values were approximately 8 to 10 mg/L, indicating oxic conditions. Nitrate and TPN concentrations were less than 1.5 mg/L. Orthophosphate concentrations were below 0.050 mg/L as P (mean: 0.046 mg/L as P), and TDP in July was measured at 0.060 mg/L as P. As discussed above, while it is possible that the slightly higher OP concentration found at this station in comparison to other areas in the central Columbia Basin represents a local variation in natural conditions, the highly permeable character of the shallow aquifer material and upgradient land uses suggest that the quality of the groundwater at MLGW-02 also may be influenced by anthropogenic activities.

While located in a similarly undeveloped area, conditions at the seep station MLGW-03 are probably not representative of ambient groundwater discharge quality due to the possible influence of livestock on the water quality of this surface sample (suggested by field observations and the higher nitrate-N and TDS values compared to MLGW-02). In addition, water contact with the atmosphere prior to sample collection probably negatively biased the phosphorus results toward lower values.

Station MLGW-01 also is located in a remote, undeveloped area of the lake, near the mouth of Rocky Ford Creek. Few residences, irrigated fields, or known point sources are present within several miles upgradient of this station, although phosphorus-bearing sediment from upstream sources on Rocky Ford Creek deposited in this area may influence the conditions observed. Despite the undeveloped character of the surrounding area, OP groundwater concentrations reported for this station (~0.150 mg/L as P) were significantly higher than other sites at the northern end of the lake.

The elevated phosphorus at this site is likely a natural condition, although additional upgradient sampling would be required to confirm this assumption. The phosphorus content above area background is interpreted to be the result of contributions by the wetland soils in the final portion of the flow path. Wetland peat and muck soils have been shown to be a natural source of groundwater phosphorus (DeBusk, 2002; USGS, 2002).

If natural organic material present in wetland soils is dried (and therefore oxidized by atmospheric exposure), aerobic decomposition can occur. Decomposition of organic matter can release organic phosphorus (this station has a significant estimated organic phosphorus fraction in comparison to other stations on the lake), some or all of which is converted to OP. The phosphorus released is normally rapidly sorbed onto the surface of metal hydroxides (e.g. iron and manganese) present within the soil. Rewetting of the soils creates a reducing condition that allows the reductive dissolution of the iron and manganese, which in turn releases sorbed phosphorus to solution.

Reducing conditions at this site are indicated by the low nitrate-N, detectable concentrations of ammonia-N, low dissolved oxygen, and elevated levels of dissolved iron and manganese. The process of release of phosphorus from wetland soils to pore water is enhanced by cyclic wetting and drying cycles, which create alternately reducing and oxidizing conditions. The annual manipulation of the lake surface elevation by approximately 4 feet presumably mimics this pattern at this station.

Field observations of the exposed littoral zone and examination of wetlands maps (USFW, 1991) suggest that the extent of wetland soils like those seen at MLGW-01 is limited on the Moses Lake shoreline. This indicates that the water quality concentrations (particularly the phosphorus) observed at this station are not representative of area background conditions for the majority of the groundwater entering the lake. Also for this reason the impact of groundwater discharge from the wetland soils surrounding the MLGW-01 station is not considered to be significant to the overall nutrient budget for the lake. The piezometer stations throughout the remainder of the study area were typically installed in mineral sediments with a significantly lower organic content then seen at the MLGW-01 station. The impact of natural *buried* organic material on the phosphorus concentrations reported for the remaining stations is unknown, but is not anticipated to change the conclusions reached in this report.

Anthropogenic (Human-Caused) Impacts on Groundwater Quality Conditions

Moving south from the MLGW-02 station, there is a progressive decline in the overall quality of the groundwater sampled from the study piezometers. Concentrations of the key indicator parameters for anthropogenic impact (TDS, chloride, nitrogen as nitrate-N or ammonia-N, and SC) generally increase towards the south, as do values for the target parameters OP, and TDP (Figures 10-16). In contrast to the conditions observed at MLGW-02, the dissolved oxygen measured at a number of the sites to the south is depressed below 5 mg/L, with several sites (MLGW-04D, -06, -07, -11, -13) showing suboxic or anoxic conditions (<1 mg/L dissolved oxygen)(Figures 15 and 16).

The occurrence and dissolved concentration of phosphate in the study area groundwater above the background condition is interpreted to be controlled by a combination of a) the phosphorus attenuation capacity of the aquifer, b) the prevailing redox (and pH) conditions in the aquifer that govern phosphate adsorption/desorption and precipitation/dissolution reactions, and c) the availability, proximity, density and loading history of upgradient anthropogenic sources.

Previous authors have noted that the aquifer material in the Moses Lake area is, for the most part, poorly suited for the attenuation of anthropogenic phosphate loads (Bain 1986, 1987a). The very coarse-grained character of the local deposits, and the slightly alkaline pH of the study area groundwater (pH 7-9), both naturally favor the presence and mobility of phosphorus in the dissolved phase. With the exception of the area southeast of Pelican Horn, the generally limited ability of area sediments to attenuate phosphate suggests that the remaining two factors (prevailing redox and pH condition, and availability and proximity of source) are the most important in controlling the occurrence of phosphate in groundwater above or below levels critical to the trophic status of the lake.

Specifically, locations where the reducing conditions that favor phosphorus desorption coincide with anthropogenic sources in close proximity to the lake (e.g. water table close to land surface; nearshore, higher density land use) show the highest OP concentrations (e.g. MLGW-11: mean OP 1.21 mg/L as P, and MLGW-13: mean OP 0.23 mg/L as P). Locations where oxic conditions coincide with sources that are distant from the shoreline (e.g. water table deep below land surface; low density land use further from shore) show the lowest OP concentrations (e.g. MLGW-15: mean OP 0.054 mg/L as P). Areas where an upgradient source is present, but reducing conditions are absent, show intermediate OP concentrations (e.g. MLGW-08, -09, -10, -12: OP range 0.067-0.184 mg/L as P). The presence of elevated phosphorus in the dissolved phase even under aerobic conditions suggests the possibility that the available attenuation capacity of the aquifer is nearly or completely exhausted at these locations (Walter et al., 1995).

With the exception of several stations discussed below, the data collected during this study cannot identify a specific anthropogenic source area that is the likely origin of the elevated nutrients measured. Most stations probably are influenced by a mixture of sources and land uses both close and distant. As discussed earlier, land use data indicate that stations MLGW-05, -07, -12, and -14 are the locations most likely to show impact by agricultural activities, while stations MLGW-07,-08, -09, -10, -11, -12, and -13 are the most likely locations to show impact by the release of wastewater.

Bain (1987a) suggested that the nitrogen-to-phosphorus ratio in groundwater of the Moses Lake area could be used as a distinguishing characteristic between agricultural and urban sources of phosphorus. However, the influence of heterogeneities in geochemical conditions at the point of measurement on the presence or absence of nitrogen or phosphorus species indicates that this is not a reliable approach.

The upgradient land use, and elevated TDS, chloride, and ammonia-N concentrations observed at stations MLGW-11 and -13 suggest that the reducing conditions (i.e. depleted dissolved oxygen values, elevated concentrations of iron and manganese in the dissolved phase, the occurrence of available nitrogen as ammonia-N) are, in contrast to MLGW-01, driven by anthropogenic loading of organic-rich wastewater. Local reducing conditions often occur as oxygen is consumed by biodegradation of wastewater released to the subsurface from drain fields, infiltration ponds or leaking sewer pipes (Walter et al., 1995; Ptacek, 1998; Zanini et al., 1998; Zilkey et al., 2001).

The OP concentrations at station MLGW-11 were notably higher than other stations, even though the adjacent neighborhood is served by the municipal sewer system. Portions of the buried sewer lines in this area are known to be located near the water table, and are several decades old (Henning, 2002).

In mid to late 2001, in response to inflow and infiltration concerns, the City of Moses Lake relined or replaced an approximately 1-mile long section of the sewer main in the area north and northeast of MLGW-11 (as close as 500 feet to the sample station). Records maintained by the city indicate significant changes in flow from the system since the remediation of the sewer line, evidence that leakage from the line to the aquifer may have been a long-term condition. Leakage from these pipes could explain the high phosphate values measured at station MLGW-11, although additional sampling and investigation would be required to confirm this. Additional sampling along this shoreline also would be required to confirm if the OP conditions observed are a local anomaly, or if they represent a wider condition adjacent to this particular land use setting (e.g. the age of the neighborhood, the age of the sewer utility).

The OP concentration at MLGW-11 (and MLGW-13) increased significantly during the study, while the chloride value progressively dropped. The conservative nature of chloride transport in water suggests that a dilution process occurred at these stations during the study period. The increase in OP concentration may be related to an increase of phosphate-bearing recharge, or recharge-triggered changes in the geochemistry of the aquifer that enhance desorption of previously loaded phosphate (Stollenwerk, 2002). Potential recharge sources in residential areas could include watering of fertilized or unfertilized lawns.

Station MLGW-07 also shows evidence for reducing conditions controlling the occurrence of an elevated OP concentration. However, this station does not show coincidentally high chloride and TDS values. This suggests that the reducing conditions and elevated phosphorus values may not be related to a septic or wastewater source. Alternative sources for the elevated phosphorus could include natural processes similar to those observed at MLGW-01, or an agricultural source such as the movement of phosphorus-bearing fertilizer to the water table. The fact that a) this station is immediately downgradient of irrigated cropland, b) that changes in chloride concentration over the study period point to a significant recharge effect on the station's water quality, and c) that the surface deposits in the littoral zone at this site are largely mineral, all suggest the possibility of an agricultural source for the phosphate.

The large increase in the concentrations of nitrate-N, TPN, TDS, chloride, and SC at station MLGW-08 between the May and July sampling rounds (with continued increase through the October round) suggests the impact of a specific upgradient source of wastewater contamination to the aquifer beginning after May. The MLGW-08 station is located offshore of a city park on Lewis Horn called Cascade Park. A public restroom facility serving the park and adjacent campground is located within 100 feet of the shoreline, approximately 130 feet along shore (northwest) from the piezometer.

The data suggest that the restroom is a likely source for the observed change in water quality. The facility is only open to the public beginning in mid-April, and sees the bulk of its use during June and July. Disposal of wastewater from the restroom is via a septic system and on-site drain field (Gonzales, 2002). The deterioration in water quality observed in the piezometer corresponds closely to the change in use of the facility from early May to July, indicating loading to the drain field directly impacts the water quality of the groundwater discharging to the lake. Orthophosphate concentrations are elevated at this station in comparison to background conditions (mean OP = 0.089 mg/L), but the moderately oxic environment appears to limit the mobility of phosphorus at this site.

As discussed earlier, there is no evidence that indicates that the elevated phosphate concentration measured in the groundwater above the area background condition originates from a natural mineral source. The amount of phosphorus present in groundwater in the study area that is attributable to natural sources appears to be negligible when compared to anthropogenic sources.

Relationship of Groundwater Orthophosphate to Land Use

To evaluate the relationship between OP concentrations and anthropogenic sources on a study-wide basis, a GIS-based land use analysis was conducted. To perform this analysis each sampling station in the study area was buffered using a one-mile radius circle. The relative percentage of the total of each mapped land use was calculated for the upgradient half of each circle, using the digital National Land Use Cover Data illustrated in Figure 3 (USGS, 1999).

Land use categories were simplified to allow three major classes of land use: urban development, agricultural development, and undeveloped. Land use types included in the urban development category include urban and suburban residential development, commercial, industrial, and transportation activities, and urban grass landscapes. The agricultural land use category included irrigated cropland, managed pasture/hay fields, and fallow agricultural fields. The undeveloped land use category included native shrubland, unmanaged grasslands and range, wetlands, and other natural landscapes.

Figure 17 presents the relationship between the relative percentage of each land use class in the station buffer area vs. the reported mean OP concentration (note that OP concentrations are presented on a logarithmic scale). The MLGW-01 wetland station is specifically noted on the charts due to the assumption that the elevated OP concentration at this station is a naturally occurring condition that is not related to the upgradient anthropogenic land use. The MLGW-03 seep station and the MLGW-14 spring station are identified separately on the charts due to concerns that the OP concentration at these stations may underestimate the true concentration in the groundwater system, due to oxidation of the water and removal of phosphorus from solution prior to collection.

Figure 17 indicates that OP concentrations increase in relation to the relative percentage of urban development immediately upgradient of the sampling station. An evaluation of these data indicates a statistically significant relationship between these two factors at the 95% confidence level (p-value <0.05; correlation coefficient = 0.60; wetland station removed from the data set). In contrast, the data on Figure 17 suggest that OP concentrations in groundwater entering Moses Lake are not systematically related to the relative percentage of agricultural land present in the vicinity of the site[2]. The data also suggest that OP concentrations are inversely related to the amount of undeveloped land, although no statistically significant correlation was established.

Considering the association of elevated OP with elevated concentrations of wastewater indicator parameters, the correlation between the degree of urban development and the OP concentration indicates that urban sources of wastewater to the aquifer are the primary source impacting the groundwater discharging to the lake. The sources for such wastewater probably include a combination of leachate from septic drain fields, leakage from municipal sewer lines, and the direct discharge/infiltration of partially treated municipal wastewater. Other sources and land uses (e.g. downward migration of phosphorus fertilizer from irrigated agricultural fields) probably also contribute to raising the ambient OP concentration in the study area groundwater,

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[2] This finding applies only to the groundwater pathway and is not applicable to other potential loading pathways to the lake such as surface water discharges.

but are not considered the primary source of the elevated OP concentrations observed at the southern end of the lake.

The fact that agricultural land use does not correlate to elevated phosphorus concentrations in groundwater discharging to the lake may be due to the absence of the reducing conditions that favor phosphorus mobility beneath most of the irrigated agricultural fields in the study area, or the distance of the majority of the agricultural activity from the shoreline (allowing attenuation of the phosphorus). Two of the stations that show the highest percentage of near-shore agricultural development (MLGW-12 and MLGW-14) also are located in the area where the phosphorus attenuation capacity of the aquifer in hydraulic communication with the lake is likely higher than elsewhere.

An established relationship between OP concentrations in study area groundwater and the degree of urban development is notable considering the non-conservative transport characteristics of phosphorus in the subsurface. A similar correlation was established for surface water in the Johnson Creek watershed of northern Oregon by Sonoda et al., 2001, but few examples exist for groundwater-based investigations (Ator and Denis, 1997; PCDPWU, 1998). This further supports the conclusion that the phosphorus attenuation capacity of the subsurface deposits in the Moses Lake area is comparatively limited in its ability to diminish loading from urban sources.

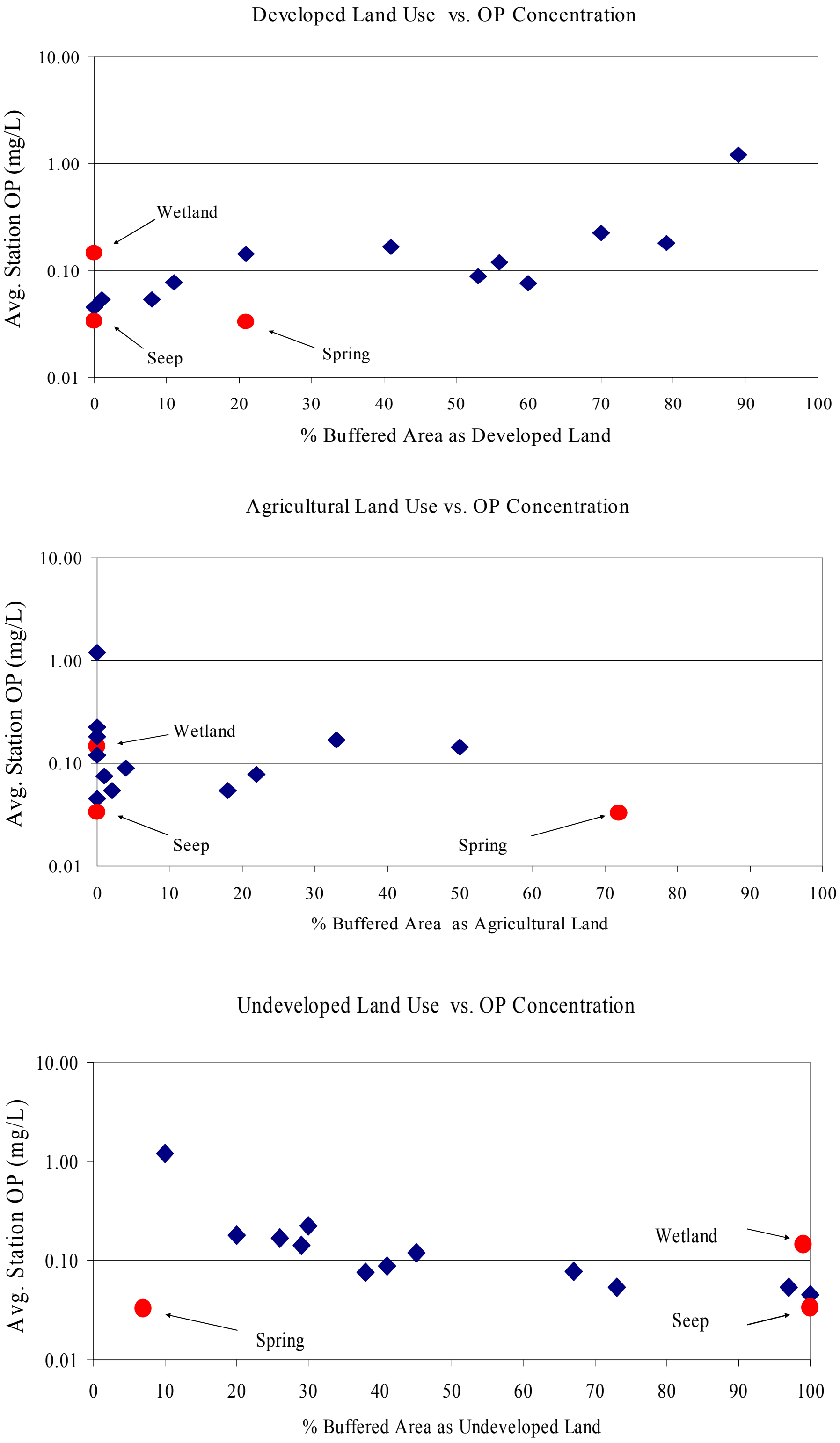

Figure 17 – Comparison of Vicinity Land Use to Orthophosphate Concentration

Loading

Variations in loading of phosphorus by groundwater inflow to different portions of Moses Lake are influenced by three major factors: 1) spatial differences and temporal changes in groundwater phosphorus concentration, 2) spatial differences and temporal changes in the hydraulic gradient between the aquifer and the lake, and 3) spatial differences in the hydraulic properties of the aquifer adjacent to the shoreline.

The data indicate that phosphorus concentrations and vertical gradients were relatively stable during the study period along most of the lake shoreline studied. A significant increase in the vertical gradient was noted during the July round for stations MLGW-07, -08, and -09, and a significant increase in the OP concentration was recorded at station MLGW-11 between May and October. Both of these changes would presumably result in an increase in the groundwater load to the lake in these areas.

The available evidence suggests that the hydraulic properties of the aquifer adjacent to the discharge portions of the lake shoreline are relatively similar, with the exception of the area southwest of the airport (the ‘big bend’), and the area adjacent to the southeastern shoreline of Pelican Horn. In these locations, groundwater entering the lake is moving through the finer-grained deposits of the Ringold Formation. The field descriptions and estimated hydraulic conductivity of these sediments suggest that the rate of phosphorus loading is likely to be lower in these areas (lower groundwater inflow rates and higher phosphorus attenuation capacity).

To provide an estimate of the probable range of phosphorus mass loaded to Moses Lake on an annual basis by groundwater discharge, a loading analysis was conducted. Appendix F describes in detail the procedures and assumptions used for that analysis. The analysis results indicate a lower- and upper-bound range from 400 to 40,000 kg_{op}/yr enter the lake on an annual basis via groundwater discharge. A value between 10,000-20,000 kg_{op}/yr is considered the best estimate of field conditions.

In preliminary evaluations of management strategies for phosphorus loads to Moses Lake, Carroll et al. (2000) identified groundwater contributions as a “controllable” load. This suggests that anthropogenically-derived phosphate loading to the lake by groundwater can be reduced or eliminated by implementing best management practices. However, there remains a possibility that a reservoir of sorbed (or precipitated) phosphorus stored on the aquifer matrix may prevent significant reductions in groundwater concentrations moving towards the lake for many years to come, even if further loading is discontinued. The phosphorus observed in the samples collected during this study may be the result of a) ongoing loading to the aquifer, b) ongoing desorption or dissolution of phosphate mass from past loading, or c) a combination of the two. These factors imply that anthropogenically-derived groundwater loading of phosphate to the lake may not be “controllable” in the short-term.

Conclusions

An evaluation of existing hydrogeologic data indicates that Moses Lake is a flow-through surface waterbody, i.e. groundwater discharges to the lake along portions of the lake bottom, while lake water recharges groundwater in other areas. The bulk of groundwater inflow to the lake is interpreted to occur within the littoral zone along the northwestern and eastern shorelines, with limited additional inflow along the southeastern shoreline. Lake water recharges the aquifer along the southwestern and southern shorelines.

Due to the wide presence and position of fine-grained deposits of the Ringold Formation, a majority of the groundwater discharge to the lake probably derives from shallow, unconsolidated flood deposits, with limited direct interaction from the underlying basalt aquifer. The coarse deposits of the surficial aquifer appear to have a limited natural capacity to restrict the movement of phosphorus input to the system from anthropogenic sources.

No evidence was found to indicate that a natural geologic source of phosphate is present that could explain concentrations of phosphorus in groundwater above the 0.050 mg/L TP threshold criteria established for the lake. As expected, the large majority of the phosphorus in the study area groundwater occurs as OP. The regional groundwater quality data indicate ambient OP concentrations are routinely <0.050 mg/L in the central Columbia Basin. The OP concentration for the study area background station is also <0.050 mg/L.

For the majority of stations measured during this study, the OP concentration in groundwater just prior to its discharge to the lake is higher than the 0.050 mg/L TP threshold criteria. Orthophosphate concentrations generally increase from north to south, and parallel increases in the concentration of parameters that indicate anthropogenic impact on groundwater quality.

The occurrence of OP in groundwater above the area background concentration is interpreted to be a function of variations in the geochemical conditions and attenuation capacity of the aquifer, and variations in loading from anthropogenic sources. Analysis of the land uses upgradient of each sampling station revealed a statistically significant relationship between groundwater OP, and the percentage of urban development upgradient of the site. These findings indicate that urban wastewater sources, probably a combination of leachate from septic drain fields, leakage from municipal sewer systems, and direct discharge/infiltration of partially treated municipal waste water, are the dominant source of phosphate in groundwater discharging to the lake. Agricultural sources likely play a role in raising the ambient phosphate concentration of area groundwater, but are not interpreted to be the primary source of the phosphorus entering the lake via groundwater discharge.

A loading analysis indicates a probable range from 10,000 to 20,000 $\mathrm{kg_{op}}$/yr enters Moses Lake in a dissolved form on an annual basis via groundwater discharge. Given that phosphorus loads from anthropogenic sources are accumulated over time within an aquifer, a reservoir of sorbed or precipitated phosphate mass may be present in the subsurface. Ongoing desorption and dissolution from a reservoir of stored phosphorus could sustain elevated phosphate concentrations in groundwater discharge for many years, even after the cessation of loading. This suggests that anthropogenically-derived groundwater loading of phosphorus to the lake may not be a controllable source under short-term management timeframes.

Recommendations

1. Due to the presence of a public beach in the near vicinity, additional investigation should be considered in the MLGW-08 area to confirm the suspected impact of the public restroom facility at Cascade Park on the water quality along the shoreline.

2. Additional investigation should be considered to evaluate the upgradient and cross gradient extent and origin of elevated orthophosphate as P concentrations in the MLGW-11 area.

3. Additional field and laboratory investigation and modeling should be considered to better evaluate the likelihood of groundwater acting as a long-term source of phosphorus load to the lake.

4. In light of the limited capacity of the study area deposits to attenuate groundwater phosphorus concentrations below concentrations significant to eutrophication, all efforts should be made to limit the continued loading of phosphorus from urban sources to the aquifer. Recommended efforts include:

    - Reevaluate local ordinances governing the design, placement, and use of on-site septic drain fields, particularly in near-shore settings.

    - Continue to evaluate the potential for leakage of wastewater from the municipal sewage system, particularly in near-shore settings.

    - Reevaluate treatment and discharge alternatives, as well as discharge impacts for the Larson Wastewater Treatment Plant. In support of this effort, the sampling regime for groundwater samples collected from monitoring wells in the area of the infiltration system should include the analysis of filtered samples for orthophosphate as P and total dissolved phosphorus.

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Appendices

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Appendix A

Rocky Ford Springs Source Evaluation

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Introduction and Background

In 1997, Cusimano and Ward (1998) conducted a water quality study of the Rocky Ford Creek drainage. Their work included the analyses of samples of the creek's headwater spring, located at the Troutlodge fish hatchery (Figure A-1; note that samples collected at the Troutlodge hatchery actually represent the composite water quality of a number of smaller springs that are integrated at a single point via a manifold system).

During that study, OP concentrations ranging between 0.056 and 0.076 mg/L as P were measured in the spring water. The spring was identified as the most significant source of nutrients to the creek; the authors suggested that agricultural land management practices in the upper Crab Creek drainage was the most probable cause for the elevated nutrients in the discharging groundwater. Carroll (2002) conducted additional sampling at the spring during the 2001 water year, and reported an OP range between 0.065 and 0.109 mg/L as P, with a median value of 0.084 mg/L as P, confirming the spring as an ongoing source of dissolved phosphorus.

Since Rocky Ford Creek ultimately drains to the main arm of Moses Lake, and is a significant surface water source of nutrients to the lake, Ecology's Eastern Regional Office requested an effort to assess the source of the spring water. A detailed, field-based effort to answer this question was beyond the available resources of Ecology's Environmental Assessment Program. However, existing published information regarding the hydrogeologic regime of the area surrounding the spring was evaluated to determine if a likely source area could be identified, or alternatively, ruled out.

A limited field sampling effort was also undertaken to support this evaluation. The purpose of the sampling was to characterize the hydrochemical profile of the spring water at the hatchery, and determine if there are similarities to the hydrochemical profile of several area lakes that may act as a source for the spring flow. Figure A-1 shows the four sampling locations selected, which include: a sample from the Rocky Ford Creek source spring (RFS-1), a sample from Soap Lake (SPLK-1), a sample from Brook Lake (BRLK-1), and for comparative purposes, a sample from the Gloyd Seeps area (GLSP-1). Plans to obtain a sample from Round Lake were unsuccessful due to access issues.

Review of Existing Data

Soap Lake, a mineral-rich evaporite lake located approximately five miles north-northwest of the Rocky Ford Springs (Figure A-1), has been suggested as a potential source area for a significant proportion of the dissolved phosphorus detected in the spring water. While historic water quality information indicates that Soap Lake does have a highly elevated concentration of dissolved phosphorus, existing evidence indicates that Soap Lake is not hydraulically connected to the springs.

Soap Lake has been described as lying within a hydraulically and geologically closed basin, bounded by low permeability CRBG bedrock and fine grained deposits of the Ringold Formation (Mundorff and Bodhaine, 1954). Detailed geologic mapping by Mundorff and Bodhaine between Soap Lake and Rocky Ford Springs identified a broad ridge on the basalt bedrock

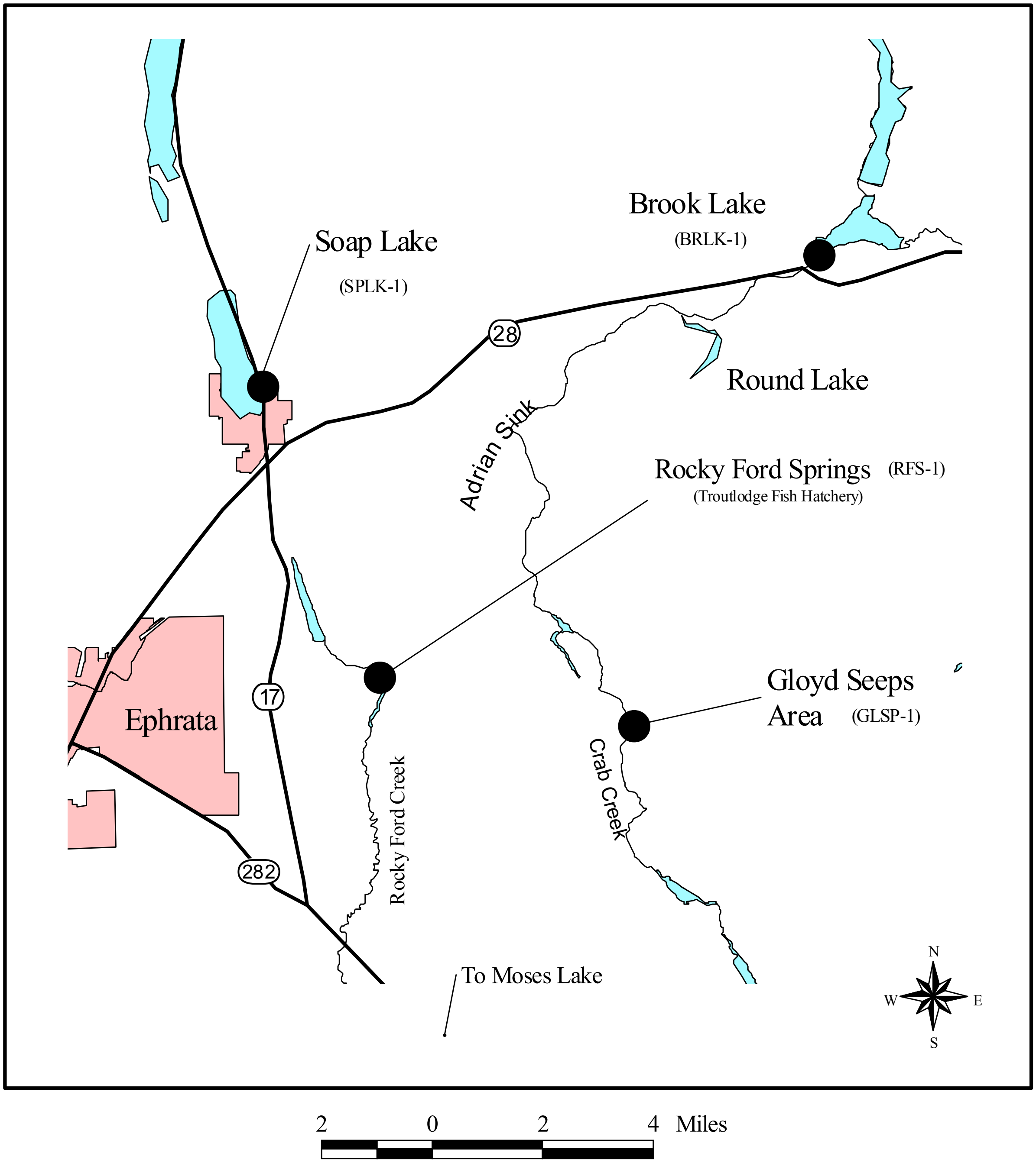


Figure A-1
Geochemistry Sampling Locations
Rocky Ford Springs Source Evaluation

surface underlying younger unconsolidated deposits, running roughly east-west, approximately $1^{1/2}$ miles south of the lake. The lowest point on this ridge lies at approximately 1085 feet elevation; in contrast the Soap Lake surface elevation has recently been measured at approximately 1073 feet (Hubbs, 2002).

The contour of the local water table surface is strongly controlled by this structural feature, as shown by Mundorff and Bodhaine's 1953 data, as well as Walters and Grolier's 1958 water table maps for the Columbia Basin project area (Walters and Grolier, 1960). Groundwater flow north of this ridge is directed northward towards the southern end of Soap Lake; groundwater south of this feature flows south and southeast towards Rocky Ford Creek. A flow regime of this character will prohibit the movement of lake water towards the area of the springs. Figure A-2 shows the approximate position of this hydraulic boundary.

While the effect of large-scale irrigation from the Columbia Basin Project has undoubtedly resulted in changes to the water table configuration within the Basin since the 1950's, there is no evidence that the flow directions of groundwater at the southern end of the lake have been reversed. In fact, the area-wide rise in water-table levels (from both canal leakage and infiltration from irrigated agriculture) has been suggested as the source of *increased* flow of groundwater into the Soap Lake basin from the south (Mundorff and Bodhaine, 1954; Hubbs, 2002). This change reportedly resulted in an elevation rise in the lake water surface, and flooding of shoreline properties.

In response to these issues the U.S. Bureau of Reclamation (USBR) implemented the Soap Lake Protective Works in 1952-53. The objective of this hydraulic control project, which remains active, is to intercept irrigation-derived groundwater that is flowing towards the southern end of the lake. A narrow, north-south trending gravel-filled channel in the basalt surface directs the majority of this flow. The protective works is comprised of a line of production wells across this channel that intercept and withdraw groundwater from the south prior to its entry into the lake (Mundorff and Bodhaine, 1954). This pumping, in effect, imposes further hydraulic isolation on the lake, prohibiting connection to the Rocky Ford Springs.

Mundorff et al. (1952) reported that groundwater in the shallow aquifer between Brook Lake and Adrian Sink (Figure A-2) occupies a narrow, highly transmissive gravel paleochannel underlying the surface drainage of Crab Creek. This channel is bounded to the north and south by lower permeability flows of the Wanapum Basalt. The surface flow of Crab Creek is reportedly lost to the aquifer in this area under all but the most extreme flow conditions. Mundorff and coauthors stated that this water resurfaces further downgradient as discharge to the Rocky Ford Creek source springs, a conclusion supported by Bain (1985).

In addition to geologic evidence, the authors reported that water level fluctuations in wells installed in the gravels in the Adrian Sink area corresponded closely to fluctuations in the spring discharge rate. The 1958 potentiometric head map of Walters and Grolier (1960) indicates a distinct hydraulic connection between groundwater in the Brook Lake/Stratford area and the Rocky Ford source springs. A narrow trough exists on the water table surface between these two areas, reflecting the highly transmissive nature of the underlying deposits. The hydraulic gradient distribution within this trough directs groundwater flow to the southwest, towards the springs. Figure A-2 illustrates the approximate groundwater flow directions in the study area, as determined from existing published data.

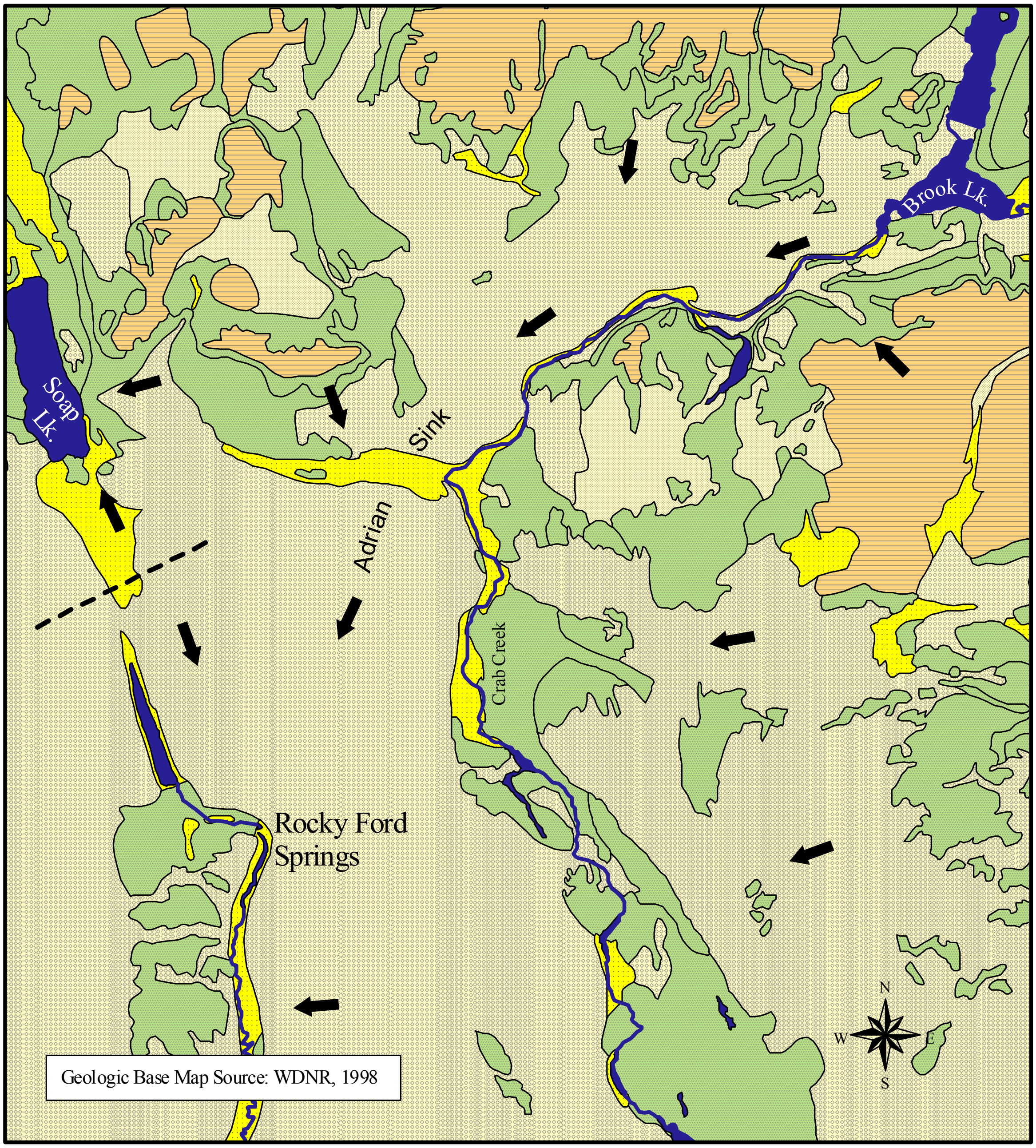
Bain (1985) reported on an investigation he conducted in the area surrounding the Rocky Ford springs in an attempt to identify the source of phosphorus in the spring water. His work included water quality sampling of both local shallow groundwater and surface water, with analysis for phosphorus and specific conductivity.

In examining the available water quality data, Bain identified a large contrast between the specific conductivity measured in the groundwater adjacent to Soap Lake and the springs, indicating that these samples were from distinct water sources. Bain also found that the phosphorus concentrations in the groundwater lying to the west of the spring (the Ephrata area) were significantly lower than those measured at the spring. This evidence, as well as an examination of the geologic conditions of the area between the springs and Ephrata, led Bain to conclude that the primary source of the phosphorus in the springs did not lie to the west. In contrast, Bain found that the phosphorus content of groundwater at several locations upgradient of the Adrian Sink area was comparable to or even greater than that seen at the springs.

Bain also supported the interpretation that there is a hydraulic connection between groundwater in the Adrian and Brook Lake area, and the Rocky Ford springs. On this basis, he concluded that impoundments located upstream of the Adrian area (including Brook Lake and Round Lake) were the most likely source of the majority of the phosphorus in the groundwater moving to the springs. Bain noted that the nutrients stored in these lakes ultimately originate from agricultural activities in the upper Crab Creek basin in upper Grant and Lincoln counties. Bain found no available evidence for a natural stratigraphic source of phosphate that could explain the elevated phosphorus concentrations present in the groundwater feeding the spring (Bain, 1987a).

An alternative, or additional, explanation for the phosphorus condition at the Rocky Ford springs may be found in the historic land-use patterns overlying the high permeability deposits between Stratford and the area southwest of Adrian (and perhaps, additionally, those immediately north of the springs in the Grant Orchards area) (Figure A-3). Recent land use patterns in the Stratford-Adrian area include irrigated alfalfa, corn and wheat cultivation, along with livestock pasture and low density rural residential development (Larson, 2002; USGS, 1999). Review of historic aerial photographs indicates these land uses have been present for at least 30 years. These activities may result in the movement of phosphorus to the water table via loss of phosphorus fertilizers applied to crops, as well as contributions from septic systems and manure. The highly permeable nature of the soils and subsurface deposits in this area suggest the low soil attenuation capacity necessary to mobilize phosphorus to, and then through, the aquifer.

The OP concentrations measured in the spring discharge are not unlike those measured at station MLGW-05 on Moses Lake (0.69-0.86 mg/L). This station is located downgradient of mixed low density rural residential development and agricultural land use similar to that found in the Adrian area. This suggests the possibility that in this geologic/land-use setting, the concentration range observed at the springs may be the expected “ambient” condition for dissolved phosphorus in upgradient groundwater, regardless of the presence of the impoundments. Additional detailed field study of the groundwater quality and flow patterns within and upgradient of the Adrian area would be required to determine the specific phosphorus sources to area groundwater.



Geologic Base Map Source: WDNR, 1998

Figure A-2
Surficial Geology and
General Groundwater Flow Direction
Rocky Ford Springs Source Evaluation

Legend

🟨 Alluvium

🟧 Loess

▨ Missoula Flood - grvls

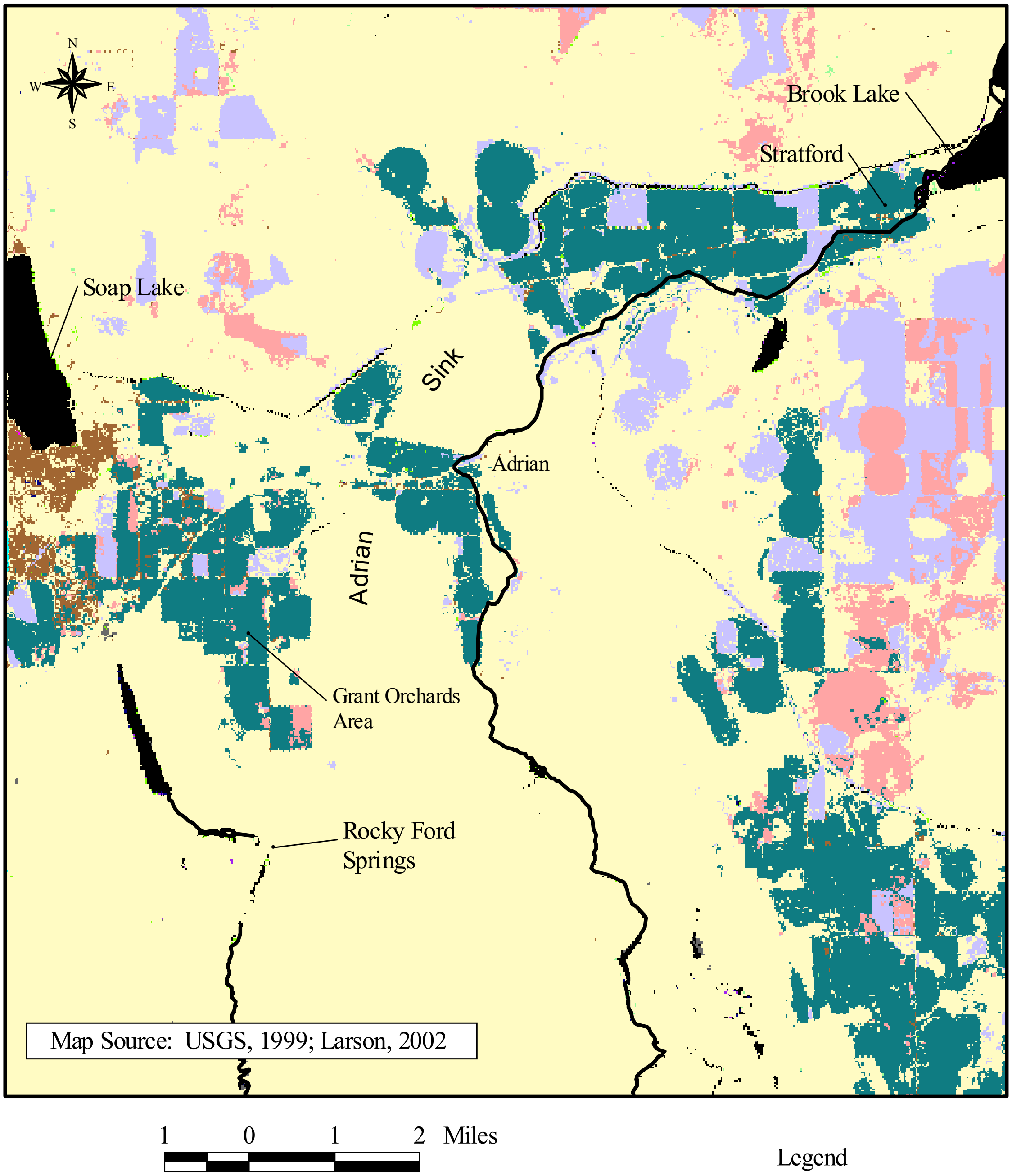
⬜ Missoula Flood - snd/slt

🟩 Col. Rvr. Basalt

🟦 Water

⬉ Approximate Groundwater Flow Direction

- - - Approximate Location of Groundwater Divide



Map Source: USGS, 1999; Larson, 2002

Figure A-3
Land Use Map
Rocky Ford Springs Source Evaluation

Legend

■ Water

▨ Small Grain Crops

▨ Urban Development

□ Shrubland and Grassland

▨ Mixed Irrigated Cropland and Pasture

▨ Fallow Ag Land

Hydrochemical Sampling

The four water samples collected for hydrochemical comparison (Figure A-1) were submitted to Manchester Laboratory for the analysis of TDS, and major cations and anions, including: calcium, magnesium, sodium, potassium, sulfate, chloride, and alkalinity for an approximation of carbonate/bicarbonate concentration (Table A-1). Bicarbonate is assumed to be the predominant carbon dioxide species present under local conditions (Hem, 1989); the bicarbonate concentration was assumed to be 110% of the laboratory-measured alkalinity value.

The data values were collectively plotted on a Piper trilinear diagram for comparison purposes. Table A-2 presents the analytical results, and charge balance errors for these samples. An assumed nitrate concentration of 2.5 mg/L was used for all stations for the purpose of calculating the charge balance. The charge balance error percent for all of the samples was considered acceptable for the purposes of the evaluation.

Table A-2 - Major Cation, Anion, and TDS Concentrations, and Sample Charge Balance Error
Rocky Ford Springs Source Evaluation

Station	Cations (mg/L)				Anions (mg/L)			TDS	Charge Balance[b]
	Ca	Mg	Na	K	HCO3+CO3[a]	SO4	Cl	(mg/L)	Error (%)
RFS-1	49.2	30.9	23.3	5.17	243	49.5	16.5	361	10.7
SPLK-1	2.45	3	5120	468	7700	2950	1800	15900	-1.5
BRLK-1	34.3	18	34.7	9.68	184	45.9	16	324	10.3
GLSP-1	56.8	27.4	55	8.95	342	47.2	11.5	455	10.3

[a] Dominant species assumed to be bicarbonate, estimated as 110% of lab-measured alkalinity

[b] An assumed nitrate concentration of 2.5 mg/L was used for all samples for the purposes of calculating the charge balance.

Results and Discussion

Figure A-4 shows the resulting trilinear diagram plot for all four stations. For Figure A-5, the Soap Lake station has been removed from the trilinear diagram, and the TDS scale has been rescaled to allow easier comparison of the remaining stations.

The Piper data plots indicate that the RFS-1, BRLK-1, and GLSP-1 samples show similarities in water composition, each of the samples plotting as a calcium-bicarbonate or calcium-magnesium-bicarbonate hydrochemical type. This finding is consistent with Turney's (1986) data for groundwater samples from the shallower portions of the aquifer system in upper Grant County. Sample SPLK-1, in contrast, shows a mixed anion-sodium water type, with a distinctly elevated TDS value.

Table A-1 - Parameters, Container Types, Holding Times, Sample Preservation, Test Method, and Practical Quantitation Limit
Rocky Ford Springs Source Evaluation

Parameter	Container	Holding Time	Preservation	Test Method	Practical Quantitation Limit
Total Dissolved Solids	1 L w/m polyethylene	7 days	Chill to 4^{o}C	EPA 160.1	1 mg/L
Chloride	500 ml w/m polyethylene	28 days	Chill to 4^{o}C	EPA 300.0	0.1 mg/L
Calcium	1 L HDPE	6 months	Adjust pH to <2 w/ HNO_3 and chill to 4^{o}C	EPA 200.7	0.010 mg/L
Magnesium	1 L HDPE	6 months	Adjust pH to <2 w/ HNO_3 and chill to 4^{o}C	EPA 200.7	0.010 mg/L
Sodium	1 L HDPE	6 months	Adjust pH to <2 w/ HNO_3 and chill to 4^{o}C	EPA 200.7	0.010 mg/L
Potassium	1 L HDPE	6 months	Adjust pH to <2 w/ HNO_3 and chill to 4^{o}C	EPA 200.7	0.100 mg/L
Alkalinity	500 ml w/m polyethylene	14 days	Chill to 4^{o}C	SM 2320	1 mg/L
Sulfate	500 ml w/m polyethylene	28 days	Chill to 4^{o}C	EPA 330.0	0.5 mg/L

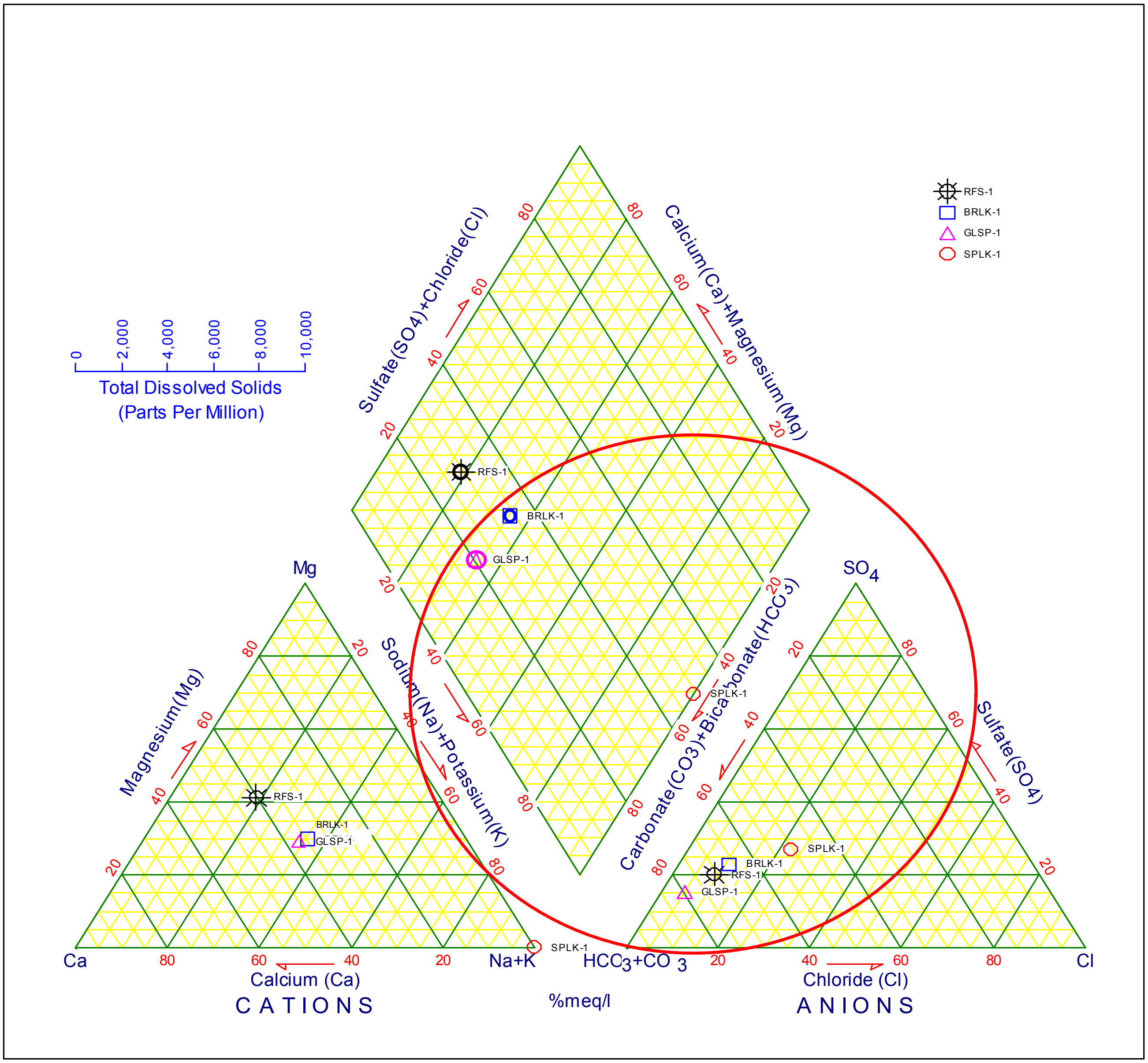


Figure A-4
Piper Diagram Analysis
Rocky Ford Springs Source Evaluation
July 2001
(with Soap Lake data)

Note - The radius of the circles in the diamond-shaped field surrounding each station symbol represent the station concentration for total dissolved solids (TDS)

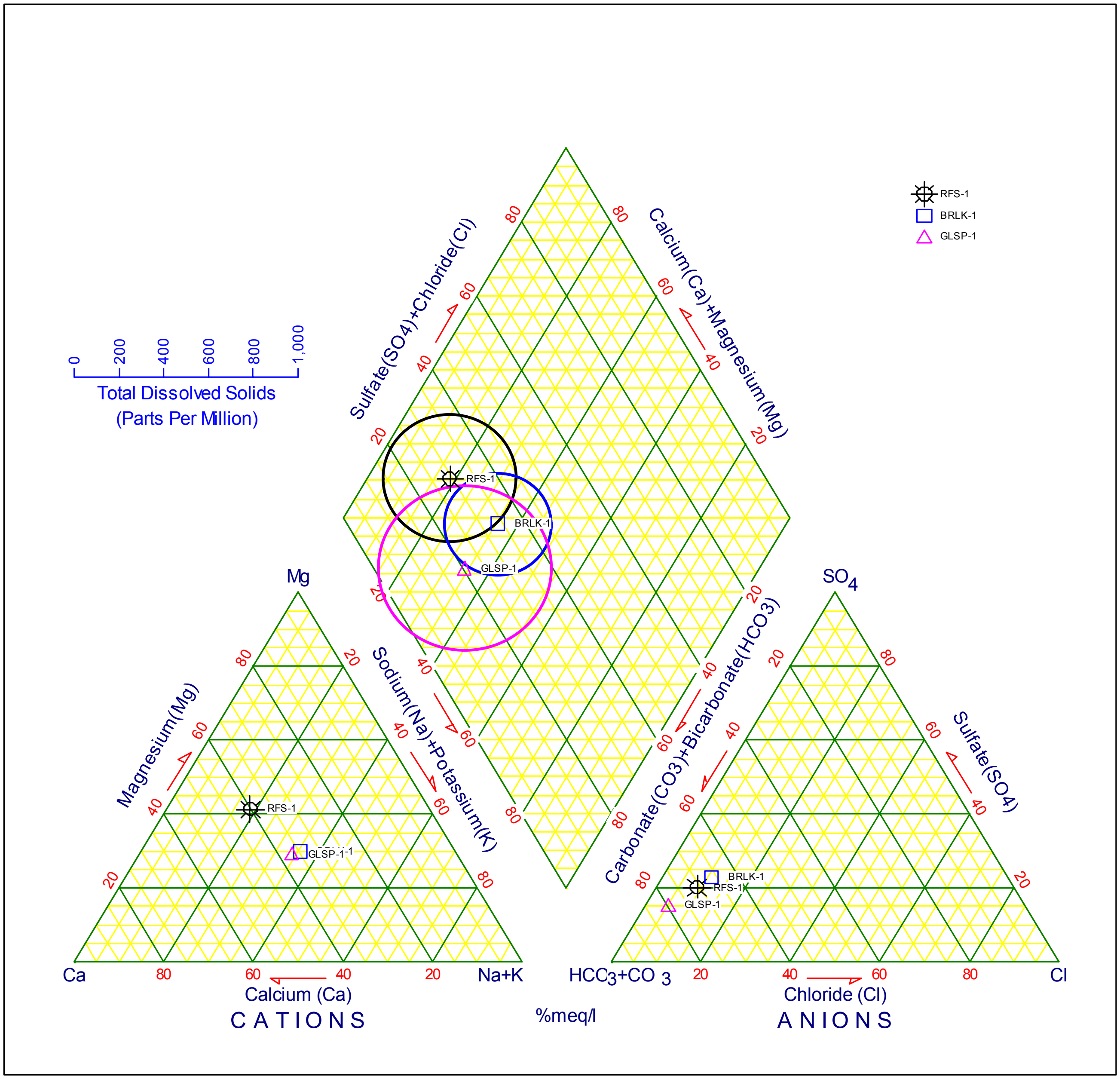


Figure A-5
Piper Diagram Analysis
Rocky Ford Springs Source Evaluation
July 2001
(Soap Lake data removed)

Note - The radius of the circles in the diamond-shaped field surrounding each station symbol represent the station concentration for total dissolved solids (TDS)

Trilinear diagram analysis is normally best used to distinguish water sources when coupled with other hydrogeologic data. Figure A-4 indicates that the geochemical profile and TDS concentration of the Soap Lake station (SPLK-1) is quite distinct from the remaining stations, consistent with other lines of evidence indicating that the lake is not a source of the water to the stations to the south. The data also indicate that from a hydrochemical standpoint it is reasonable to consider Brook Lake as a potential source of water for Rocky Ford Springs. The general similarity of the hydrochemical profile between Rocky Ford Springs and the Gloyd Seeps sample, however, suggests that Brook Lake is not a unique water type match to the springs.

For further comparison, Stiff diagrams were developed for the RFS-1, BRLK-1, and GLSP-1 stations (Figure A-6). Stiff diagrams provide an alternative method to the Piper diagram for viewing the relative ionic character of a water sample. For context in the interpretation of the diagrams, a comparison was made to the data from the 1991 Golder Associates, Inc. study of the City of Moses Lake Larson well field (GAI, 1991).

When compared to the GAI data, the samples from the BRLK-1 and GLSP-1 stations best match the “mixed” or “intermediate” water type identified by Golder; i.e. the water exhibits an enrichment of the sodium vs. the magnesium. This water type is normally observed in samples that have experienced prior contact with both basalts and unconsolidated material. Turney (1986) reported that groundwater in the Columbia Basin is progressively enriched in sodium, and loses calcium and magnesium to precipitation as a function of increased residence time in the basalt aquifers (see also Whiteman et al, 1994).

In contrast, the RFS-1 station diagram best matches the Stiff diagrams presented by Golder for a “shallow” water type, i.e. water that has primarily been in contact with the unconsolidated material (and has had a shorter residence time in the aquifer). This is expressed as an enrichment of magnesium vs. sodium on the diagram. The spring water does not show a hydrochemical profile indicating a long residence time in the aquifer, or extended contact with basalts. These data favor the interpretation that the bulk of the spring water is derived from recharge occurring relatively close to the springs, possibly from the irrigated agricultural fields down gradient of Brook Lake. Additional study would be required to determine if this recharge is also the source for the phosphorus observed in the springs.

Summary

A review of existing published information was conducted to evaluate the hydrogeologic setting surrounding the Rocky Ford Springs, in order to determine if a likely source area for the Rocky Ford Springs could be identified, or alternatively, ruled out. In support of this review, a limited hydrochemical sampling effort was undertaken. Sampling included the collection and analysis of a water sample from the springs, as well as two additional samples from area lakes that have been suggested as potential source reservoirs for the spring discharge. An additional water sample was collected and analyzed from the Gloyd Seeps area for comparative purposes. Water samples were analyzed for major ionic components, and the data results were plotted on trilinear and stiff diagrams to provide an evaluation of the hydrochemical facies of the different sources.

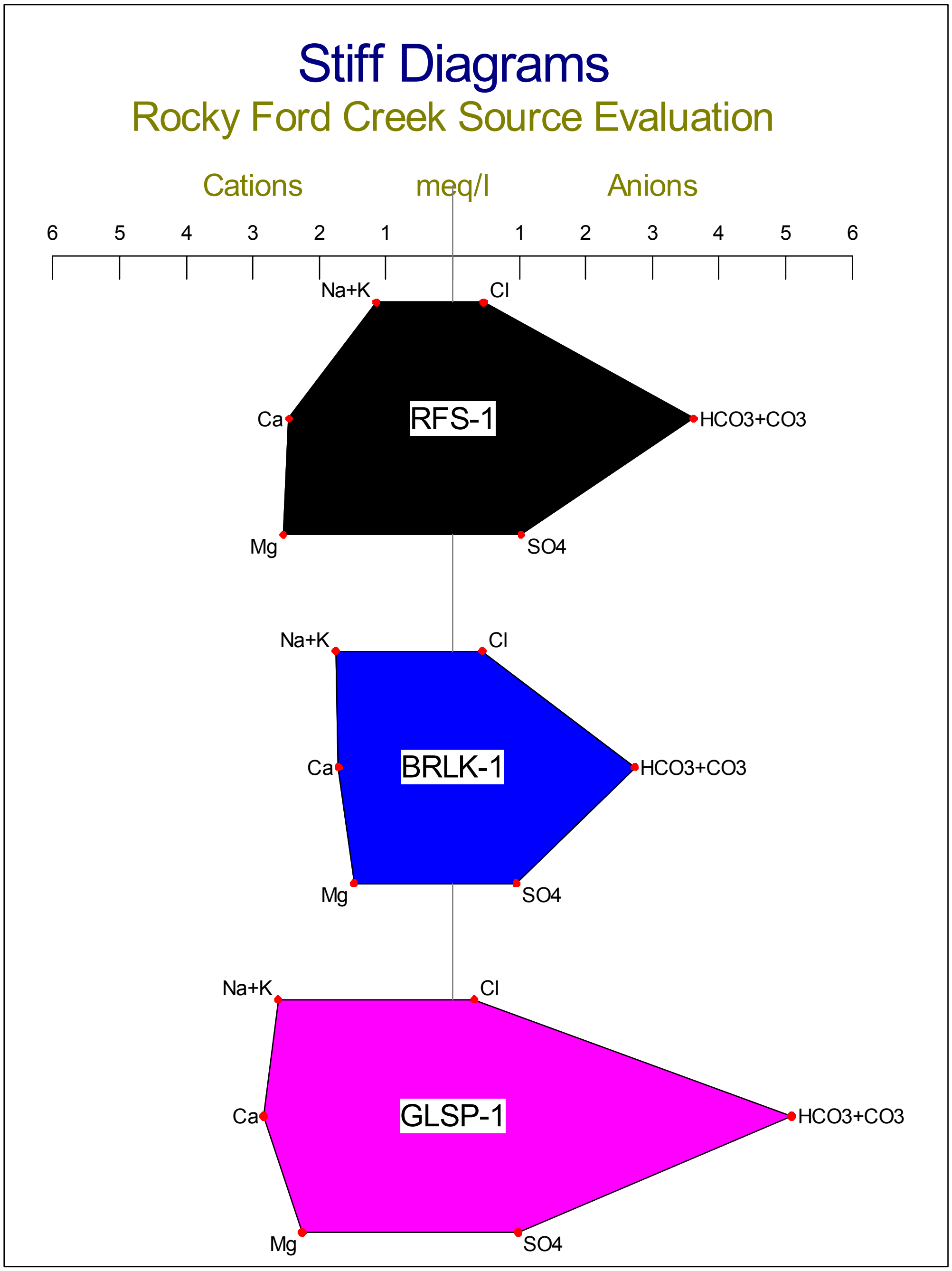


Figure A-6
Stiff Diagram Analysis
Rocky Ford Springs Source Evaluation
July 2001
(Soap Lake data removed)

The results of this evaluation confirm that Soap Lake is not a likely origin for the spring discharge occurring at the Troutlodge hatchery. The existing data regarding groundwater flow patterns in the area of interest indicate that the spring discharge is hydraulically connected to shallow groundwater located to the northeast of the springs. The spring water's ionic composition best matches groundwater samples with a comparatively short residence time in the shallow unconsolidated deposits of the aquifer system (with limited long-term contact with basalts).

In contrast, the samples from Brook Lake and Gloyd Seeps exhibit an enrichment in sodium (typical of water that has had prior contact with basalts) that is not observed in the spring water sample. These data favor the interpretation that the origin for the bulk of the spring water is the irrigation recharge to the unconsolidated deposits present downgradient of Brook Lake. Additional groundwater sampling and study would be required to confirm if this recharge is also the source for the phosphorus observed at the springs.

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Appendix B

Sampling Procedures

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Table B-1 - Physical Description and Location of Sample Stations

Station name	Station type	Location description	Dominant upgradient land uses	Depth of installation below sediment surface (ft)	Screen length (ft)	Depth of screen mid-pt. below sediment surface (ft)	Distance from shore (ft)	Dye test?	Comments
MLGW-01	Mini-piezo	Near mouth of Rocky Ford Crk.	undeveloped wetland/rangeland	3.0	1	2.5	10	N	
MLGW-02	Mini-piezo	W shore of upper main arm	rangeland/shrubland	6.0	1	5.5	39	N	piezometer flowing
MLGW-03	Seep	W shore of upper main arm	steep bluff to rangeland/shrubland	N/A	N/A	N/A	N/A	N/A	Seep just above water line of lake, ~1-2 gpm
MLGW-04S	Mini-piezo	NE shore of upper main arm	rural development/rangeland/irrigated cropland	3.0	1	2.5	26	Y	
MLGW-04D	Mini-piezo	NE shore of upper main arm	rural development/rangeland/irrigated cropland	5.0	1	4.5	25	Y	Approx. 1 foot E. of MLGW-04S
MLGW-05	Mini-piezo	N shore of main arm	rural development/rangeland/irrigated cropland	3.0	1	2.5	35	Y	
MLGW-06	Mini-piezo	NE shore of main arm	suburban development/industrial/commercial	4.0	1	3.5	23	Y	
MLGW-07	Mini-piezo	E shore of main arm	irrigated cropland/suburban development	3.5	1	3.0	15	Y	
MLGW-08	Mini-piezo	Cascade Park on Lewis Horn	city park/campground/public restroom	4.6	1	4.1	27	Y	
MLGW-09	Mini-piezo	NW shore of upper Parker Horn	urban residential	5.0	1	4.5	32	Y	
MLGW-10	Mini-piezo	SE shore of upper Parker Horn	urban park/urban commercial	5.0	1	4.5	18	N	
MLGW-11	Mini-piezo	NW shore of upper Pelican Horn	urban residential	5.2	1	4.7	36	Y	
MLGW-12	Mini-piezo	SE shore of upper Pelican Horn	urban park/suburban residential	5.0	1	4.5	20	Y	
MLGW-13	Mini-piezo	NW shore of lower Pelican Horn	boat launch/suburban development/orchard	3.4	1	2.9	24	Y	
MLGW-14	Spring	SE shore of lower Pelican Horn	suburban development/irrigated cropland	N/A	N/A	N/A	N/A	N/A	Spring flow from collector pipe - ~4-5 gpm
MLGW-15	Well	E shore of upper main arm	rural development/rangeland/shrubland	N/A	N/A	N/A	900	N/A	Total depth = 140 feet bgs

N/A - Not applicable

Table B-2 - Parameters, Container Types, Holding Times, and Sample Preservation

Parameter	Container	Holding Time	Preservation
Total Dissolved Solids	1 L w/m polyethylene	7 days	Chill to 4^oC
Chloride	1 L w/m polyethylene	28 days	Chill to 4^oC
Ammonia-N	125 ml w/m clear Nalgene	28 days	Adjust pH to <2 w/ H_2SO_4 and chill to 4^oC
Nitrate-N	125 ml w/m clear Nalgene	28 days	Adjust pH to <2 w/ H_2SO_4 and chill to 4^oC
Total Persulfate Nitrogen	125 ml w/m clear Nalgene	28 days	Adjust pH to <2 w/ H_2SO_4 and chill to 4^oC
Total Kjeldahl Nitrogen	125 ml w/m clear Nalgene	28 days	Adjust pH to <2 w/ H_2SO_4 and chill to 4^oC
Total Phosphorus/Total Dissolved Phosphorus	125 ml w/m clear Nalgene	28 days	Adjust pH to <2 w/ H_2SO_4 and chill to 4^oC
Orthophosphate	125 ml w/m amber Nalgene	48 hours	Chill to 4^oC
Total Dissolved Iron	125 ml w/m clear Nalgene	6 months	Adjust pH to <2 w/ HNO_3 and chill to 4^oC
Total Dissolved Manganese	125 ml w/m clear Nalgene	6 months	Adjust pH to <2 w/ HNO_3 and chill to 4^oC

Table B-3 - Parameters, Test Methods, and Practical Quantitation Limits

Parameter	Test Method EPA Method[a]/Standard Methods[b]	Practical Quantitation *L* imit
Field pH	WTW Field Probe	0.1 Std. unit
Field temperature	WTW Field Probe	0.1°C
Field specific conductance	WTW Field Probe	10 uS/cm
Field dissolved oxygen	WTW Field Probe	0.1 mg/L
Total Dissolved Solids	EPA 160.1	1 mg/L
Chloride	EPA 300.0	0.1 mg/L
Ammonia-N	SM4500NH3H	0.010 mg/L
Nitrate-N	SM4500NO3I	0.010 mg/L
Total Persulfate Nitrogen	SM 4500NB	0.010 mg/L
Total Kjeldahl Nitrogen	SM4500NORD	0.500 mg/L
Total Phosphorus/Total Dissolved Phosphorus	SM4500PI (May, July)/SM4500PH (Oct.)	0.010 mg/L
Orthophosphate[c]	EPA 365.3M (May)/SM4500PG (July, Oct.)	0.005 mg/L
Total Dissolved Fe	EPA 200.7	0.020 mg/L
Total Dissolved Mn	EPA 200.7	0.002 mg/L (July), 0.001 mg/L (Oct.)

[a] USEPA, 1983, Methods for Chemical Analysis of Water and Wastes, EPA-600/4-79-020.

[b] American Public Health Association, 1995, Methods for the Examination of Water and Wastewater, 19th Edition.

[c] The methods used for analysis of orthophosphate actually measure the dissolved reactive phosphorus (DRP) present in a water sample, which may include more than orthophosphate. It is presumed that orthophosphate is the predominant species of phosphorus in groundwater; in this report the terms DRP and orthophosphate are considered interchangeable.

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Appendix C

Field Water Quality Data

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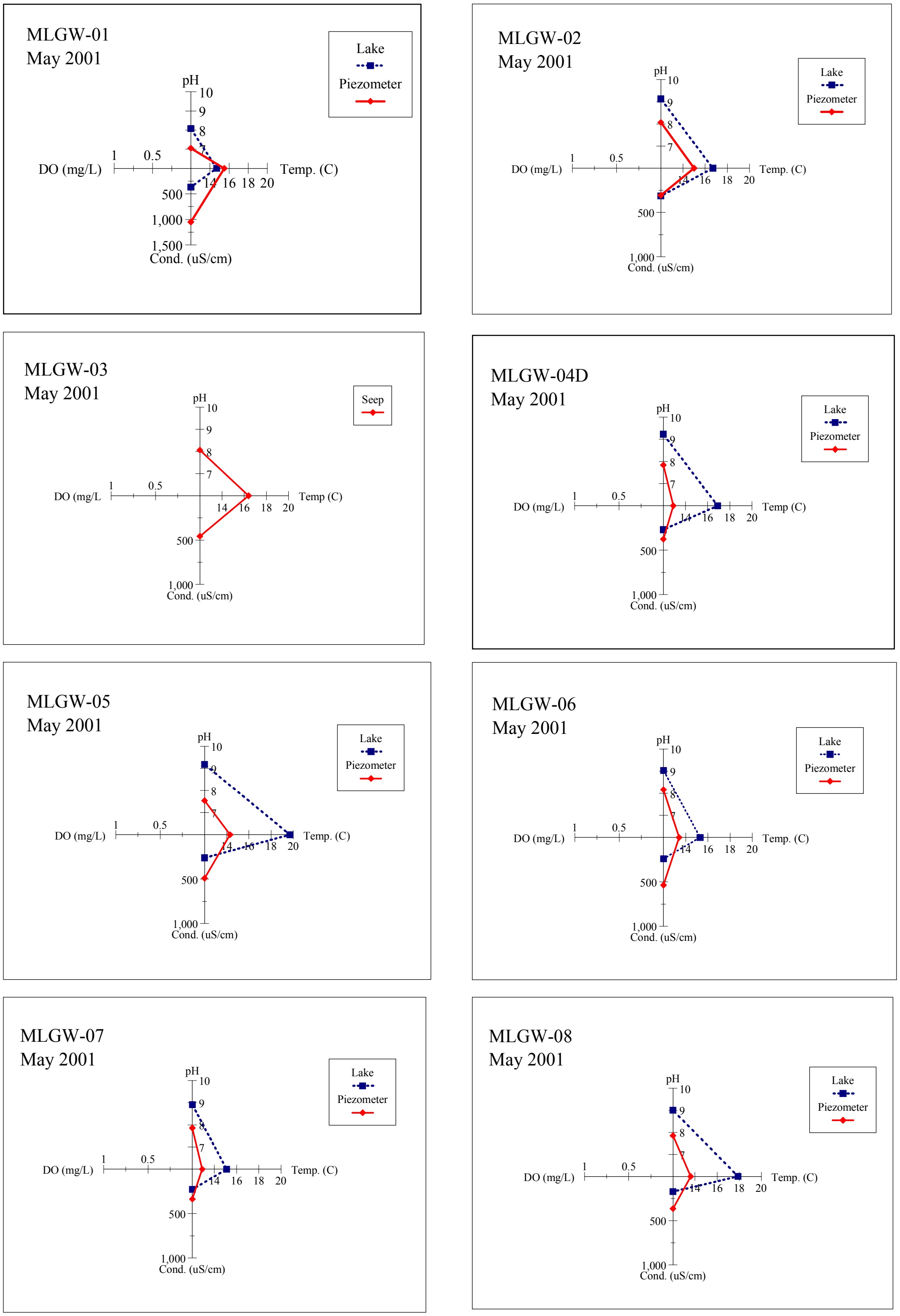


Figure C-1 – Comparison of Lake vs. Groundwater Field Water Quality Parameters

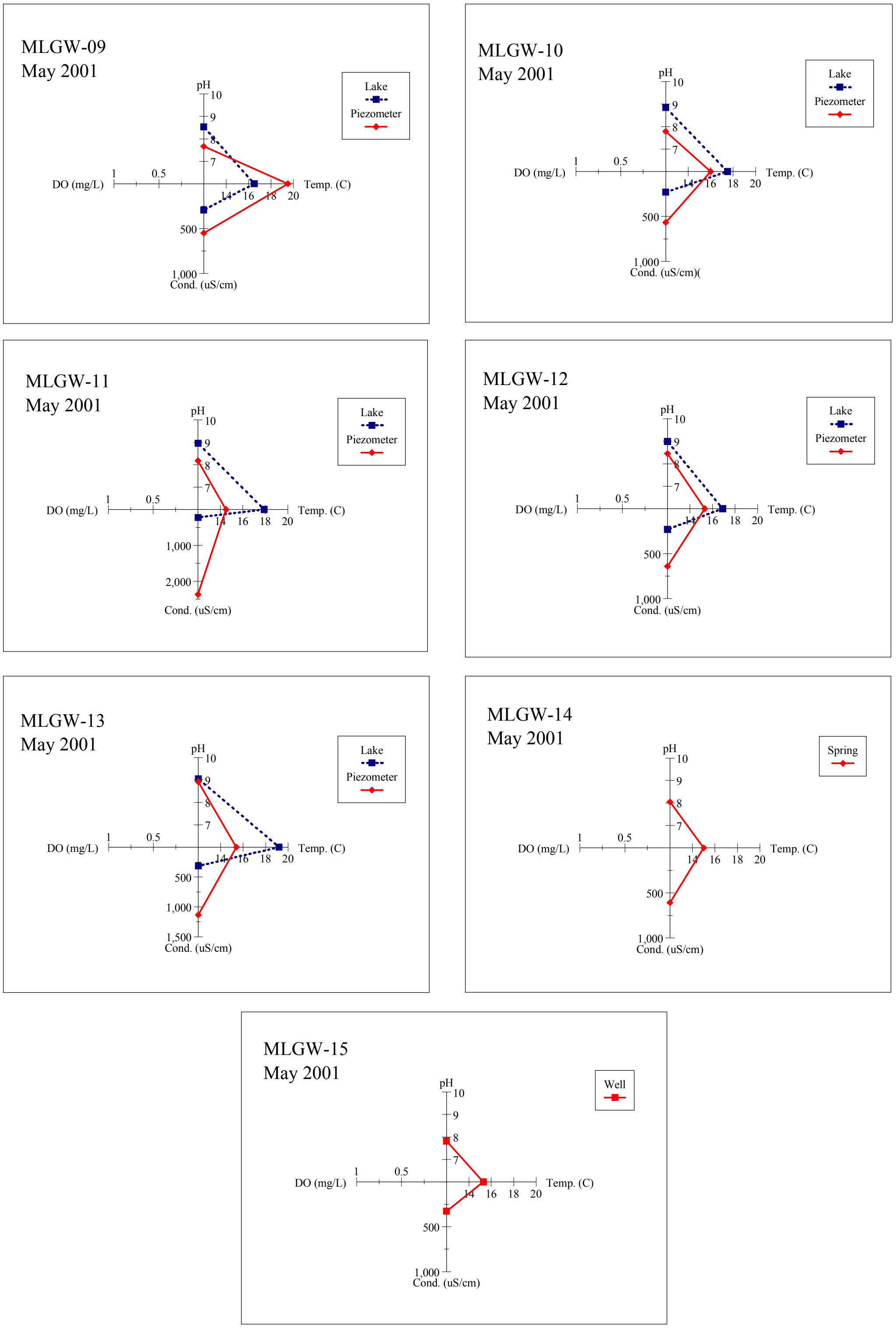


Figure C-1 (page 2)

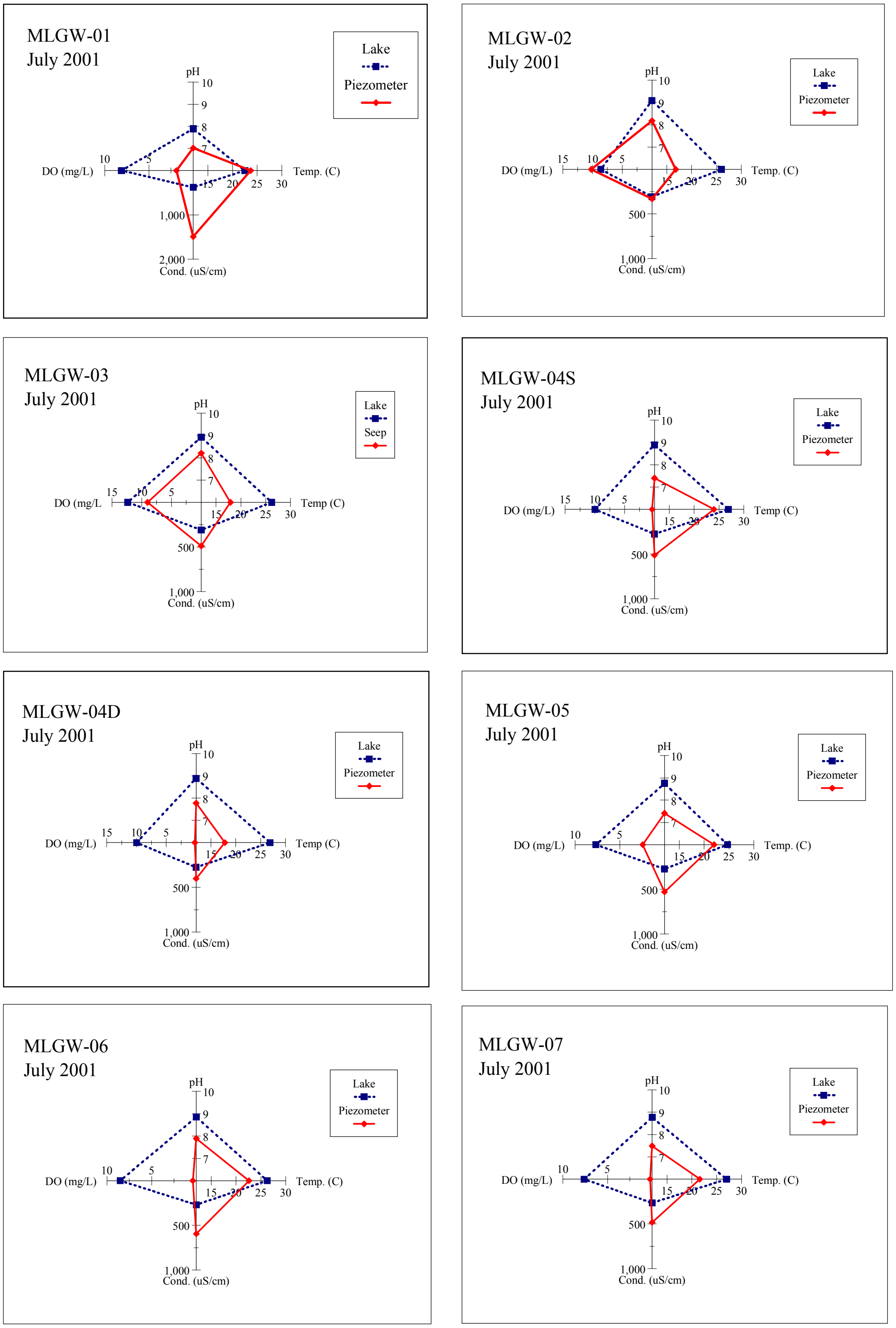
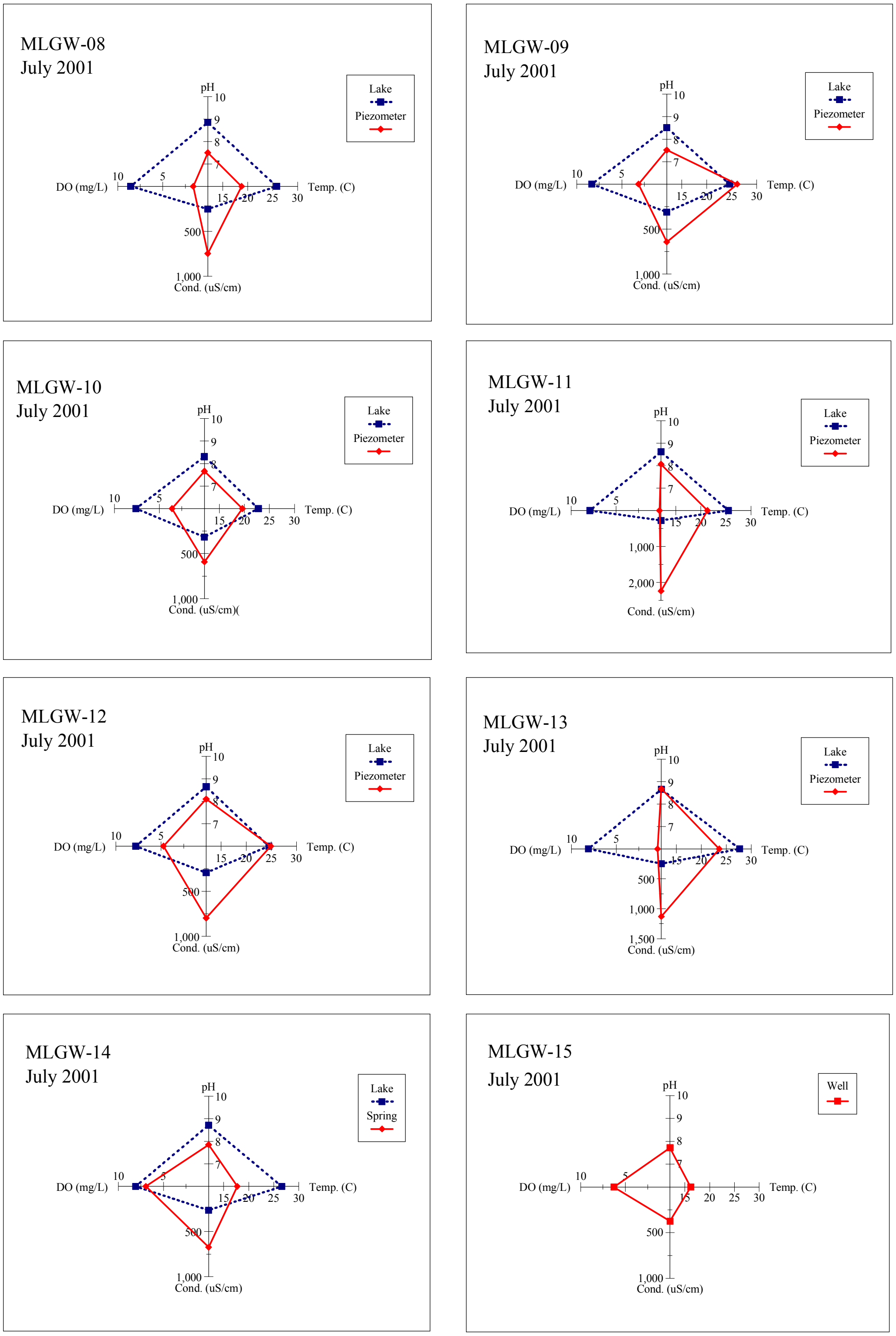


Figure C-1 (page 3)



MLGW-08
July 2001

MLGW-09
July 2001

MLGW-10
July 2001

MLGW-11
July 2001

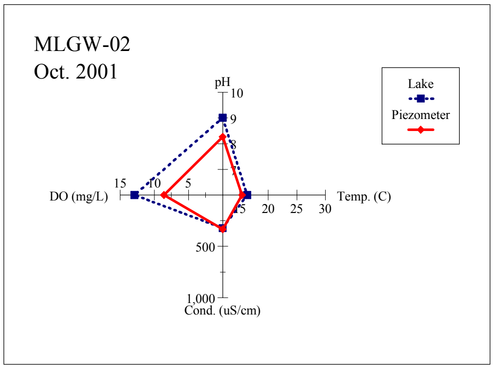
MLGW-12
July 2001

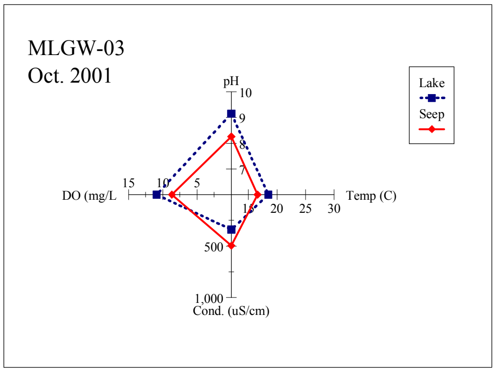
MLGW-13
July 2001

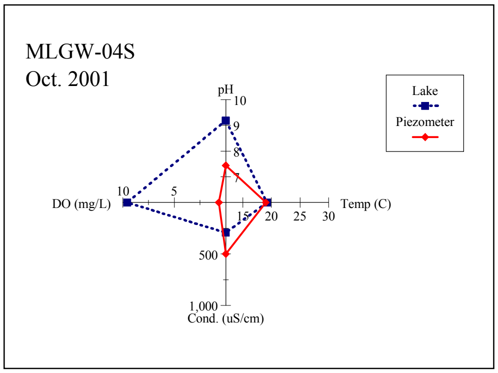
MLGW-14
July 2001

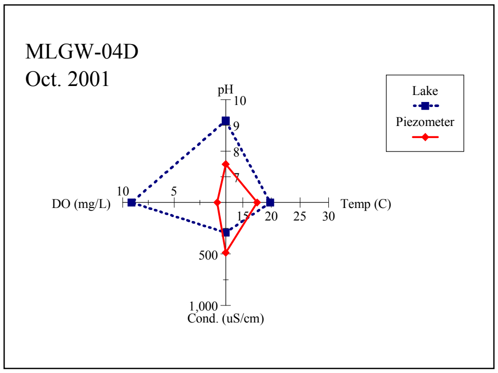
MLGW-15
July 2001

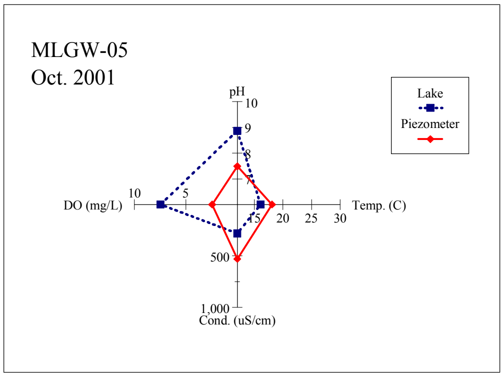
Figure C-1 (page 4)

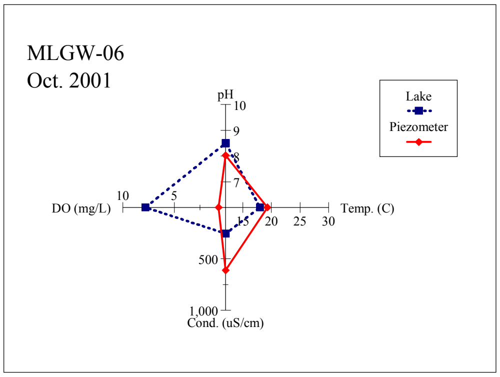


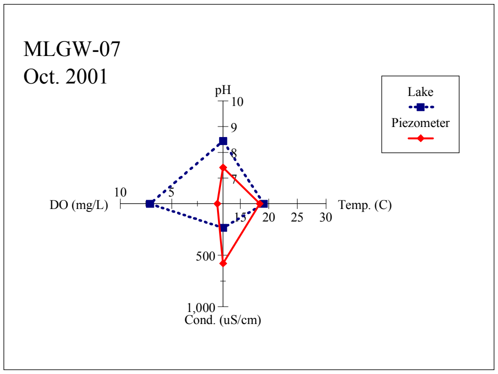


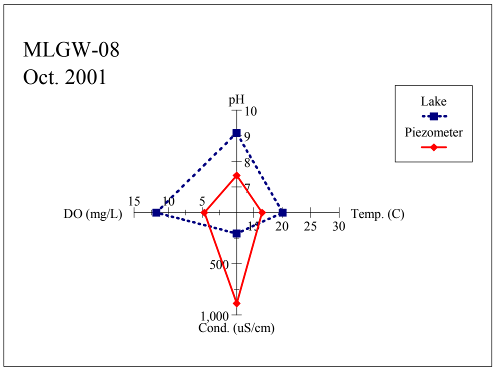












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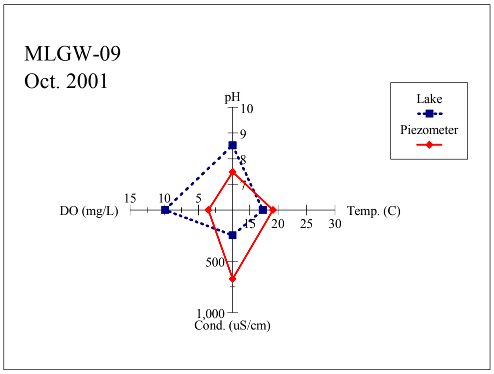

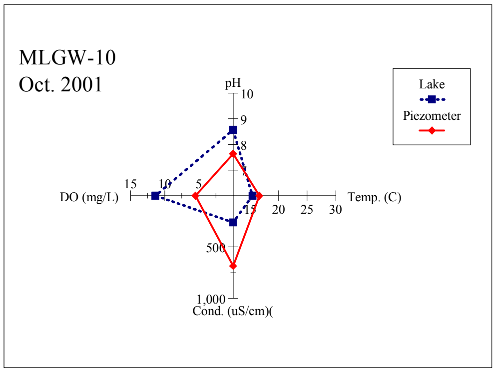

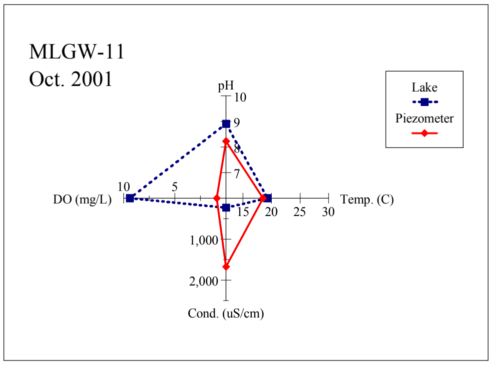

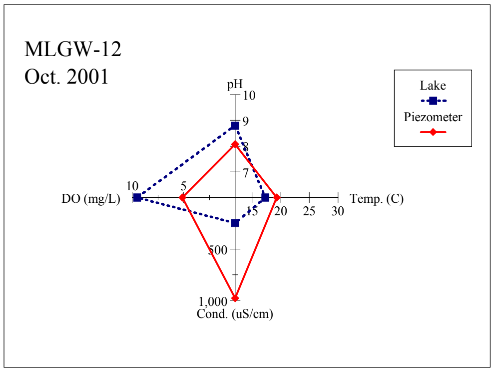

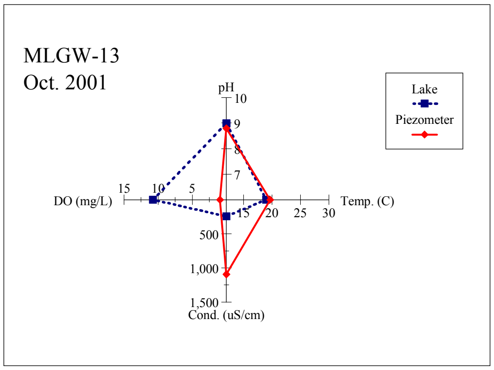

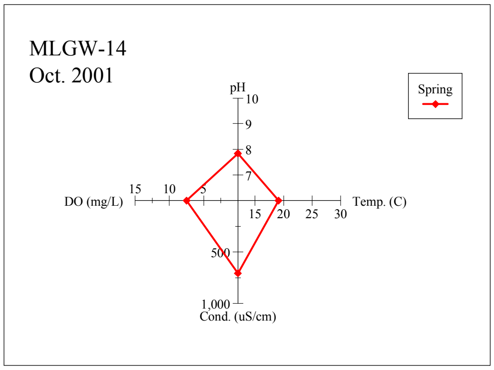

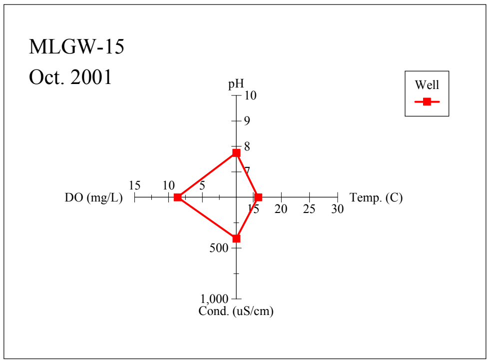

Figure C-1 (page 6)

Table C-1 - Field Water Quality Data

Lake Conditions May 2001							Groundwater Conditions May 2001						
Station	Date	Time	pH	Temp (°C)	Cond. (uS/cm)	DO (mg/L)	Station	Date	Time	pH	Temp (°C)	Cond. (uS/cm)	DO (mg/L)
MLLW-01	5/8/01	915	8.08	14.7	365	NM	MLGW-01	5/8/01	930	7.05	15.5	1050	NM
MLLW-02	5/8/01	1145	9.13	16.7	313	NM	MLGW-02	5/8/01	1200	8.07	15.0	306	NM
MLLW-03	NM	NM	NM	NM	NM	NM	MLGW-03	5/8/01	1245	8.06	16.4	456	NM
MLLW-04S	NM	NM	NM	NM	NM	NM	MLGW-04S	NM	NM	NM	NM	NM	NM
MLLW-04D	5/8/01	1310	9.23	16.9	269	NM	MLGW-04D	5/8/01	1330	7.84	12.9	376	NM
MLLW-05	5/8/01	1410	9.18	19.7	260	NM	MLGW-05	5/8/01	1430	7.54	14.3	490	NM
MLLW-06	5/9/01	935	9.03	15.3	239	NM	MLGW-06	5/9/01	945	8.16	13.4	537	NM
MLLW-07	5/9/01	1100	8.91	15.1	227	NM	MLGW-07	5/9/01	1130	7.86	12.9	336	NM
MLLW-08	5/10/01	1225	9.00	17.9	172	NM	MLGW-08	5/10/01	1300	7.85	13.6	365	NM
MLLW-09	5/9/01	1345	8.54	16.5	291	NM	MLGW-09	5/9/01	1415	7.67	19.5	548	NM
MLLW-10	5/9/01	1245	8.85	17.5	230	NM	MLGW-10	5/9/01	1300	7.99	16.0	566	NM
MLLW-11	5/10/01	1020	8.96	17.9	219	NM	MLGW-11	5/10/01	1030	8.18	14.5	2370	NM
MLLW-12	5/10/01	910	8.99	16.9	230	NM	MLGW-12	5/10/01	945	8.45	15.3	640	NM
MLLW-13	5/10/01	1120	9.05	19.2	315	NM	MLGW-13	5/10/01	1145	8.92	15.4	1134	NM
MLLW-14	NM	NM	NM	NM	NM	NM	MLGW-14	5/10/01	1045	8.04	15.0	607	NM
MLLW-15	NA	NA	NA	NA	NA	NA	MLGW-15	5/9/01	1530	7.82	15.3	327	NA

NM - Not measured
NA - Not applicable

Table C-1 (page 2) - Field Water Quality Data
Moses Lake TMDL Groundwater Study

Lake Conditions July 2001							Groundwater Conditions July 2001						
Station	Date	Time	pH	Temp (^{o}C)	Cond. (uS/cm)	DO (mg/L)	Station	Date	Time	pH	Temp (^{o}C)	Cond. (uS/cm)	DO (mg/L)
MLLW-01	7/23/01	1000	7.89	22.5	378	8.10	MLGW-01	7/23/01	1045	7.02	23.7	1493	1.89
MLLW-02	7/23/01	1230	9.09	26.0	307	8.62	MLGW-02	7/23/01	1245	8.18	16.8	330	10.2
MLLW-03	7/23/01	1315	8.92	26.2	309	12.32	MLGW-03	7/23/01	1320	8.24	17.9	488	9.04
MLLW-04S	7/23/01	1400	8.89	26.9	275	10.00	MLGW-04S	7/23/01	1500	7.40	24.0	513	0.39
MLLW-04D	7/23/01	"	"	"	"	"	MLGW-04D	7/23/01	1430	7.77	17.8	402	0.21
MLLW-05	7/26/01	1000	8.76	24.7	271	7.67	MLGW-05	7/26/01	1045	7.41	22.0	528	2.43
MLLW-06	7/26/01	1130	8.85	26.3	269	8.52	MLGW-06	7/26/01	1200	7.89	22.6	594	0.39
MLLW-07	7/26/01	1335	8.77	27.0	265	7.60	MLGW-07	7/26/01	1400	7.49	21.6	482	0.23
MLLW-08	7/24/01	1010	8.85	25.8	253	8.57	MLGW-08	7/24/01	1030	7.50	18.8	748	1.62
MLLW-09	7/25/01	1410	8.51	24.6	310	8.34	MLGW-09	7/25/01	1430	7.52	26.1	641	3.14
MLLW-10	7/24/01	725	8.31	22.8	316	7.60	MLGW-10	7/24/01	745	7.66	19.6	591	3.60
MLLW-11	7/25/01	1140	8.62	25.5	273	7.90	MLGW-11	7/25/01	1215	8.07	21.3	2240	0.17
MLLW-12	7/25/01	1010	8.64	24.5	294	7.80	MLGW-12	7/25/01	1100	8.09	24.9	797	4.73
MLLW-13	7/24/01	1410	8.65	27.7	247	8.11	MLGW-13	7/24/01	1445	8.68	23.6	1129	0.42
MLLW-14	7/25/01	1250	8.71	26.6	264	8.09	MLGW-14	7/25/01	1300	7.85	17.7	675	6.95
MLLW-15	NA	NA	NA	NA	NA	NA	MLGW-15	7/24/01	1300	7.72	16.2	374	6.28

NM - Not measured
NA - Not applicable

Table C-1 (page 3) - Field Water Quality Data
Moses Lake TMDL Groundwater Study

Lake Conditions October 2001							Groundwater Conditions October 2001						
Station	Date	Time	pH	Temp (^{o}C)	Cond. (uS/cm)	DO (mg/L)	Station	Date	Time	pH	Temp (^{o}C)	Cond. (uS/cm)	DO (mg/L)
MLLW-01	NM	NM	NM	NM	NM	NM	MLGW-01	NM	NM	NM	NM	NM	NM
MLLW-02	10/1/01	1055	9.01	16.3	322	12.89	MLGW-02	10/01/01	1100	8.27	15.4	333	8.59
MLLW-03	10/1/01	1230	9.15	18.5	339	10.89	MLGW-03	10/01/01	1245	8.26	16.6	495	8.67
MLLW-04S	10/1/01	1310	9.18	19.8	292	9.15	MLGW-04S	10/01/01	1345	7.44	19.0	499	0.67
MLLW-04D	10/1/01	"	"	"	"	"	MLGW-04D	10/01/01	1430	7.49	17.5	487	0.81
MLLW-05	10/2/01	845	8.87	16.1	280	7.46	MLGW-05	10/02/01	915	7.49	18.1	528	2.44
MLLW-06	10/2/01	1000	8.50	18.0	255	7.80	MLGW-06	10/02/01	1030	8.02	19.3	612	0.68
MLLW-07	10/2/01	1110	8.44	19.1	234	7.10	MLGW-07	10/02/01	1145	7.41	18.4	579	0.56
MLLW-08	10/2/01	1410	9.11	20.1	204	11.76	MLGW-08	10/02/01	1445	7.45	16.5	886	4.71
MLLW-09	10/2/01	1310	8.53	17.3	246	9.90	MLGW-09	10/02/01	1340	7.49	19.1	671	3.56
MLLW-10	10/3/01	850	8.57	15.4	260	11.35	MLGW-10	10/03/01	930	7.64	16.6	685	5.51
MLLW-11	10/3/01	1230	8.91	19.3	231	9.39	MLGW-11	10/03/01	1315	8.23	18.5	1672	0.9
MLLW-12	10/3/01	1030	8.80	17.3	248	9.52	MLGW-12	10/03/01	1115	8.07	19.3	978	5.12
MLLW-13	10/3/01	1345	8.99	19.0	244	10.76	MLGW-13	10/03/01	1415	8.79	19.7	1093	0.95
MLLW-14	NM	NM	NM	NM	NM	NM	MLGW-14	10/03/01	1330	7.84	19.1	707	7.48
MLLW-15	NA	NA	NA	NA	NA	NA	MLGW-15	10/03/01	1800	7.75	15.9	404	8.66

NM - Not measured
NA - Not applicable

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Appendix D

Project Quality Assurance

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Analytical Quality Assurance – Field

Quality assurance tests used to evaluate the bias introduced into the project analytical results by the various filtering procedures and equipment used during this project included:

Filter blanks – Clean laboratory supplied de-ionized (DI) water was pumped through the sample collection and filtering apparatus between sample stations once per sampling round to determine if there was cross contamination between stations. This procedure was also used to determine if the filters contributed any positive bias to the sample results for filtered samples. The results (Table D-1) indicate that no significant bias or cross contamination by the sampling and filtration apparatus was observed during the course of the project.

Table D-1 – Filter Blank Data

	May[(a)]			July[(b)]			October[(b)]		
Analyte	Value	Units	Qualifier	Value	Units	Qualifier	Value	Units	Qualifier
OP	0.005	mg/L	U	0.005	mg/L	U	0.003	mg/L	U
TP	-	-	-	0.010	mg/L	U	0.010	mg/L	UJ
NH3	-	-	-	0.010	mg/L	U	0.010	mg/L	UJ
NO3+NO2	-	-	-	0.013	mg/L		0.01	mg/L	U
TPN	-	-	-	0.019	mg/L		0.01	mg/L	UJ
Iron	-	-	-	0.02	mg/L	U	0.02	mg/L	U
Manganese	-	-	-	0.002	mg/L	U	0.001	mg/L	U

"-" - not measured
U - not detected at or above the reporting limit
J - result is an estimate
(a) - syringe filtered
(b) - capsule filtered

Filter split comparison – To test the influence of filter type on the analytical results for OP, several split samples were collected and alternately filtered using a syringe filter and an in-line capsule filter. This test was used to determine if the results for OP samples filtered during the May round using a syringe filter could be compared to later OP samples filtered using a capsule filter. The results (Table D-2) indicate no significant difference in the concentrations reported for the target analytes as a function of filter type, suggesting the OP results from the different rounds can be directly compared.

Table D-2 – Filter Comparison Data

Station	Analyte	Reported Concentration Syringe Filtered mg/L	Reported Concentration Capsule Filtered mg/L
MLGW-10	OP	0.171	0.169
MLGW-11	OP	1.16	1.15

Piezometer equipment blanks – A new piezometer was immersed into clean laboratory supplied DI water for 15 minutes. A sample of the DI water was then pumped into a randomly selected clean sample bottle and preserved to determine if the piezometer materials were contributing a positive bias to the sampling results for any of the target analytes. The results (Table D-3) indicate that no significant positive bias or contamination of the target analytes was introduced into the study results by the piezometer materials.

Table D-3 – Piezometer Equipment Blank Data

	May[(a)]			July			October[(b)]		
Analyte	Value	Units	Qualifier	Value	Units	Qualifier	Value	Units	Qualifier
TP	0.010	mg/L	U	-	-	-	-	-	-
NH3	0.010	mg/L	U	-	-	-	-	-	-
NO3+NO2	0.010	mg/L	U	-	-	-	-	-	-
TPN	0.021	mg/L		-	-	-	-	-	-
TKN	0.500	mg/L	U	-	-	-	-	-	-
Iron	-	-	-	-	-	-	0.02	mg/L	U
Manganese	-	-	-	-	-	-	0.001	mg/L	U

"-" - not measured
U - not detected at or above the reporting limit
(a) - no filtration prior to analysis
(b) - filtered prior to analysis

Reference standard transfer sample (container blanks) – a laboratory-prepared TP reference solution (0.2 mg/L) was directly transferred in the field to a randomly selected clean sample container containing preservative to determine if there was phosphorus contamination introduced by the bottles or preservative used during the study. This sample was submitted as a blind sample to the laboratory. The relative percent difference (RPD) between the reference standard and the reported result for the transfer sample (0.217 mg/L) is 8%, less than the ±20% laboratory acceptance criteria. This indicates that no significant bias was introduced to the phosphorus results for this study by the project bottles or preservative.

Reference standard piezometer samples – a new, randomly selected piezometer was immersed into 1 liter of laboratory reference standard for TP (0.2 mg/L) for 15 minutes. The standard was then pumped through a filter into a clean sample bottle and preserved. This sample was used to determine if the piezometer materials caused a negative bias in the sampling results for phosphorus due to sorption of P onto the piezometer materials. The RPD between the reference standard and the piezometer sample (0.221 mg/L) is 10%, less than the ±20% laboratory acceptance criteria. This indicates that no significant negative bias was introduced by the sorption of P to the piezometer materials.

Analytical Quality Assurance – Laboratory

Precision and accuracy of laboratory results were estimated using the laboratory quality control testing run for each batch of 20 or fewer samples. Laboratory quality control testing consisted of duplicate blanks, duplicate samples, spiked samples, and control standards. Manchester Laboratory's quality control procedures are discussed in MEL (1988). Quality assurance reviews were completed for each round of sampling. All data are considered acceptable by the laboratory without qualification with the following exceptions:

- May 2001 OP concentration data for stations MLGW-06, -06 (dup), -07, -09, -10, and -15 were qualified as estimates due to the arrival of the samples at the laboratory above a temperature of 4°C.
- The May 2001 TKN and TPN results for station MLGW-01 were qualified as estimates due to a poor comparison of the results.
- The May 2001 TPN results for stations MLGW-06, and MLGW-11 were qualified as results due to spike recoveries outside the acceptable range.
- The July 2001 samples tested for ammonia-N from stations MLGW-02, -03, -04D, -04D (dup), -05, -06, -07, -08, -09, -10, -12, -14, and -15 were qualified as estimates due to matrix interference affects. The ammonia results for these samples may be biased low.
- The July 2001 OP concentration result for station MLGW-01 was qualified as an estimate due to the arrival of the sample at the laboratory several hours after the 48 hours allowable holding time.
- The October 2001 results for ammonia-N, TPN, and TP were qualified as estimates due to a refrigeration failure at the laboratory. The TP samples for this batch were manually digested prior to analysis due to laboratory equipment failure. Frozen splits of the samples were reanalyzed in December 2001 for TPN and TP. The December re-analysis results for the samples were consistently lower than the October values for TPN by an average of 8%, and were consistently higher than the October values for TP by an average of 8%.
- All October 2001 results for ammonia-N were also qualified as estimates due to contamination of a laboratory reagent.
- October 2001 results for TPN for stations MLGW-07 and MLGW-13 were qualified as estimates due to poor spike recoveries.
- Occasionally OP values reported by the laboratory were higher than the TP values for the same station. When the difference exceeded 20% the values were qualified as estimates.

Analytical Quality Assurance – Blind Field Duplicates

Blind field duplicate samples were submitted to the Manchester Laboratory during each sampling round. First round duplicate sample locations were selected randomly; duplicate locations selected for subsequent rounds were chosen on the basis of the first round data results. Field duplicates provide a measure of the overall sampling and analytical precision. Precision estimates are influenced not only by the random error introduced by collection and measurement procedures, but are also influenced by the natural variability of the concentrations in the media being sampled.

Table D-4 presents the reported concentration data for each of the duplicate pairs, and shows the % RSD calculated for each pair, grouped by parameter. Figure D-1 presents a chart for each of the target analytes showing the % RSD for each duplicate pair vs. the mean of the pair concentrations. These charts indicate that the % RSD for all of the parameters was normally below the target %RSD set in the project plan (Pitz, 2001) in the concentration ranges of interest. The mean % RSD for the key study parameters, OP and TDP, were 1.5% and 4.6%, respectively. Precision estimates for ammonia-N, nitrate-N, and TPN were occasionally above the project % RSD target values for samples with concentration ranges approaching the practical quantitation limit. As a general rule, precision estimates are less representative of random error as the measured values approach the detection limit.

Table D-4 - Field Duplicate Data

Round	Station	Value	Qual.	Units	Analyte	Abs. Diff. [D]	Std. Dev. s	Abs. Mean x	RSD% s/x*100
1	MLGW-06	21.1		mg/L	Chloride				
1	MLGW-06 Dup	21.3		mg/L	Chloride	0.20	0.141	21.2	0.67
1	MLGW-14	14.4		mg/L	Chloride				
1	MLGW-14 Dup	14.3		mg/L	Chloride	0.10	0.071	14.35	0.49
2	MLGW-04D	7.73		mg/L	Chloride				
2	MLGW-04D Dup	7.70		mg/L	Chloride	0.03	0.021	7.715	0.27
2	MLGW-11	46.6		mg/L	Chloride				
2	MLGW-11 Dup	47.1		mg/L	Chloride	0.50	0.354	46.85	0.75
3	MLGW-07	8.71		mg/L	Chloride				
3	MLGW-07 Dup	8.75		mg/L	Chloride	0.04	0.028	8.73	0.32
3	MLGW-13	12.4		mg/L	Chloride				
3	MLGW-13 Dup	12.4		mg/L	Chloride	0.00	0.000	12.4	0.00
1	MLGW-14	0.010	UNF	mg/L	NH3				
1	MLGW-14 Dup	0.010	UNF	mg/L	NH3	-	-	-	-
2	MLGW-04D	0.010	UJ	mg/L	NH3				
2	MLGW-04D Dup	0.010	UJ	mg/L	NH3	-	-	-	-
2	MLGW-11	0.090		mg/L	NH3				
2	MLGW-11 Dup	0.105		mg/L	NH3	0.02	0.011	0.0975	10.88
3	MLGW-07	0.023	J	mg/L	NH3				
3	MLGW-07 Dup	0.018	J	mg/L	NH3	0.01	0.004	0.0205	17.25
3	MLGW-13	0.038	J	mg/L	NH3				
3	MLGW-13 Dup	0.019	J	mg/L	NH3	0.02	0.013	0.0285	47.14
1	MLGW-06	4.70	NF	mg/L	NO2+NO3				
1	MLGW-06 Dup	5.12	NF	mg/L	NO2+NO3	0.42	0.297	4.91	6.05
1	MLGW-14	9.56	NF	mg/L	NO2+NO3				
1	MLGW-14 Dup	9.18	NF	mg/L	NO2+NO3	0.38	0.269	9.37	2.87
2	MLGW-04D	0.104		mg/L	NO2+NO3				
2	MLGW-04D Dup	0.079		mg/L	NO2+NO3	0.03	0.018	0.0915	19.32
2	MLGW-11	0.010	U	mg/L	NO2+NO3				
2	MLGW-11 Dup	0.010	U	mg/L	NO2+NO3	-	-	-	-
3	MLGW-07	0.112		mg/L	NO2+NO3				
3	MLGW-07 Dup	0.112		mg/L	NO2+NO3	0.00	0.000	0.112	0.00
3	MLGW-13	0.010	U	mg/L	NO2+NO3				
3	MLGW-13 Dup	0.010	U	mg/L	NO2+NO3	-	-	-	-

U - Not detected at or above the reporting limit

J - Reported result is an estimate

NF - Sample not filtered, reported value may overestimate true dissolved concentration

Table D-4 (page 2)

Round	Station	Value	Qual.	Units	Analyte	Abs. Diff. [D]	Std. Dev. s	Abs. Mean x	RSD% s/x*100
1	MLGW-06	0.116	J	mg/L	OP				
1	MLGW-06 Dup	0.118	J	mg/L	OP	0.00	0.001	0.117	1.21
1	MLGW-14	0.029		mg/L	OP				
1	MLGW-14 Dup	0.029		mg/L	OP	0.00	0.000	0.029	0.00
2	MLGW-04D	0.062		mg/L	OP				
2	MLGW-04D Dup	0.065		mg/L	OP	0.00	0.002	0.0635	3.34
2	MLGW-11	1.15		mg/L	OP				
2	MLGW-11 Dup	1.17		mg/L	OP	0.02	0.014	1.16	1.22
3	MLGW-07	0.157		mg/L	OP				
3	MLGW-07 Dup	0.157		mg/L	OP	0.00	0.000	0.157	0.00
3	MLGW-13	0.249		mg/L	OP				
3	MLGW-13 Dup	0.237		mg/L	OP	0.01	0.008	0.243	3.49
1	MLGW-06	403		mg/L	TDS				
1	MLGW-06 Dup	405		mg/L	TDS	2.00	1.414	404	0.35
1	MLGW-14	458		mg/L	TDS				
1	MLGW-14 Dup	446		mg/L	TDS	12.00	8.485	452	1.88
2	MLGW-04D	267		mg/L	TDS				
2	MLGW-04D Dup	271		mg/L	TDS	4.00	2.828	269	1.05
2	MLGW-11	1590		mg/L	TDS				
2	MLGW-11 Dup	1590		mg/L	TDS	0.00	0.000	1590	0.00
3	MLGW-07	392		mg/L	TDS				
3	MLGW-07 Dup	387		mg/L	TDS	5.00	3.536	389.5	0.91
3	MLGW-13	695		mg/L	TDS				
3	MLGW-13 Dup	692		mg/L	TDS	3.00	2.121	693.5	0.31
1	MLGW-14	0.500	UNF	mg/L	TKN				
1	MLGW-14 Dup	0.500	UNF	mg/L	TKN	-	-	-	-
1	MLGW-14	0.042	NF	mg/L	TP				
1	MLGW-14 Dup	0.045	NF	mg/L	TP	0.00	0.002	0.0435	4.88
2	MLGW-04D	0.069		mg/L	TDP				
2	MLGW-04D Dup	0.074		mg/L	TDP	0.00	0.004	0.0715	4.94
2	MLGW-11	1.40		mg/L	TDP				
2	MLGW-11 Dup	1.21		mg/L	TDP	0.19	0.134	1.305	10.30
3	MLGW-07	0.142	J	mg/L	TDP				
3	MLGW-07 Dup	0.147	J	mg/L	TDP	0.01	0.004	0.1445	2.45
3	MLGW-13	0.233	J	mg/L	TDP				
3	MLGW-13 Dup	0.231	J	mg/L	TDP	0.00	0.001	0.232	0.61

U - Not detected at or above the reporting limit

J - Reported result is an estimate

NF - Sample not filtered, reported value may overestimate true dissolved concentration

Table D-4 (page 3)

Round	Station	Value	Qual.	Units	Analyte	Abs. Diff. [D]	Std. Dev. s	Abs. Mean x	RSD% s/x*100
1	MLGW-14	10.6	NF	mg/L	TPN				
1	MLGW-14 Dup	9.23	NF	mg/L	TPN	1.37	0.969	9.915	9.77
2	MLGW-04D	0.182		mg/L	TPN				
2	MLGW-04D Dup	0.129		mg/L	TPN	0.05	0.037	0.1555	24.10
2	MLGW-11	1.23		mg/L	TPN				
2	MLGW-11 Dup	0.828		mg/L	TPN	0.40	0.284	1.029	27.62
3	MLGW-07	0.346	J	mg/L	TPN				
3	MLGW-07 Dup	0.351	J	mg/L	TPN	0.01	0.004	0.3485	1.01
3	MLGW-13	0.255	J	mg/L	TPN				
3	MLGW-13 Dup	0.26	J	mg/L	TPN	0.01	0.004	0.2575	1.37
2	MLGW-04D	0.020	U	mg/L	Iron				
2	MLGW-04D Dup	0.020	U	mg/L	Iron	-	-	-	-
2	MLGW-11	0.089		mg/L	Iron				
2	MLGW-11 Dup	0.088		mg/L	Iron	0.00	0.001	0.0885	0.80
3	MLGW-07	0.062		mg/L	Iron				
3	MLGW-07 Dup	0.061		mg/L	Iron	0.00	0.001	0.0615	1.15
3	MLGW-13	0.020	U	mg/L	Iron				
3	MLGW-13 Dup	0.020	U	mg/L	Iron	-	-	-	-
2	MLGW-04D	0.010		mg/L	Manganese				
2	MLGW-04D Dup	0.012		mg/L	Manganese	0.00	0.001	0.011	12.86
2	MLGW-11	0.0287		mg/L	Manganese				
2	MLGW-11 Dup	0.0292		mg/L	Manganese	0.00	0.000	0.02895	1.22
3	MLGW-07	0.0575		mg/L	Manganese				
3	MLGW-07 Dup	0.0564		mg/L	Manganese	0.00	0.001	0.05695	1.37
3	MLGW-13	0.0207		mg/L	Manganese				
3	MLGW-13 Dup	0.0209		mg/L	Manganese	0.00	0.000	0.0208	0.68

U - Not detected at or above the reporting limit

J - Reported result is an estimate

NF - Sample not filtered, reported value may overestimate true dissolved concentration

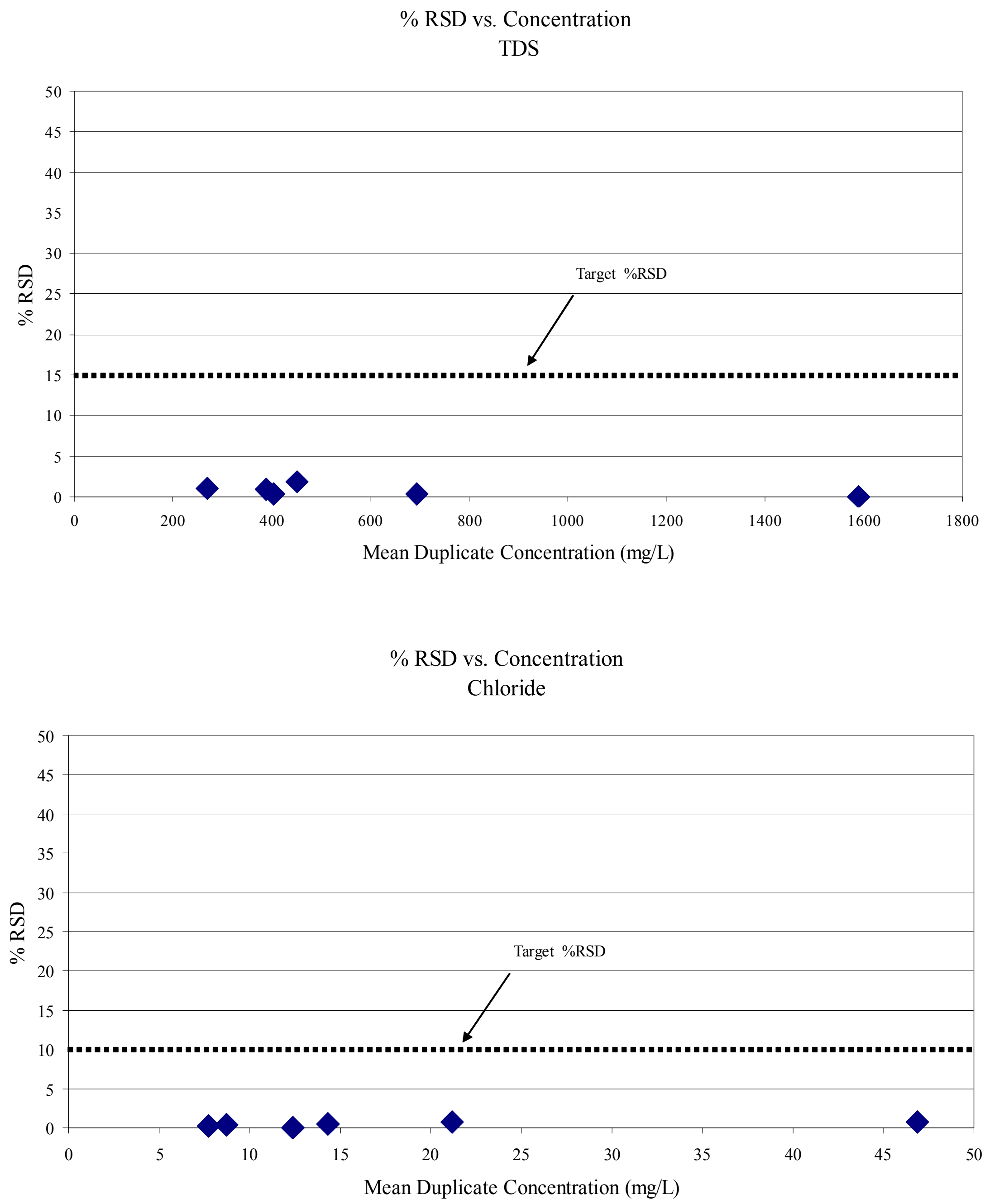
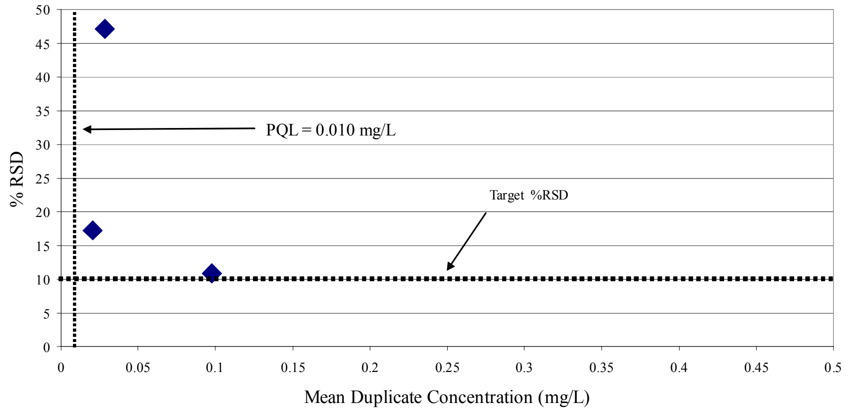
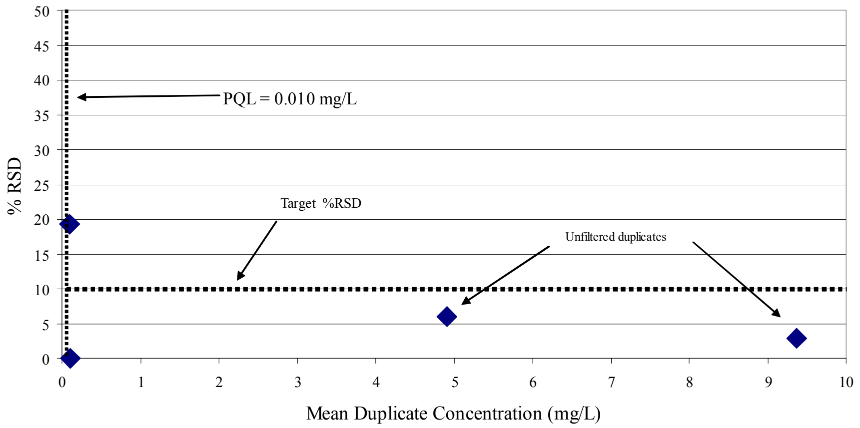


Figure D-1
Blind Field Duplicate Precision % RSD
as a Function of Concentration

% RSD vs. Concentration
Ammonia-N



% RSD vs. Concentration
Nitrate-N



% RSD vs. Concentration
Total Persulfate Nitrogen

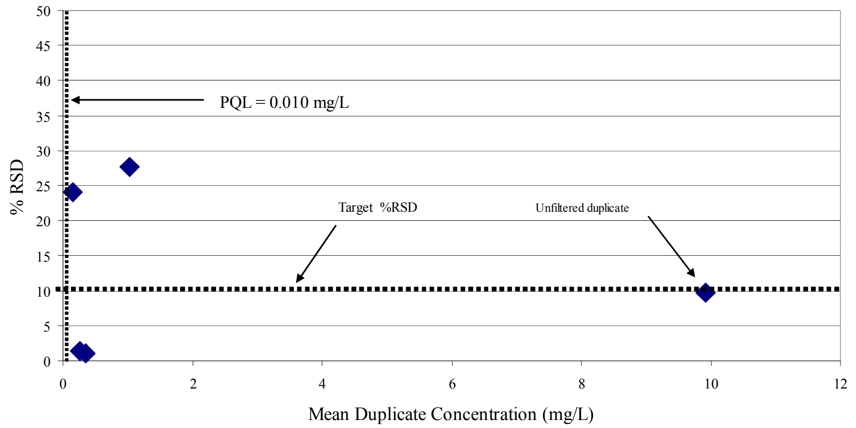
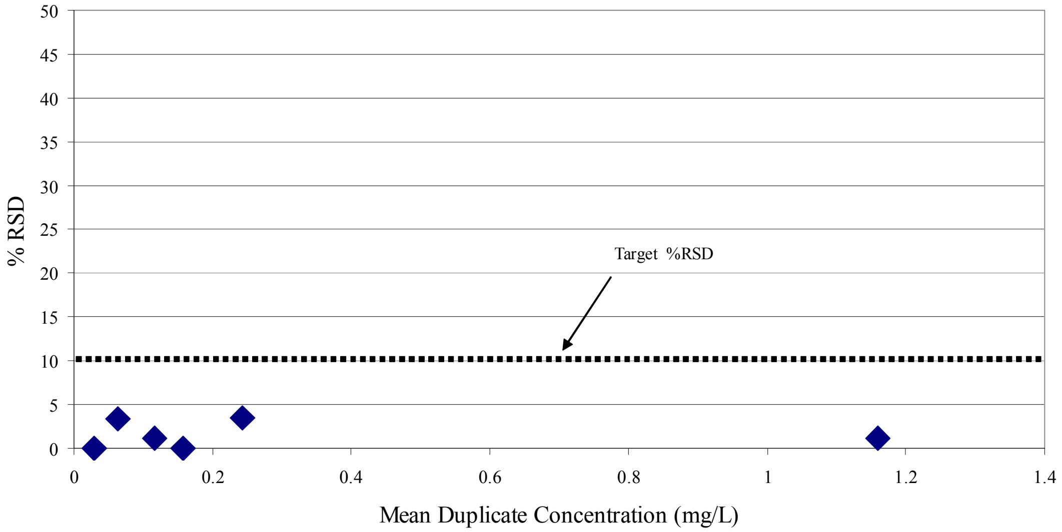


Figure D-1 (page 2)

% RSD vs. Concentration
Orthophosphate



% RSD vs. Concentration
Total Dissolved Phosphorus

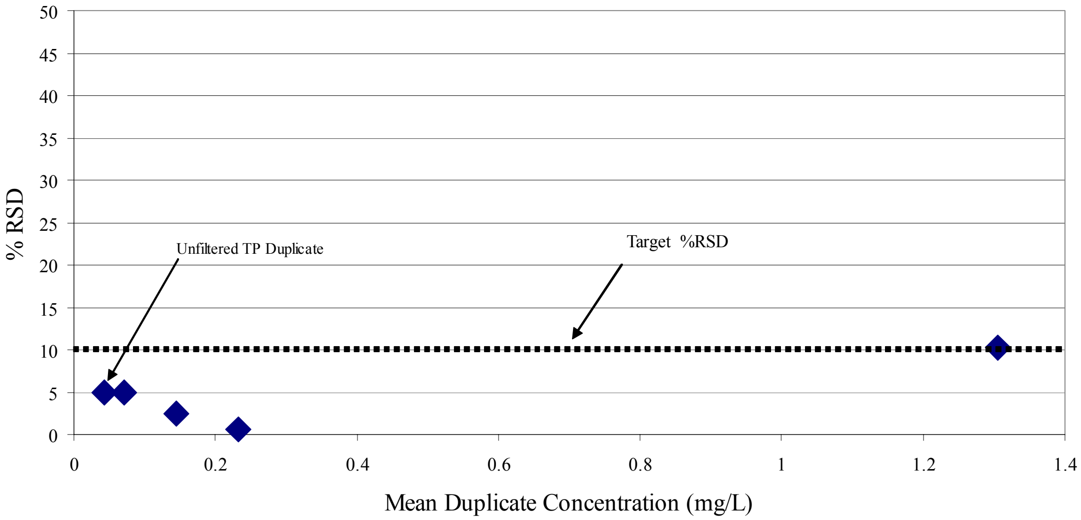
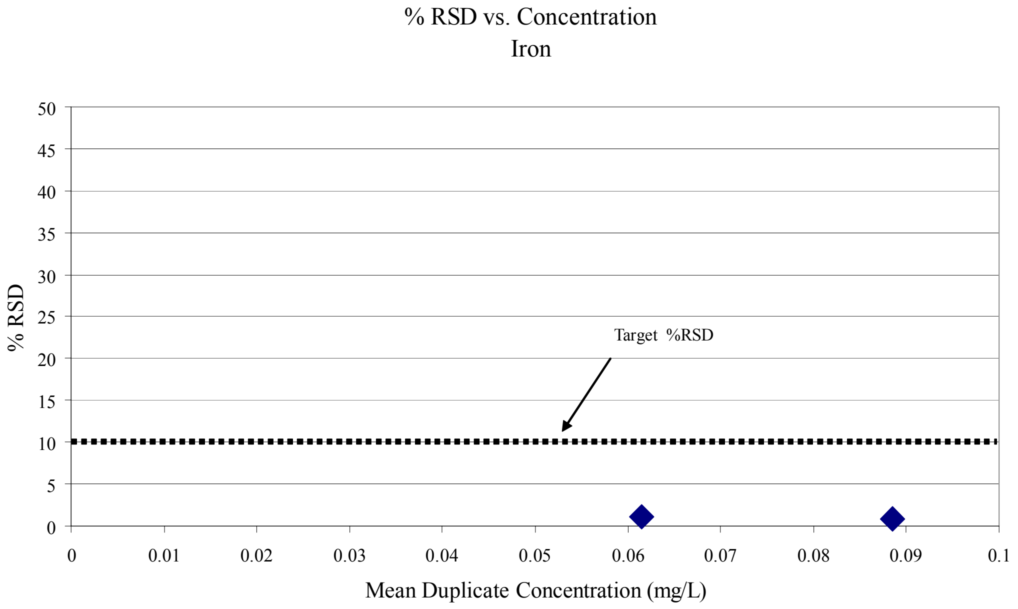


Figure D-1 (page 3)



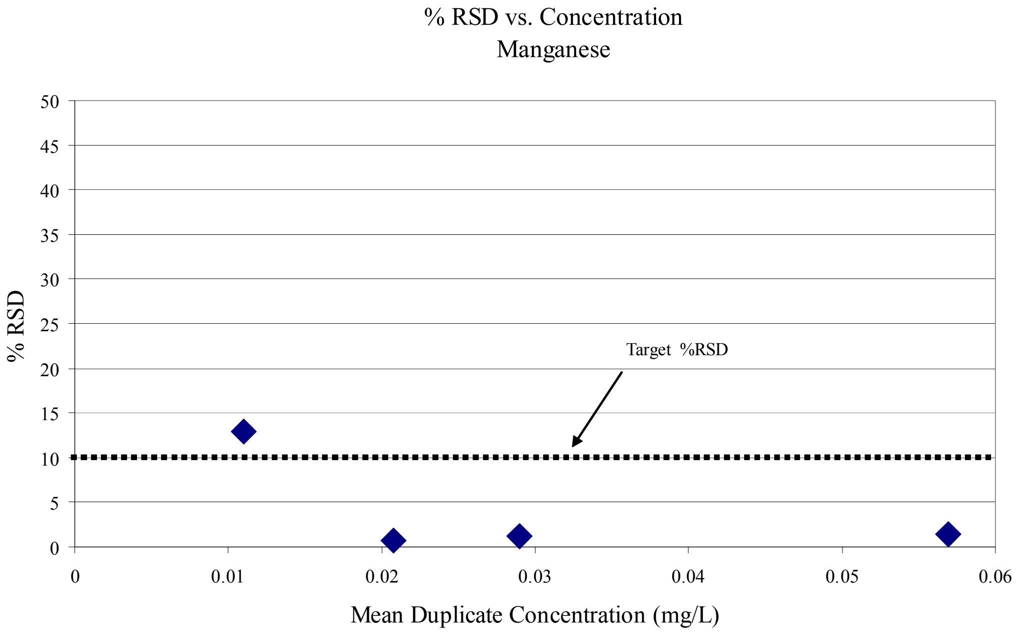


Figure D-1 (page 4)

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Appendix E

Piezometer Head and Gradient Data

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Table E-1 - Field Measurements and Vertical Gradient Estimates - Piezometers

Station Name	Head above lake water surface May-01 (h) (mm)	Distance between screen mid-point and sediment surface May-01 (z) (mm)	Vertical Gradient May-01 (i = h/z)	Head above lake water surface July-01 (h) (mm)	Distance between screen mid-point and sediment surface July-01 (z) (mm)	Vertical Gradient July-01 (i = h/z)	Head above lake water surface October-01 (h) (mm)	Distance between screen mid-point and sediment surface October-01 (z) (mm)	Vertical Gradient October-01 (i = h/z)
MLGW-01	17	762	0.022	30	762	0.039	NM	NM	NM
MLGW-02	110	1676	0.066	127	1676	0.076	100	1676	0.060
MLGW-04S	NM	NM	NM	0	762	*0.000*	17	762	0.022
MLGW-04D	7	1372	0.005	10	1372	0.007	11	1372	0.008
MLGW-05	25	762	0.033	22	762	0.029	21	762	0.028
MLGW-06	5	1067	0.005	1	1067	0.001	5	1067	0.005
MLGW-07	0	914	*0.000*	**(-12)**	914	**(-0.013)**	8	914	0.009
MLGW-08	2	1250	0.002	10	1250	0.008	26	1250	0.021
MLGW-09	22	1372	0.016	21	1372	0.015	62	1372	0.045
MLGW-10	60	1372	0.044	21	1372	0.015	66	1372	0.048
MLGW-11	0	1433	*0.000*	3	1433	0.002	4	1433	0.003
MLGW-12	10	1372	0.007	22	1372	0.016	1	1372	0.001
MLGW-13	10	884	0.011	10	884	0.011	5	884	0.006

Bolded values indicate a negative (downward) gradient between the lake and groundwater, suggesting lake recharge of the aquifer at that location.
Non-bolded values indicate a positive (upward) gradient between the lake and groundwater, suggesting groundwater discharge to the lake.
Italicized values indicate no discernable gradient could be measured.

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Appendix F

Estimates of Groundwater Loading of Phosphorus to Moses Lake

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Introduction

On the basis of existing information, as well as data collected during this study, estimates were developed for the annual load of dissolved phosphorus that enters Moses Lake via groundwater discharge. This appendix describes the procedures and assumptions used to develop these estimates.

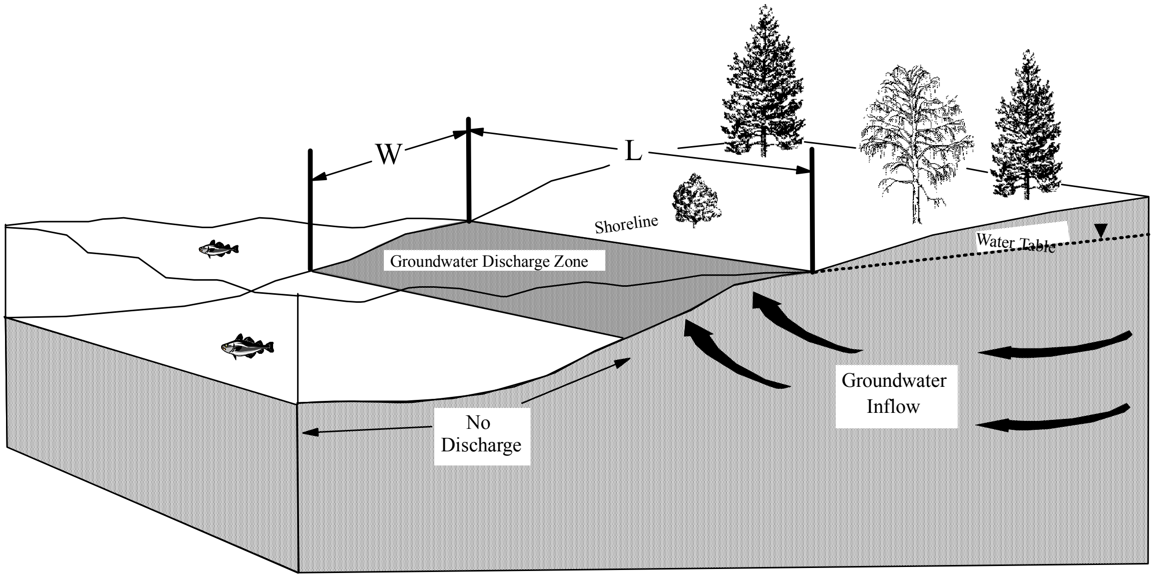
Conceptual Model and Background Information

In most geologic settings the discharge of groundwater (and groundwater-borne solute) to lakes is concentrated in a narrow portion of the littoral zone (Winter, 1978; McBride and Pfannkuch, 1975, Lee et al., 1980; Harvey et al., 2000, Wagner et al., 1983; Brock et al., 1982). This process is illustrated in Figure F-1, which depicts the conceptual model of the interaction between an aquifer and a lake that was used for this study. The figure shows groundwater flow lines bending sharply upward as they approach a lake from an upgradient aquifer system, ultimately discharging through the face of a near-shore discharge zone. The principal direction of groundwater flow at the point of discharge is essentially perpendicular to the lake bottom, and no discharge occurs beyond the outer limit of the zone.

Previous studies have shown that groundwater discharge rates to lakes are not constant across a discharge zone. A number of investigators have confirmed that hydraulic gradients and groundwater discharge rates decline exponentially with offshore distance (McBride and Pfannkuch, 1975; Lee et al., 1980; Harvey et al., 2000; Brock et al., 1982). In studying a lake of similar geometry to Moses Lake, McBride and Pfannkuch (1975) demonstrated that approximately 50% of groundwater discharge was concentrated within 17 m of the shoreline, and 90% of all discharge occurred within 60 m of shore. The width (W) of the discharge zone (Figure F-1), as well as the rate of decline of discharge vs. distance from shore can vary from location to location and lake to lake. Differences in W are, in part, a function of the ratio between the horizontal (K_h) and vertical (K_v) hydraulic conductivity of the discharge zone sediments (McBride and Pfannkuch, 1975; Lee et al., 1980).

In addition to spatial changes in discharge rate, there is evidence that solute concentrations also show variability across the width of a discharge zone. One of the most important controls is the effect of hydrodynamic dispersion (mixing). For example, a study by Lee et al. (1980) indicated the potential for significant reductions in the discharge concentration of a conservative tracer in comparison to the upgradient (onshore) condition. This reduction was due primarily to dispersion effects acting in the final portion of the flow path; dispersion effects were determined to be greatest at higher K_h/K_v ratios. Solute concentrations are also more likely to be attenuated with distance from a shoreline due to the longer flow path. These findings suggest that concentrations of a discharging solute may decline as a function of the distance from the upgradient edge of the discharge zone.

As the geometry of the lakes that have been studied in the past are similar to Moses Lake (i.e. the width of the lake is equal to or greater than the thickness of the underlying groundwater system), the findings reported in these studies have been used as a guide for developing the estimates presented in this appendix.



Not to scale

Figure F-1
Conceptual Diagram of Groundwater
Discharge and Subsurface
Solute Transport to Moses Lake

Procedures

To calculate phosphorus input from the aquifer system to the lake, values were first developed for groundwater volume input across the face of the assumed discharge zone using a Darcian flow analysis. The discharge volume estimates were then integrated with estimated phosphorus concentrations of the discharging groundwater to determine mass flux. This section describes the specific procedures used to calculate these values for the Moses Lake study area.

Volumetric Flux

To estimate the rate of groundwater discharge across a discharge face, the surface area of the face must be determined. In a lake this surface area (A) is calculated as the product of the length (L) and width (W- measured from the shoreline outward) of the discharge zone (Figure F-1). The area value is then integrated with estimates of the hydraulic conductivity (K) and hydraulic gradient (i) within the uppermost discharge zone sediments using Darcy's equation:

$$Q = KiA \qquad (1)$$

where: Q = groundwater discharge rate (L^3/t)
K = hydraulic conductivity in the principal direction of flow (L/t)
i = hydraulic gradient in the principal direction of flow (L/L)[1]
A = area of discharge face (L^2)

To account for the fact that groundwater discharge rates normally decline across the width of a discharge zone, the findings of McBride and Pfannkuch (1975) and others were used to further refine the conceptual model. For this analysis, the discharge zone was subdivided into two separate sections (Zones 1 and 2, Figure F-2). It was assumed that 50% of the groundwater discharge occurs in the inner zone, the remaining 50% discharging through the outer zone. Because field conditions of the hydraulic gradient (i) are best known for Moses Lake for the near-shore area (study piezometer locations averaged approximately 8 m out from the shoreline), Equation 1 was calculated specifically for Zone 1 using:

$$Q_1 = Ki_1A_1 \qquad (2)$$

where: Q_1 = groundwater discharge rate for Zone 1 (L^3/t)
K = hydraulic conductivity in the principal direction of flow (L/t)
i_1 = hydraulic gradient in the principal direction of flow in Zone 1 (L/L)
A_1 = area of Zone 1 discharge face (L^2)

where:

$$A_1 = W_1L \qquad (3)$$

W_1 = width of Zone 1 (L)
L = length of Zone 1 (L)

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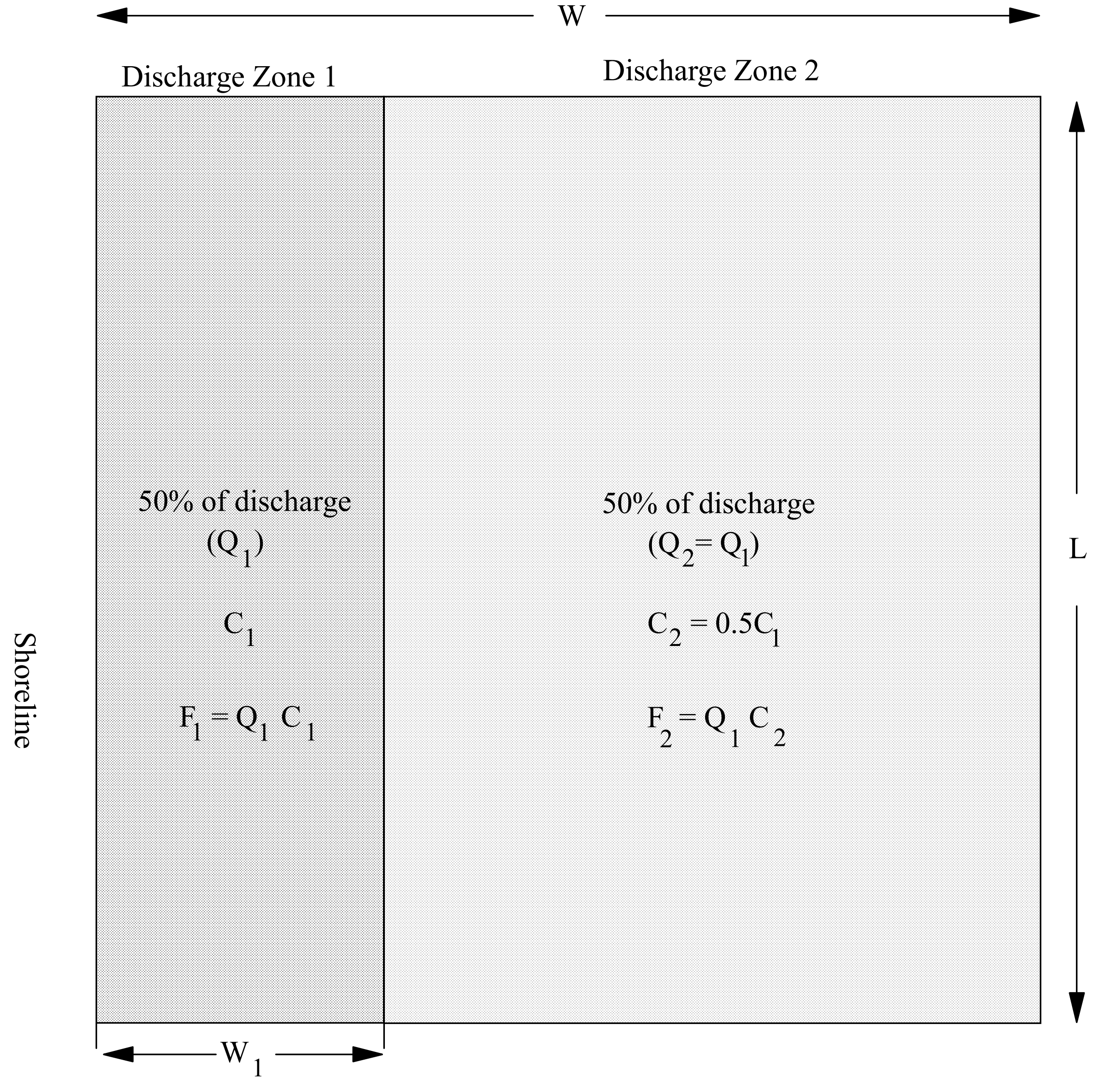
[1] Refer to Figure 8 of the main report for the method used to calculate the hydraulic gradient in the groundwater discharge zone.



Q = Groundwater discharge rate (L^3/t)

C = Discharge concentration (M/L^3)

F = Mass Flux (M/t)

Figure F-2
Conceptual Model of Flux Estimate

Doubling the groundwater discharge rate estimated for Zone 1 (Q_1) provides the groundwater discharge rate (Q) for the entire discharge zone width (W):

$$Q = 2Q_1 \qquad (4)$$

or

$$Q = 2(Ki_1W_1L) \qquad (5)$$

Mass Flux

To estimate the mass flux of phosphorus carried to Moses Lake by advective groundwater inflow, the flow rate estimate developed using equation (1) is further integrated with a representative value for the water-quality concentration of the discharging groundwater using:

$$F = QC \qquad (6)$$

where: F = total mass flux rate for parameter of interest (M/t)
C = groundwater phosphorus concentration (M/V)

To account for the effects of dispersion and attenuation on the solute concentrations moving away from shore (Lee et al., 1980), equation (5) was calculated separately for Zones 1 and 2 (Figure F-2) using:

$$F_1 = Q_1C_1 \qquad (7)$$

where: F_1 = mass flux rate for Zone 1 (M/t)
C_1 = groundwater phosphorus concentration discharging to Zone 1 (M/V)

and

$$F_2 = Q_1C_2 \qquad (8)$$

where: F_2 = mass flux rate for Zone 2 (M/t)
C_2 = groundwater phosphorus concentration discharging to Zone 2 (M/V)

where:

$$C_2 = 0.5C_1 \qquad (9)$$

To estimate the total phosphorus mass flux across the entire width (W) of the discharge zone, the mass flux values from the two sections are simply summed:

$$F = F_1+F_2 \qquad (10)$$

Estimate Assumptions

In order to estimate flux rates to the lake, a number of assumptions were required regarding appropriate values to use for the input variables of the equations presented above. In most cases the assumptions simplify the complexity of the natural system. The key assumptions used for modeling groundwater flux and phosphorus input to Moses Lake by groundwater discharge are presented and discussed below:

- The values developed for groundwater loading of phosphorus to Moses Lake are presented in terms of dissolved *mass* delivered to the lake/aquifer interface over a given time frame. The values are not adjusted to account for changes in *phase* that may occur at or beyond the point of discharge. As discussed in the main report, phosphorus that is dissolved and mobile in the groundwater system may be rapidly immobilized to a sorbed phase (typically bonded onto the surfaces of lake-bottom sediments) by changes in the geochemical environment upon entry to the lake. Sorption processes do not degrade or destroy phosphorus; they only result in the change in state (therefore potential availability) of the nutrient. Phosphorus sorption can be a reversible process if geochemical conditions change.

- To assist the development of a water quality model for the lake, the lake was subdivided into three separate areas for the load analysis: the Main Arm (Area 1), Parker Horn (Area 2), and Pelican Horn (Area 3) (Figure F-3). Flux estimates for each area were developed as described above using data drawn from the study stations located within that area. Flux rate values developed for the three separate areas can be summed to derive a lake-wide OP flux rate.

- The length (L) of each discharge area was selected based on an analysis of available information regarding regional and local groundwater flow patterns, and subsurface distribution of the study area hydrostratigraphic units. Figure F-3 shows that no groundwater discharge is assumed along much of the western and southwestern shoreline of the lake. As discussed in the main report, historic water-level data indicate that along this portion of the shoreline, the predominant flow direction is downward recharge from the lake to the adjacent aquifer.

- Two additional areas of Moses Lake were considered zones of negligible inflow: a portion of the northern shoreline of the Main Arm near the big bend, and the southeastern shoreline of Pelican Horn (Figure F-3). Both of these areas are considered unlikely to provide significant input of phosphorus due to the presence of finer-grained Ringold sediments in contact with the lake along these portions of the shoreline. Groundwater flow rates and transport of OP through the Ringold sediments are judged to be limited.

- Because the width of the groundwater discharge zone and rate of decline of discharge with distance from shore were not established by field measurement during this study, two different input values were assumed and tested for the W_1 variable (Figure F-2). As derived directly from work by McBride and Pfannkuch (1975), model calculations were run assuming $W_1 = 17$ m. This assumption is considered reasonable because of the similarities in geometry between the studied lakes. To account for the fact that other authors have reported narrower discharge zones (for example Lee et al., 1980; Wagner et al., 1983; Brock et al., 1982), the model calculations were also run assuming $W_1 = 8$ m (the approximate average distance of the study piezometers from shore). This approach results in several unique solutions (scenarios) for each discharge area, providing a range for the probable discharge and loading rate.

- The discharge estimates were derived assuming that no groundwater discharge to the lake occurs beyond the outer limit of the discharge zone (Figures F-1 and F-2). While past field investigations show that this assumption is not always true, it is assumed that any groundwater contribution occurring beyond this point is probably negligible in relation to the overall estimates presented here.

- Because of the shallow depth and low-angle bottom profile of Moses Lake, it was assumed that the face of the discharge zone is essentially horizontal. On this basis it was further assumed that the principal direction of flow at the discharge face is vertical (in an upward direction) for the entire discharge zone. Since no field measurements of K_v were collected for the uppermost discharge zone sediments during this study, the values used to represent the conductivity of the discharge zone sediments are based on values reported in the existing literature for the K_v of the local Pleistocene flood deposits. It was further assumed that the uppermost sediments of the discharge zone are laterally homogeneous.

- To address both spatial variability and uncertainty in the discharge zone grain size and conductivity condition, three different values of K_v (1, 24, and 45 m/day) were tested in the model calculations for each discharge area. The two highest values are drawn directly from published data for the K_v of study area flood deposits. The lowest value was used to account for the fact that there is often a bed layer of finer sediment deposited on the bottom of lakes. A K_v value of 1 m/day for a fine- to medium-sand (a significantly smaller grain size than reported and observed for the flood deposits) was therefore used to estimate a lower-bound conductivity condition. The approach of varying K_v (as well as W_1) resulted in a number of unique solutions (scenarios) that collectively provide an estimate of the probable range of discharge and loading rate (including upper and lower-bound values).

- The hydraulic gradient value (i_1) input for each discharge zone is drawn from the field measurements of the vertical hydraulic gradient measured in study piezometers in the final meters of the groundwater flowpath. The average of all measurements collected from the stations within an area was used as a representative value for (i_1) for that area.

- Orthophosphate (OP) is assumed to be the dominant form of phosphorus dissolved in groundwater; therefore OP concentrations were used to estimate phosphorus flux by groundwater discharge.

- The OP concentration (C_1) input for each discharge zone is drawn from filtered samples collected from piezometers screened in the final meters of the groundwater flowpath. The geometric mean of all of the measurements collected from the stations within an area was used as a representative value for (C_1) for that area. The values are considered the best estimate of the concentration of dissolved phosphorus in the groundwater discharging to the lake at that point. Sampling of the groundwater just prior to discharge allows measurement of the net effects of aquifer diffusion, dispersion, and attenuation occurring upgradient of the lake. The model assumes that no change in the OP concentration occurs between the point of measurement and the point of discharge. See the main report for further discussion of the validity of this assumption.

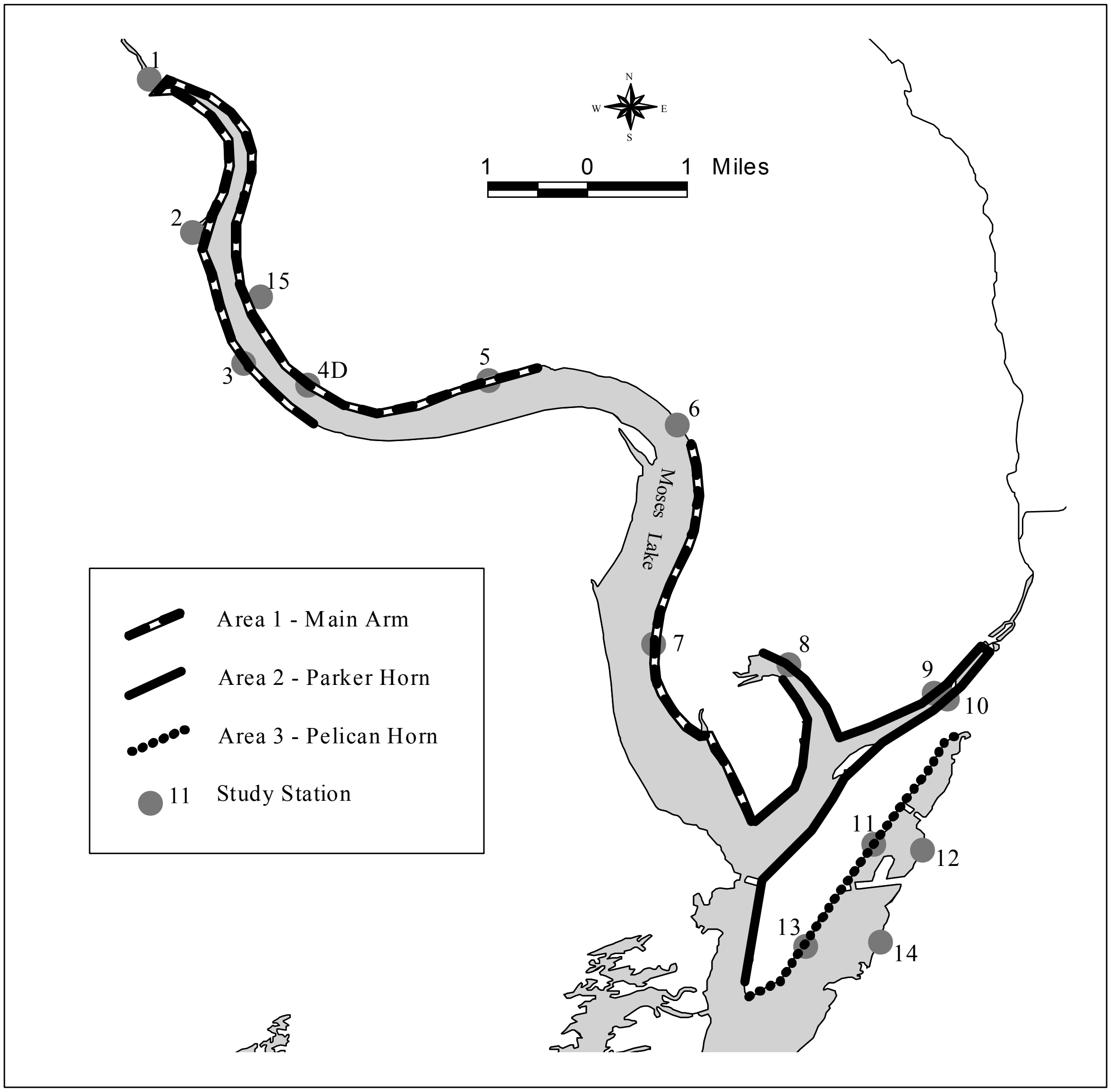


Figure F-3
Groundwater Discharge Areas

Results

Table F-1 presents the input values and resulting flux estimates for Moses Lake, by area and scenario. The range of values for lake-wide loading that were derived by summing the flux rates for the individual discharge zones are presented in Table F-2. The modeling results indicate an annual OP load to the lake via groundwater discharge between approximately 400 - 40,000 kg_{op}/yr. A value between 10,000 – 20,000 kg_{op}/yr is judged to be the best estimate of field conditions.

Table F-2 – Summary Range of Annual Orthophosphate Mass Flux to Moses Lake by Groundwater Discharge (kg_{op}/yr)

Scenarios (see Table F-1)	Estimated Annual OP Mass Flux Rate (kg_{op}/yr)
1a+2a+3a	420
1b+2b+3b	10066
1c+2c+3c	18904
1d+2d+3d	893
1e+2e+3e	21398
1f+2f+3f	40135

Table F-1 - Phosphorus Loading Estimates to Moses Lake via Groundwater Inflow

Area 1 - Main Arm (Stations 1-7)

Scenario	Length of Discharge Zone L (m)	Width of Discharge Zone 1 W_1 (m)	Area of Discharge Zone 1 $A_1 = L*W_1$ (m2)	Vertical Hydraulic Conductivity of Discharge Zone Sediments K_v (m/day)	Average Vertical Hydraulic Gradient in Discharge Zone 1 i_1 (m/m)	Estimated Daily GW Flux Zone 1 $Q_1 = K_v i_1 A_1$ (L/day)	Geometric Mean of Area OP Concentration C_1 (mg/L)	Estimated Annual OP Flux Zone 1 $F_1 = Q_1 * C_1$ (kg/yr)	Assumed Concentration of Discharge Zone 2 $C_2 = 0.5C_1$ (mg/L)	Estimated Annual OP Flux Zone 2 $F_2 = Q_1 * C_2$ (kg/yr)	Estimated Total Annual OP Flux $F = F_1 + F_2$ (kg/yr)
1a	25900	8	207200	1(A)	0.021	4.35E+06	0.071	113	0.035	56	168
1b	25900	8	207200	24(B)	0.021	1.04E+08	0.071	2695	0.035	1329	4024
1c	25900	8	207200	45(C)	0.021	1.96E+08	0.071	5079	0.035	2504	7583
1d	25900	17	440300	1(A)	0.021	9.25E+06	0.071	240	0.035	118	358
1e	25900	17	440300	24(B)	0.021	2.22E+08	0.071	5753	0.035	2811	8564
1f	25900	17	440300	45(C)	0.021	4.16E+08	0.071	10781	0.035	5314	16095

(A) Lower-bound assumption of K_v (fine sand bottom sediment covering discharge zone).

(B) Lower-bound model-calibrated K_v for Pleistocene flood deposits (USACE, 2001).

(C) Upper-bound assumption of K_v (mid-range of model-calibrated Kv for Pleistocene flood deposits, USACE, 2001).

Table F-1 (page 2)

Area 2 - Parker Horn (Stations 8-10)

Scenario	Length of Discharge Zone L (m)	Width of Discharge Zone 1 W_1 (m)	Area of Discharge Zone 1 $A_1 = L*W_1$ (m2)	Vertical Hydraulic Conductivity of Discharge Zone Sediments K_v (m/day)	Average Vertical Hydraulic Gradient in Discharge Zone 1 i_1 (m/m)	Estimated Daily GW Flux Zone 1 $Q_1 = K_v i_1 A_1$ (L/day)	Geometric Mean of Area OP Concentration C_1 (mg/L)	Estimated Annual OP Flux Zone 1 $F_1 = Q_1 * C_1$ (kg/yr)	Assumed Concentration of Discharge Zone 2 $C_2 = 0.5C_1$ (mg/L)	Estimated Annual OP Flux Zone 2 $F_2 = Q_1 * C_2$ (kg/yr)	Estimated Total Annual OP Flux $F = F_1 + F_2$ (kg/yr)
2a	15650	8	125200	1(A)	0.024	3.00E+06	0.107	117	0.054	59	176
2b	15650	8	125200	24(B)	0.024	7.21E+07	0.107	2816	0.054	1421	4237
2c	15650	8	125200	45(C)	0.024	1.35E+08	0.107	5272	0.054	2661	7933
2d	15650	17	266050	1(A)	0.024	6.39E+06	0.107	250	0.054	126	376
2e	15650	17	266050	24(B)	0.024	1.53E+08	0.107	5975	0.054	3016	8991
2f	15650	17	266050	45(C)	0.024	2.87E+08	0.107	11209	0.054	5657	16866

(A) Lower-bound assumption of K_v (fine sand bottom sediment covering discharge zone).

(B) Lower-bound model-calibrated K_v for Pleistocene flood deposits (USACE, 2001).

(C) Upper-bound assumption of K_v (mid-range of model-calibrated Kv for Pleistocene flood deposits, USACE, 2001).

Table F-1 (page 3)

Area 3 - Pelican Horn (Stations 11,13)

Scenario	Length of Discharge Zone L (m)	Width of Discharge Zone 1 W_1 (m)	Area of Discharge Zone 1 $A_1 = L*W_1$ (m2)	Vertical Hydraulic Conductivity of Discharge Zone Sediments K_v (m/day)	Average Vertical Hydraulic Gradient in Discharge Zone 1 i_1 (m/m)	Estimated Daily GW Flux Zone 1 $Q_1 = K_v i_1 A_1$ (L/day)	Geometric Mean of Area OP Concentration C_1 (mg/L)	Estimated Annual OP Flux Zone 1 $F_1 = Q_1 * C_1$ (kg/yr)	Assumed Concentration of Discharge Zone 2 $C_2 = 0.5C_1$ (mg/L)	Estimated Annual OP Flux Zone 2 $F_2 = Q_1 * C_2$ (kg/yr)	Estimated Total Annual OP Flux $F = F_1 + F_2$ (kg/yr)
3a	6000	8	48000	1(A)	0.0055	2.64E+05	0.52	50	0.26	25	75
3b	6000	8	48000	24(B)	0.0055	6.34E+06	0.52	1203	0.26	602	1805
3c	6000	8	48000	45(C)	0.0055	1.19E+07	0.52	2259	0.26	1129	3388
3d	6000	17	102000	1(A)	0.0055	5.61E+05	0.52	106	0.26	53	159
3e	6000	17	102000	24(B)	0.0055	1.35E+07	0.52	2562	0.26	1281	3843
3f	6000	17	102000	45(C)	0.0055	2.52E+07	0.52	4783	0.26	2391	7174

(A) Lower-bound assumption of K_v (fine sand bottom sediment covering discharge zone).

(B) Lower-bound model-calibrated K_v for Pleistocene flood deposits (USACE, 2001).

(C) Upper-bound assumption of K_v (mid-range of model-calibrated Kv for Pleistocene flood deposits, USACE, 2001).

Scenarios	Annual Lake-wide OP Flux ΣF (kg_{op}/yr)
1a+2a+3a	420
1b+2b+3b	10066
1c+2c+3c	18904
1d+2d+3d	893
1e+2e+3e	21398
1f+2f+3f	40135