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Lake Whatcom Total Maximum Daily Load Groundwater Study

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Public Comments on the Draft

The draft of this report was posted to the web on November 29, 2004, with comments due to the author by January 3, 2005. All comments and responses are listed in Appendix I of this report.

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WASHINGTON STATE
DEPARTMENT OF
ECOLOGY

Lake Whatcom
Total Maximum Daily Load
Groundwater Study

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by
Charles F. Pitz, L.G., L.HG.

Environmental Assessment Program
Olympia, Washington 98504-7710

January 2005

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Abstract

Lake Whatcom is listed on the 1998 Washington State 303(d) list of waterbodies not attaining the water quality criterion for dissolved oxygen. Evidence indicates that oxygen depletion in the lake is the result of several factors, including eutrophication processes driven by the availability of phosphorus and nitrogen-based nutrients. A Total Maximum Daily Load study has been initiated in order to develop a water quality model and make recommendations for reduction and allocation of pollutant loads for the lake.

To characterize the role of groundwater-derived nutrient inflow in the lake's nutrient budget, a network of 14 lake-bed piezometers was installed in the near-shore sediments around the lake perimeter. Samples of groundwater seepage to the lake were collected from the piezometers quarterly from October 2002 through August 2003. Samples were analyzed for target nutrients – total dissolved phosphorus, orthophosphate-P, nitrate+nitrite-N, and ammonia-N – as well as dissolved oxygen, iron, manganese, total dissolved solids, chloride, dissolved organic carbon, and boron.

The median dissolved orthophosphate-P concentration observed in discharging groundwater was 0.06 mg/L, with a maximum reported value of 0.35 mg/L. The median total dissolved phosphorus concentration was 0.14 mg/L, with a maximum value of 2.18 mg/L. Nitrogen was normally present in the form of ammonia, with a median concentration of 0.21 mg/L as N. At several stations, the groundwater quality profile and upgradient land-use development patterns suggest an anthropogenic (human-related) origin for the elevated nutrients observed. In other cases, the data suggest the nutrients are likely associated with the decomposition of naturally occurring organic matter present in the aquifer sediments. Loading calculations for the 2003 water year estimate an upper-bound orthophosphate-P mass flux to the lake ranging from 900 to 1300 kg_{op} via direct groundwater inflow. Changes in redox conditions in the final portion of the groundwater flow path may attenuate a significant percentage of the estimated load.

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The author would like to thank the following people for their contributions to this study:

- Joanne Snarski, Washington State Department of Natural Resources, for help in permitting arrangements for piezometers.
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Introduction

The federal Clean Water Act (1972), Section 303(d), requires that states report waterbodies that do not meet ambient water quality standards. A Total Maximum Daily Load (TMDL) study is required for all impaired waterbodies assigned to the 303(d) inventory list. A TMDL is an assessment of the pollutant loading capacity of a waterbody that still allows the waterbody to meet the assigned water quality criteria and support beneficial use.

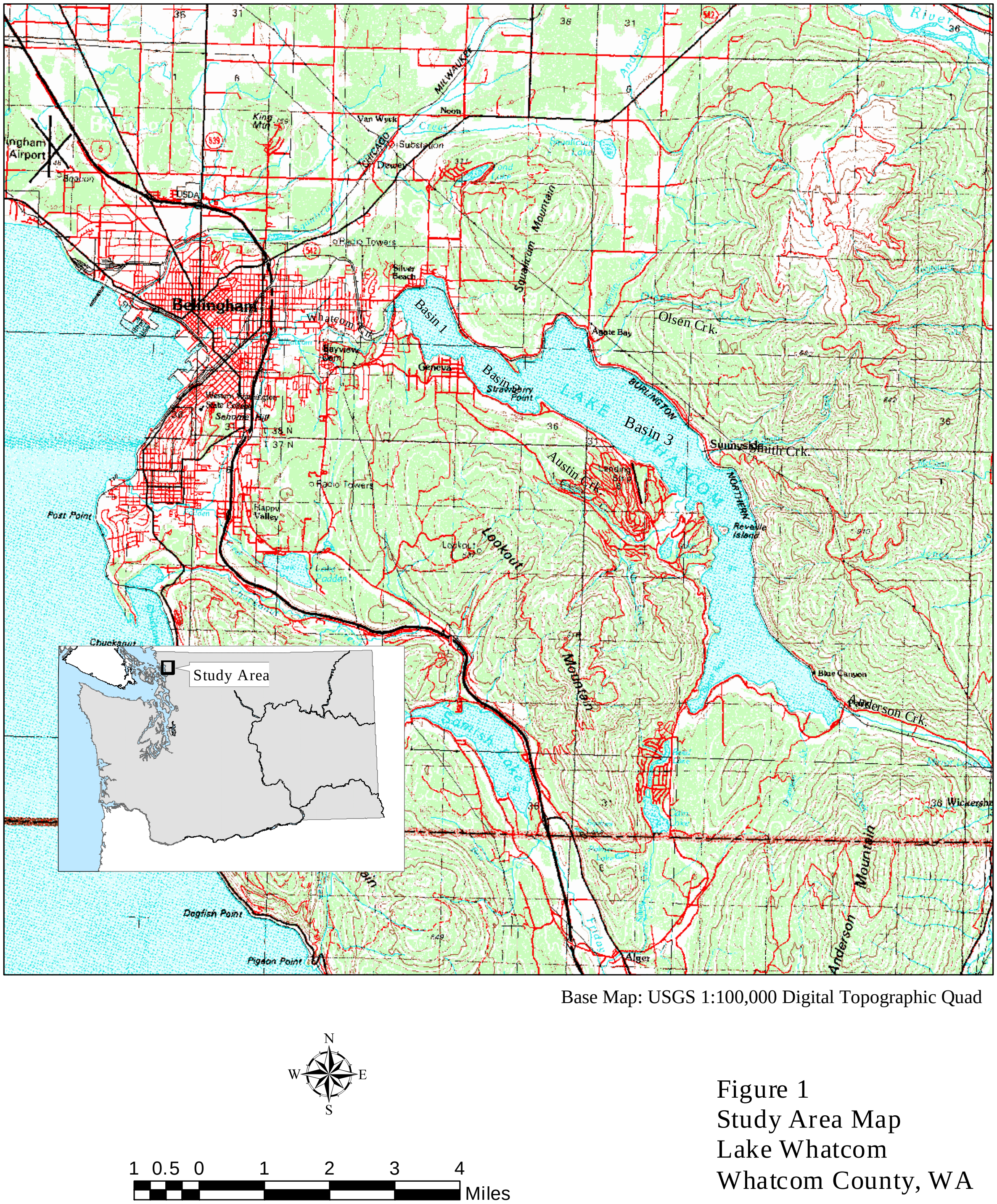
Lake Whatcom, located in Whatcom County (Figure 1), is listed on the 1998 Washington State 303(d) list of waterbodies failing to meet the water quality criterion for dissolved oxygen. Evidence indicates that the oxygen depletion, a seasonal condition observed in the lake at water depths below the thermocline (8-10 meters), has been significantly increasing over the past two decades (Pelletier, 1998). This change is interpreted to be the result of several factors, including the direct loading of organic matter from the surrounding watershed, and eutrophication processes driven by the availability of phosphorus and nitrogen-based nutrients (Cusimano et al., 2002).

The lake serves as an important natural resource to the local community, providing the primary drinking water supply for the city of Bellingham as well as a number of residents living adjacent to the lake. The lake also is used extensively for sport fishing, swimming, and other types of water recreation. The depletion of oxygen can lead to the deterioration of the water quality for drinking purposes (e.g., an increase in algal content of the water), and can limit the available habitat for oxygen-dependent organisms such as fish.

To address the above concerns, a TMDL study has been initiated for the lake on the behalf of the Northwest Regional Office and the Bellingham Field Office of the Washington State Department of Ecology (Ecology). As part of this study, Ecology’s Environmental Assessment Program (EAP) undertook a comprehensive water quality sampling program for the lake throughout portions of 2002 and 2003. The ultimate goal of this sampling program is the development of a water quality model, and recommendations for reduction and allocation of pollutant loads.

Study Purpose and Goals

To ensure a comprehensive characterization of the magnitude and origin of nutrient loading to Lake Whatcom, a better understanding of the role of groundwater-derived nutrient inflow to the lake is required. The primary goal of the study described in this report was to document the phosphorus and nitrogen content of the groundwater that is directly discharging into the lake from the adjacent aquifer system. A secondary goal of this work was to evaluate, where possible, the potential sources for the nutrients observed. Data from the groundwater study will be incorporated with information from additional EAP studies of basin surface water conditions to develop a lake-wide nutrient budget, and representative water quality model.



Base Map: USGS 1:100,000 Digital Topographic Quad

Figure 1
Study Area Map
Lake Whatcom
Whatcom County, WA

Physical Setting

Study Area Description

Lake Whatcom is a large natural lake located in western Whatcom County, Washington (Figure 1). The lake, which consists of three distinct basins separated by resistant bedrock sills (Figure 2), is approximately 10 miles long, and has an average surface elevation of approximately 315 feet above mean sea level.

The total surface area of the lake is approximately 5000 acres, with a total watershed land surface area of about 32,000 acres. Basins number 1 and 2 are comparatively small and shallow, with average water depths of approximately 10 meters. Basin 3, in contrast, is very deep, with an average water depth of over 50 meters and a maximum water depth of over 100 meters. Basin 3 is estimated to contain 96% of the lake's total water volume (Cusimano et al., 2002).

The watershed topography surrounding Lake Whatcom is dominated by rugged, mountainous terrain adjacent to Basin 3, and low-relief foothills adjacent to Basins 1 and 2. The watershed is drained by a number of perennial streams discharging to the lake, most notably Carpenter, Olsen, and Smith creeks in the eastern portions of the watershed, Austin Creek draining from the western portion of the watershed through the Sudden Valley area, and Anderson Creek at the far southern end of the lake (Figure 2).

In order to maintain optimal lake levels for storage and flood prevention purposes, the lake surface elevation and outflow rate are currently regulated by manipulation of a dam located at the upper end of Whatcom Creek. Flow is also periodically diverted from the middle fork of the Nooksack River outside of the watershed, and is routed into the lake via Anderson Creek. This diversion, used to offset withdrawals, is often the major water source to the lake during the summer.

Annual precipitation in the watershed ranges from an average of 40-45 inches/yr in the northern portion, to 50-60 inches/yr in the mountainous southern and eastern portions (Miller et al., 1973). The local maritime climate is characterized by warm, dry summers and mild, rainy winters, with the majority of the annual precipitation typically falling between October and April. To illustrate the general precipitation pattern during the study period, Figure 3 presents the daily precipitation data from the Geneva Intake and Brannian Creek rain-gage stations (City of Bellingham, 2004).

Land use in the Lake Whatcom watershed includes a mix of urban and residential development, and undeveloped forestland (Figure 4). Basin 1, which lies mostly within the city limits of Bellingham, is the most heavily developed portion of the lake. Residential development is also concentrated along the shoreline of Basin 2 and the northern half of Basin 3. There is also extensive residential and recreational development in the Sudden Valley area adjacent to the western shoreline of Basin 3. The remainder of the watershed is dominated by commercial forestland, with low density residential development typically concentrated immediately adjacent to the shoreline.

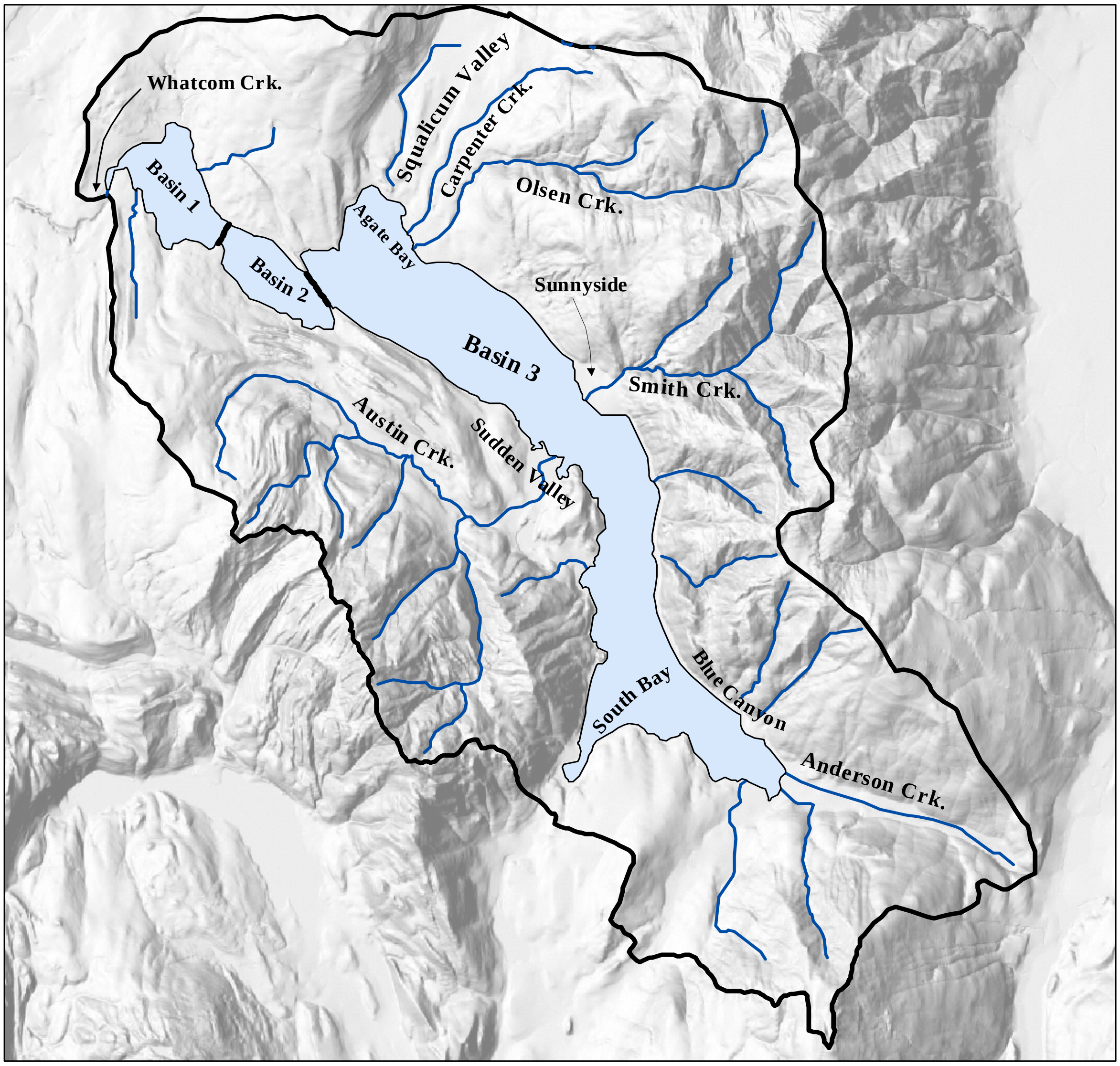


Figure 2
Drainage and Watershed
Topography - Lake Whatcom

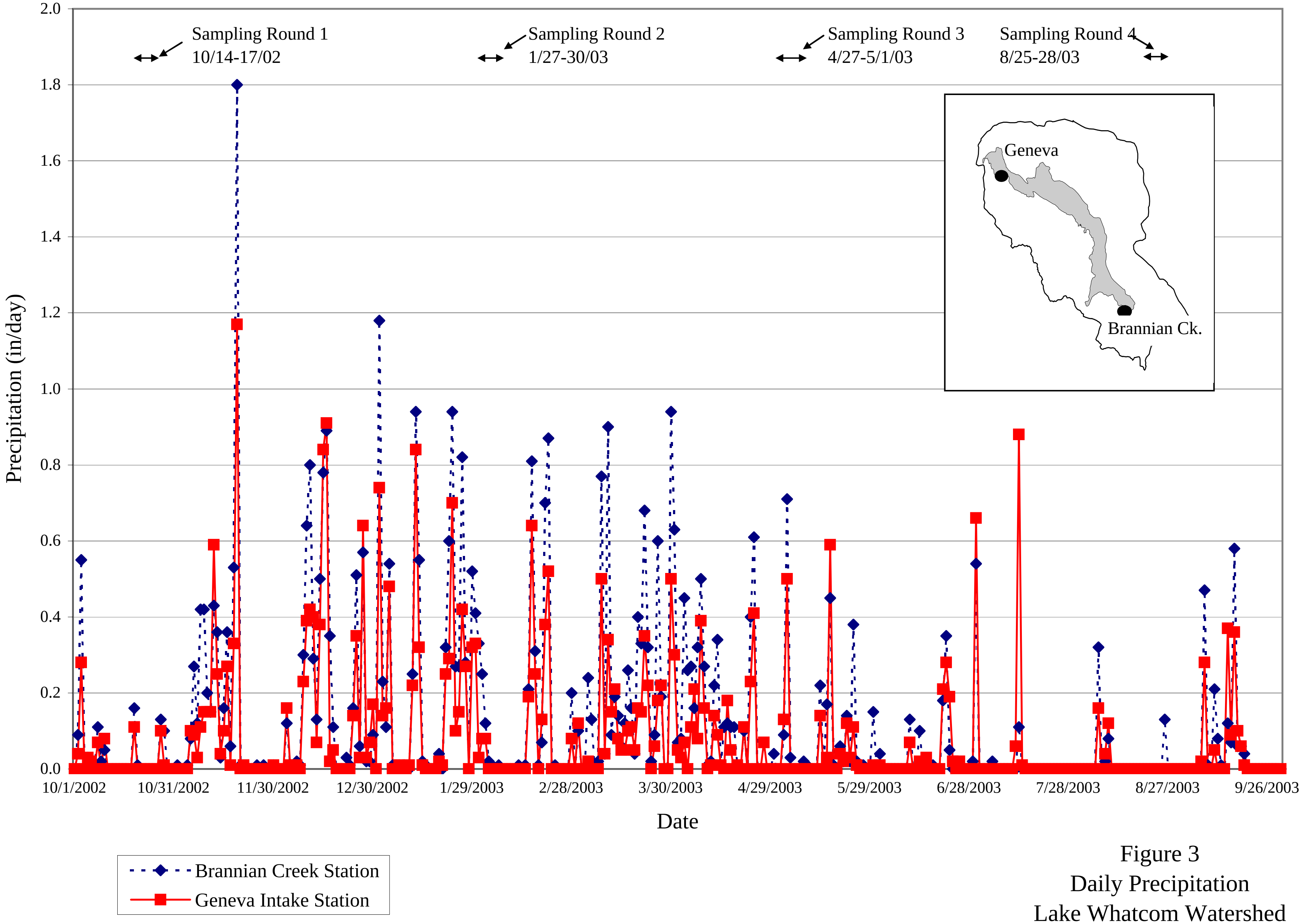


Figure 3
Daily Precipitation
Lake Whatcom Watershed

Geologic Setting

Lake Whatcom is situated in the westernmost portion of the North Cascade Range physiographic province. The study area geology is dominated by two principal bedrock units: the Jurassic-age Darrington Phyllite (present in the southern third of the watershed), and a thick sequence of deformed sedimentary strata of the Eocene-age Chuckanut Formation (present throughout the northern portion of the study area) (Figure 5). Deposits of unconsolidated, Pleistocene-age glacial sediments unconformably overlie the bedrock units to varying thickness in low-lying portions of the study area, most notably in the Squalicum Valley area and in the South Bay and Anderson Creek valleys. Localized deposits of Holocene to very-late Pleistocene-age alluvial, fan-alluvial, and landslide deposits are also present within the watershed (Dragovich et al., 2002; Easterbrook, 1971; Johnson, 1982; Jones, 1999; Lapen, 2000; WDNR, 2003; Easterbrook, 2004).

The Darrington Phyllite is comprised predominantly of interlayered, graphitic-phyllite and quartzose-graphitic-phyllite, derived from low-temperature metamorphism of sedimentary bedrock (mudstones and quartzose mudstones). The phyllites exhibit multiple foliations and lineations, and are frequently cut by or interlayered with quartz veining or lenses.

The Chuckanut Formation sediments unconformably overlie the Darrington Phyllite, with the contact exposed at the surface in the southeastern portion of the watershed. Two identifiable members of the Chuckanut Formation are present over the northern two-thirds of the study area: the early to mid-Eocene Bellingham Bay Member, and the middle to late-Eocene Padden Member (Figure 5).

Both Chuckanut units are comprised of interbedded intervals of coarse-grained and fine-grained, continentally-derived fluvial deposits. The coarse-grained intervals are comprised principally of cross-bedded or laminated sandstone, with lesser poorly- to moderately-well-sorted conglomerate and conglomeratic sandstone. The finer-grained intervals are typically comprised of massive or thinly-laminated siltstones and mudstones, interbedded with minor layers of coal or black shale.

The Chuckanut strata are typically arranged in irregular, repetitive fining-up sequences, with fine- and coarse-grained intervals alternating up to 50 meters thick. Following deposition, late Eocene regional tectonic forces deformed both members of the Chuckanut Formation into a series of north- and northwest-trending open folds. These folds, now truncated by erosion, are well expressed in the topography in the western portion of the watershed (Figures 2 and 5).

Unconformably overlying the bedrock formations are unconsolidated glacial deposits associated with the Fraser Glaciation (~11-14 Ka) of the Pleistocene epoch. In the study area, these include deposits of the Vashon Stade (mapped at the surface in the South Bay and Anderson Creek Valley areas)[1], dominantly fine-grained glaciomarine drift deposits assigned to the Everson

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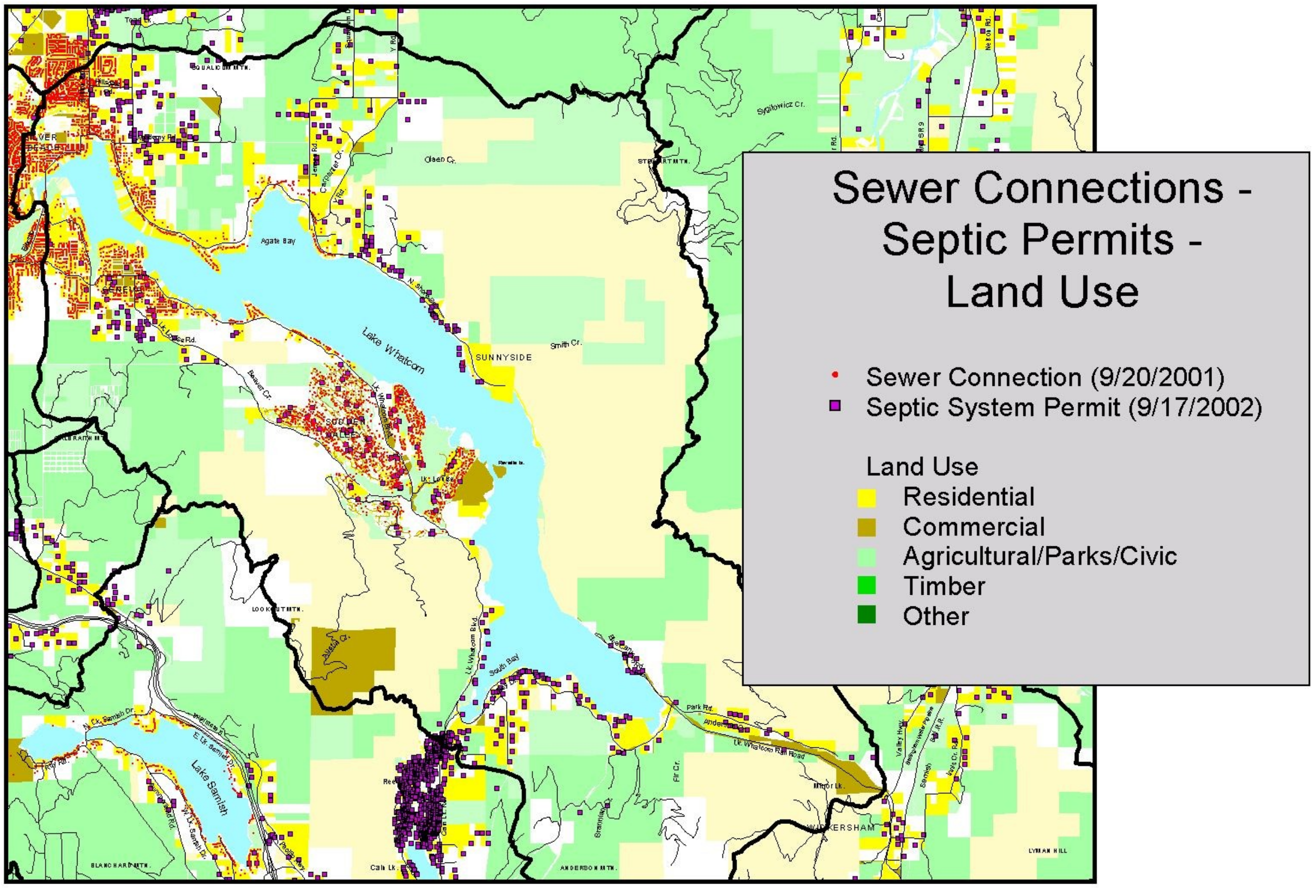
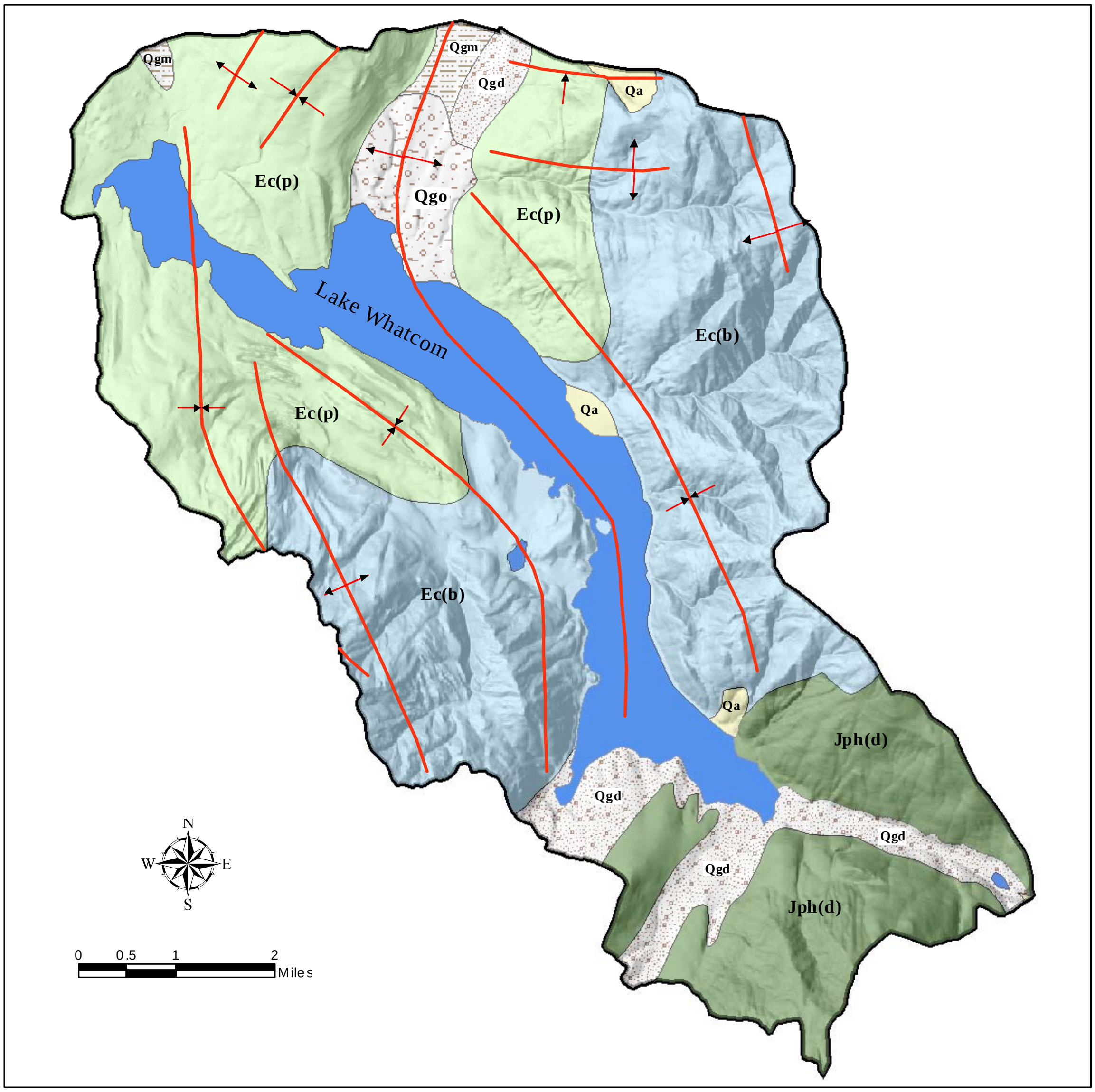
[1] The surface deposits from this area are alternatively interpreted by Kovanen and Easterbrook (2004) as post-ice sheet deposits derived from alpine glaciers extending from the south, or by Jones (1999) as Quaternary alluvium.



Map supplied by M. Arthur, Whatcom County Health Department, 2002

Figure 4
Land Use and
Sewage Management



Geologic Map Source: WDNR, 2003 (Digital Data, 1:100,000)

Legend

Folds (showing dip of bedrock)

🟦 Water

Holocene Sediments

Qa Alluvium, Fan Deposits, Land slide deposits

Pleistocene Glacial Sediments

Qgo Glacial Outwash (Sumas)

Qgm Everson Glaciomarine Drift

Qgd Undifferentiated Glacial Drift

Eocene Sedimentary Bedrock

Ec(p) Chuckanut Fm. (Padden M.)

Ec(b) Chuckanut Fm. (Bellingham Bay M.)

Jurassic Metamorphic Bedrock

Jph(d) Darrington Phyllite

Figure 5
Study Area
Geologic Map

Interstade (present at the surface at the northern end of the Squalicum Valley), and dominantly coarse-grained outwash (recessional) deposits assigned to the Sumas Stade (present at surface at the southern end of Squalicum Valley adjacent to Agate Bay)(Figure 5).

In addition, unmapped, patchy deposits of mid- to late-Pleistocene till and glaciomarine deposits mantle the bedrock hills throughout the remainder of the watershed (ranging in thickness from a thin veneer to many meters thick) or underlie younger glacial deposits in the valley areas (Easterbrook, 2004). The glacial deposits encountered in the study area are primarily remnants of a continental ice sheet that periodically over-rode and retreated from the watershed, alternatively scouring, deepening, and depositing sediments within the Lake Whatcom basin.

Drilling logs from the Squalicum Valley area indicate an alternating series of dominantly coarse- and fine-grained glacial deposits to a depth of at least 160 feet below ground surface. At the southern end of the valley, the surface unit of coarser-grained outwash sediments (missing from the northern end of the valley) overlies a thicker sequence of predominantly fine-grained deposits. The finer-grained strata, which are frequently interlayered with minor, coarser-grained lenses, are likely comprised of glaciomarine deposits overlying a till base. The presence of till at depth is supported by driller descriptions referencing units of intermixed gravel and clay, cemented gravel with clay, and hardpan. Underlying the till in many of the borings from this area is a unit of moderately coarse, water-bearing gravels and sands (advance outwash?) beginning at a depth of approximately 100 to 130 feet below ground surface.

Drilling logs from the South Bay area indicate similar alternating coarse- and fine-grained glacial deposits, at least 120’ thick, also with frequent mention of hardpan or inter-mixed gravel-and-clay units at intermediate depths. Water-bearing sand and gravel deposits are routinely noted on boring logs from this area between 60-100 feet below ground surface. In the Anderson Creek valley, boring logs indicate a maximum thickness of 65-75 feet of interbedded, heterogeneous sand, gravel, and clay deposits overlying the bedrock substrate.

Two map-scale, Holocene-age deposits of note are present in the study area. A large alluvial-fan deposit is present immediately adjacent to the lake in the Sunnyside area, and a landslide deposit (perhaps very-late Pleistocene?) is present in the Blue Canyon area (Figure 5). These deposits are characterized by heterogeneous, poorly-sorted mixtures of clays, silts, sands, gravels, and cobbles reworked from the underlying formations. The total thickness of these deposits is presumed to be relatively limited. Additional unmapped Holocene-age deposits are frequently encountered at the mouths of creeks entering the lake (e.g., silty sands distributed about the mouth of Austin Creek), often fanning over the lake bottom.

Soils

Soils and soil character encountered in the study area are closely related to the underlying substrate. For example, those soils developed directly over bedrock are typically less than 40-60 inches deep, moderately well-drained, contain a comparatively high percentage of lithic fragments of the underlying rock type, and often show very low permeabilities in the lowermost horizons (e.g., the Andic Xerochrepts-Rock Outcrop complex of the southern mountains) (USDA, 1992).

Many of the soils in the watershed were formed over a mixture of glacial deposits and bedrock (e.g., the Chuckanut loam and Squalicum gravelly loam series found throughout much of the northern portion of the watershed). These soils, which are formed in a mixture of volcanic ash, colluvium, slope alluvium, and loess, tend to be deep (40-60 inches) and well-drained to moderately well-drained. Soil permeabilities are moderate in the upper horizons, but frequently very low in the lowermost horizons.

In contrast, soils developed over alluvial deposits, or deeper coarse-grained glacial deposits (e.g., the Everett series very gravelly sandy loams of the southern Squalicum valley area) are typically deep to very deep, very well to excessively well-drained, and have rapid to very rapid permeabilities, often increasing with depth. Weakly compact glacial till, or cemented pan, is occasionally encountered at the base of the soil profile. Soils developed over deep, finer-grained glaciomarine drift (e.g., the Whatcom series silt loams of the northern end of Squalicum valley) tend to be deep to very deep, moderately well-drained loamy soils with moderate to poor permeabilities.

Groundwater Hydrology

Hydrogeologic Framework and Flow Processes

The geologic setting of the study area strongly dictates the occurrence and character of groundwater flow and interaction with the lake. The local groundwater system can be divided for the purposes of this study into two principal hydrogeologic flow regimes: 1) flow occurring within the pre-Pleistocene bedrock (and associated overburden sediments) and 2) flow occurring within the unconsolidated valley-fill deposits. The bedrock-dominated portions of the groundwater system are best characterized by dual-porosity flow (fracture flow) processes, and as a result have a restricted capacity to transmit (and supply) groundwater. In contrast, porous-media flow processes prevail in the larger valleys of the study area that are filled by unconsolidated sediments. While these deposits cover only a small minority of the basin surface area (Figure 5), they are highly transmissive in contrast to the bedrock. As a result, the valley sediments play a key role in the transport of groundwater to the lake.

In dual-porosity groundwater flow regimes, the *primary* porosity of bedrock is typically extremely low, limiting groundwater flow within the rock matrix to low velocity diffusion transport (Anderson and Woessner, 1992). In the case of metamorphic bedrock such as the Darrington Phyllite, it is the consolidating effects of heat and pressure, and the resulting recrystallization of the source rock, that produces this low porosity condition. For sedimentary bedrock such as the Chuckanut strata, matrix porosity is reduced by compaction and cementation of the sediment particles. Previous investigators have noted the low primary porosity character of the study area bedrock in their descriptions (Newcomb et al., 1949; Easterbrook, 1976; Johnson, 1982; Lapen, 2000).

Groundwater flow related to *secondary* porosity, i.e., flow related to fracture planes, joints, and weathered or sheared zones, is often where the bulk of water movement occurs in bedrock. The secondary porosity (and permeability) of bedrock is determined by the number, width, and interconnectedness of the structural planes, as well as the degree of secondary alteration or infilling in the structure apertures. In many cases, offsets along fracture or joint planes can act as barriers to continuous groundwater flow.

The available evidence suggests that the secondary porosity and permeability of the study area bedrock is also limited. No large-scale shear zones or faults have been mapped within the study area, and most bedrock fractures or joints appear to occur as discontinuous, mesoscopic scale features (Easterbrook, 1971; Lapen, 2000; Johnson, 1982). While Newcomb et al. (1949) did note that limited well yields are possible in some of the fractured shales present in the study area at depth, they concluded that, overall, the bedrock formations present in the Lake Whatcom area have limited capacity for groundwater movement and supply (see also additional descriptions of area bedrock hydraulic properties in Cox and Kahle, 1999).

This interpretation is supported by the small number of water supply wells that have been successfully developed in the areas mapped as bedrock on Figure 5. A significant percentage of the boring logs reviewed from these areas indicate abandonment of the borehole after drilling due to a lack of adequate yield, or to the presence of connate water unsuitable in quality for domestic water supply (implying low permeability conditions preventing infiltration of surface recharge of fresh rainwater or snowmelt).

Those wells that are successfully developed in the bedrock areas are typically completed in the thin, low-permeability glacial deposits that mantle the Chuckanut bedrock at the northern end of the study area. The yield recorded for the majority of these wells is relatively small, mainly suitable for single household use, with significant drawdown response to pumping. Analysis of specific capacity test data from a limited set of these wells suggests horizontal hydraulic conductivities for the adjacent aquifer material of typically less than 10 feet per day. Few completed wells were identified within the metamorphics of the Darrington Phyllite.

The large majority of the wells completed in the watershed are concentrated in the valley-fill deposits encountered in the Squalicum, South Bay, and Anderson Creek valleys. The productivity of these sediments reflects the transmissive character of the deposits, and is evidence that significant groundwater flow to the lake is likely focused in these areas.

The valley-fill deposits can be divided into alternating pairs of loose, coarser-grained aquifers, and compact, finer-grained semi-confining units. The coarse-grained outwash deposits encountered beneath the till in the Squalicum and South Bay valleys joining the lake are the most hydraulically productive aquifers of the study area. Estimated horizontal hydraulic conductivity values for these deposits range between 100-1000 ft/day (many of the deeper wells from the Squalicum Valley area reportedly exhibited artesian flow at the time of construction). A limited number of wells are alternatively completed in coarse-grained intervals or lenses occurring above or within the till units; the horizontal hydraulic conductivity values estimated for these intervals are typically between 10 and 100 ft/day, and well yields appear to be largely limited to single household supply.

Flow Field

With a comparatively small number of wells in the study area, particularly in bedrock-dominated terrane, the development of detailed maps of groundwater water-level elevations and groundwater flow boundaries is difficult. However, based on the descriptions discussed above, Figure 6 presents a conceptual model of the study area groundwater flow field. The scaling of the flow arrows presented on the figure reflects the relative magnitude of flow within the basin; the largest flows are concentrated in the areas of unconsolidated deposits, while groundwater movement through the bedrock terrane is minimal. Flow lines in the bedrock areas are generally inferred from topography and structure; in reality, flow directions in these areas are likely more complex than shown.

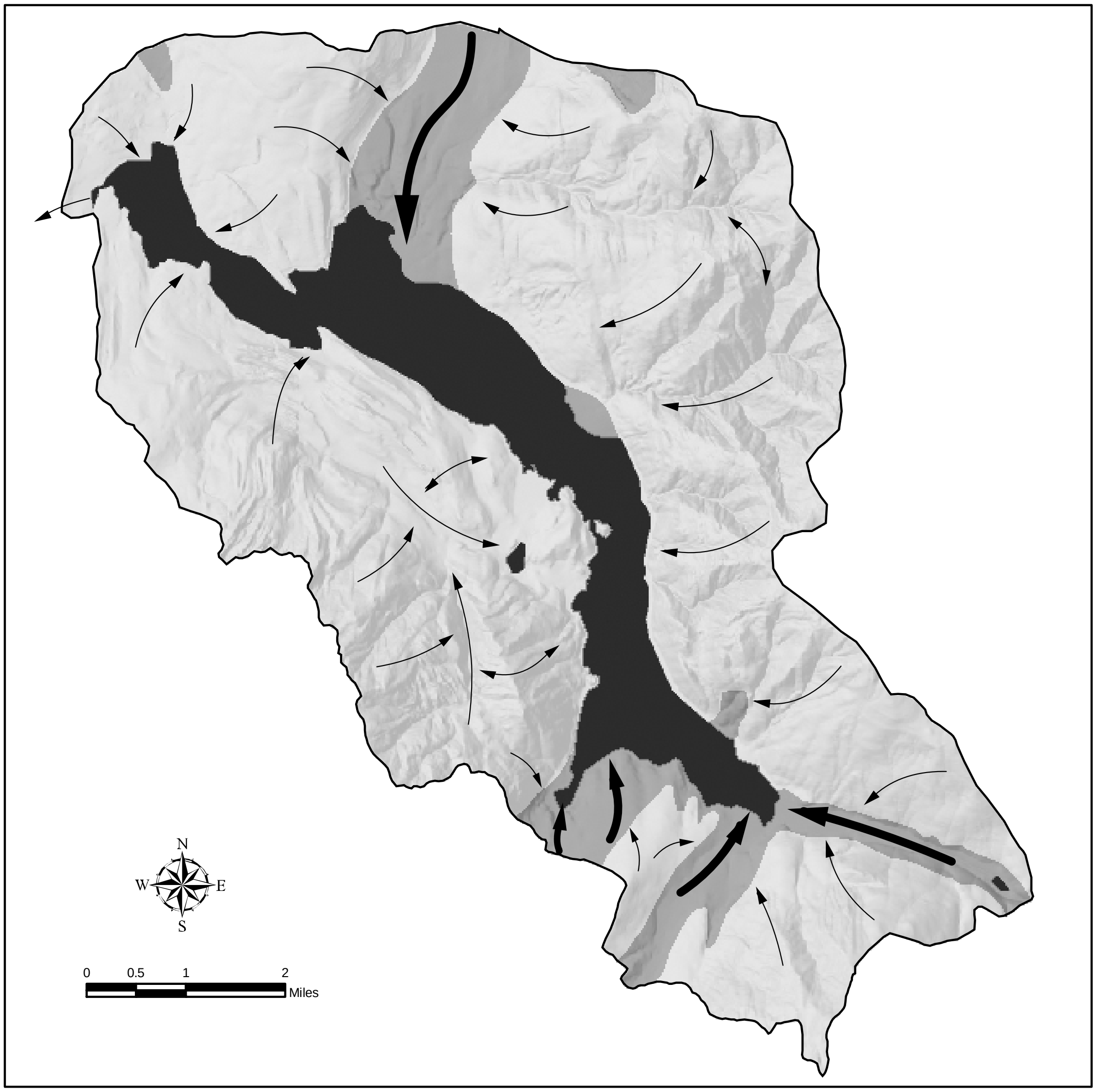
For the purposes of this study, it is assumed that the extent of the groundwater flow system contributing to and interacting with the lake approximates the boundaries of the surface watershed. The available data support this assumption for the Squalicum, South Bay, and Anderson Creek valleys, where static water elevations reported in wells at the upper valley ends indicate groundwater divides coincident with the watershed boundary. The absence of well data and the complexity of structurally-related groundwater flow in fractured bedrock prevent an estimation of the true position of groundwater basin boundaries throughout most of the remainder of the study area. However, in view of the hydraulic characteristics of the regional bedrock, it is assumed that deep inflow from areas outside of the watershed boundary is negligible.

Water enters the aquifer system via recharge by direct infiltration of precipitation (or snowmelt), and flows vertically and laterally downgradient along pathlines through both the unconsolidated deposits and the bedrock. Lake Whatcom is assumed to be the ultimate point of discharge for the majority of the study area groundwater, although some portion of the annual recharge may discharge as baseflow to watershed surface streams upslope of the lake (particularly in bedrock areas) or may exit the basin via deep regional groundwater flow. A small volume of deep lateral inflow to the valley aquifers from adjoining bedrock areas probably occurs (Figure 6). Groundwater elevations from wells located in the far northwestern portion of the watershed suggest that there is a limited amount of groundwater underflow discharging out of the basin in the area of Whatcom Creek. The magnitude of this outflow is unknown.

Groundwater Recharge

Recharge rates to the study area groundwater system are strongly influenced by the underlying geologic substrate. For example, the hydraulic characteristics that inhibit lateral movement of groundwater through the bedrock also limit the volume of water that can enter the subsurface by downward infiltration. Infiltration to the bedrock is likely further limited by the low permeability of the glacial sediments and soil profiles that mantle these older formations throughout much of the study area.

Most precipitation that falls over the bedrock areas probably remains as surface runoff or as shallow interflow at the soil-bedrock interface, with limited infiltration to the water table through



➜ Inferred Groundwater Flow Direction - (Arrow scale reflects relative magnitude of flow)

▨ Unconsolidated valley-fill deposits

□ Bedrock terrane

Figure 6
Conceptual Model
of Study Area
Groundwater
Flow Field

fracture and joint planes. Shallow interflow would likely result in short distance flow paths with discharge as baseflow to local streams, or as springs and seeps on mountainsides during high precipitation periods.

No estimates of the groundwater recharge rate to the bedrock portions of the study area have been published; therefore, for the purpose of calculating groundwater discharge and loading estimates later in this report, a mean annual recharge rate of 2 inches/year is assumed for the areas mapped as bedrock on Figure 5. This value is consistent with previously published estimates for recharge for similar geologic settings in the region (Orr et al., 2002; Hodge, 1995; Foweraker, 1974).

Annual recharge rates for the more transmissive unconsolidated deposits present in the study area have been estimated as part of the U.S. Geological Survey Regional Aquifer-System Analysis (RASA) of the Puget Sound Aquifer System (Vaccaro et al., 1998). Figure 7 presents a map of estimated recharge rates derived from the RASA data for the study area (recharge rates for the bedrock portion of the basin were not estimated during the RASA analysis and are therefore assumed as 2 inches/year). The figure shows that mean annual recharge rates for these deposits are predicted to range between approximately 7 to 50 inches/year, with the highest estimates encountered in the valley-fill deposits of the South Bay area. The RASA recharge values are derived from a very large-scale study, and are intended to represent average annual conditions. The RASA values are not necessarily an exact prediction of the recharge rate of the local unconsolidated sediments for any given water year. However, the RASA values do reflect the typical annual range and relative magnitude of recharge estimated for deposits of these types.

Groundwater Inflow to Lake Whatcom

Three-dimensional groundwater flow fields in the vicinity of lakes can be highly complex, complicating the prediction of the point of discharge for any individual water particle entering the surrounding aquifer system (Winter, 1978). Many authors have shown, however, that the bulk of groundwater inflow to lakes is ultimately concentrated within a narrow, near-shore zone via upward and lateral seepage from the adjacent aquifer system through lake bed sediments. As a result of the upward bend in groundwater flow lines immediately adjacent to lakes, the rate of groundwater inflow typically declines exponentially with distance from shore (Lee, 1976; Winter, 1978; Wagner et al., 1983; McBride and Pfannkuch, 1975; Harvey et al., 2000; Lee et al., 1980).

Groundwater seepage to a lake can be a highly localized phenomenon laterally, being strongly influenced by small-scale heterogeneities in the hydraulic character of the bottom sediment. In addition, geologic controls could, in some cases, cause submarine inflow deeper (further from shore) than is normally observed. One example that may be applicable to Lake Whatcom is fracture-controlled flow through bedrock intersecting the lake at depth[2]. The extension of a confining unit (such as a deep till deposit) to the lake shoreline could also extend and deepen the focus of inflow at the end of the main study area valleys.

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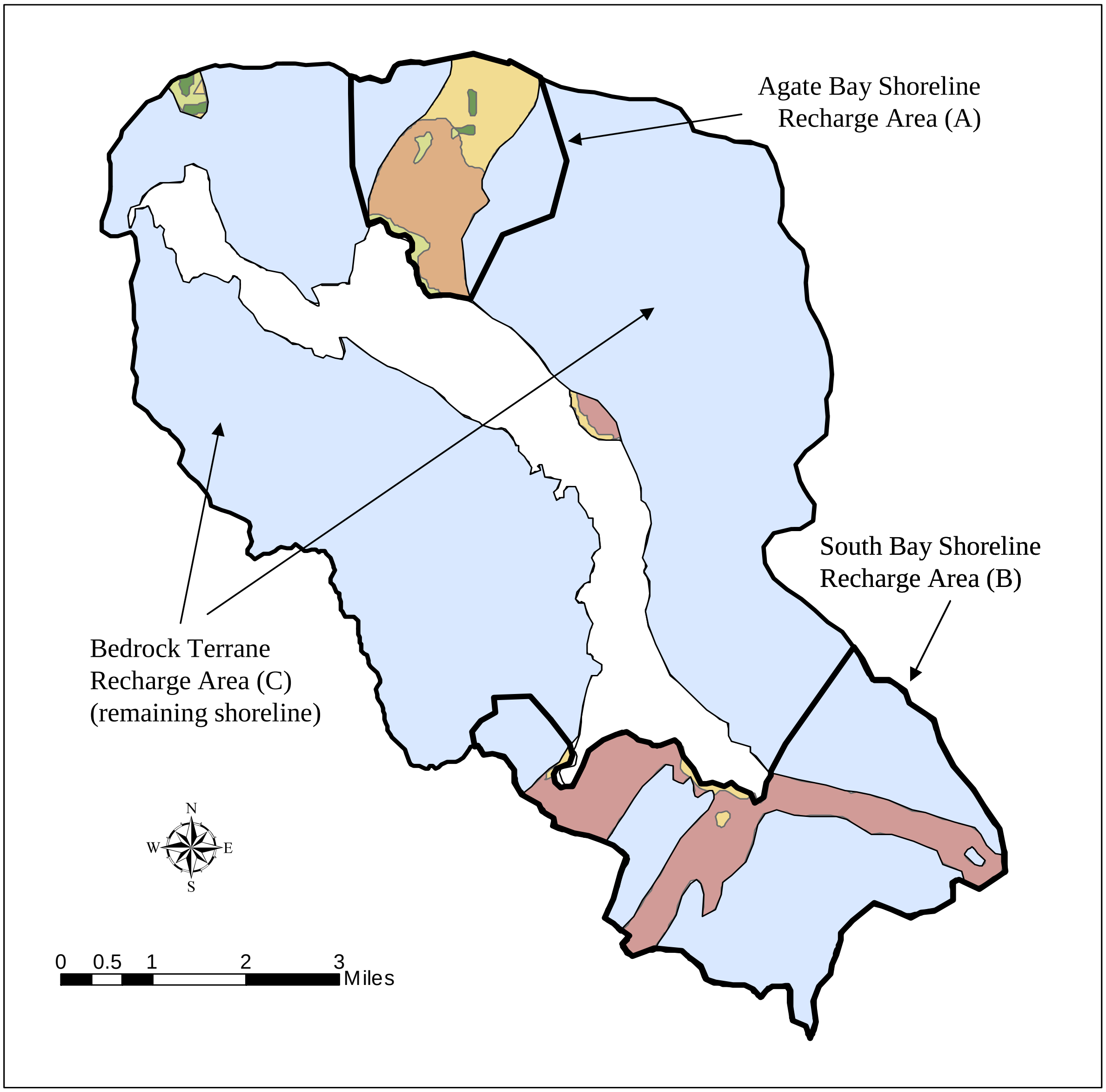
[2] Groundwater discharge to the lake from a discrete fracture plane would have a tendency to be diffused in areas where a layer of bottom sediments covers the bedrock.



Source: Vaccaro et al., 1998 (Digital Data)

Estimated Mean Annual Recharge

- ▭ 2"/yr (Bedrock - assumed value)
- ▨ 7"-11"/yr
- ▨ 12"-16"/yr
- ▨ 17"-21"/yr
- ▨ 27"-36"/yr
- ▨ 37"-50"/yr

Figure 7
Map of Estimated
Groundwater Recharge

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Given the complexity of lake/aquifer interactions, and the lack of reliable data on aquifer conditions for the majority of the study basin, a water-balance approach was used to estimate the volume of groundwater inflow to Lake Whatcom during the 2002-2003 study period. Figure 8 illustrates the conceptual model used for this analysis. Several simplifying assumptions are necessary to apply this approach:

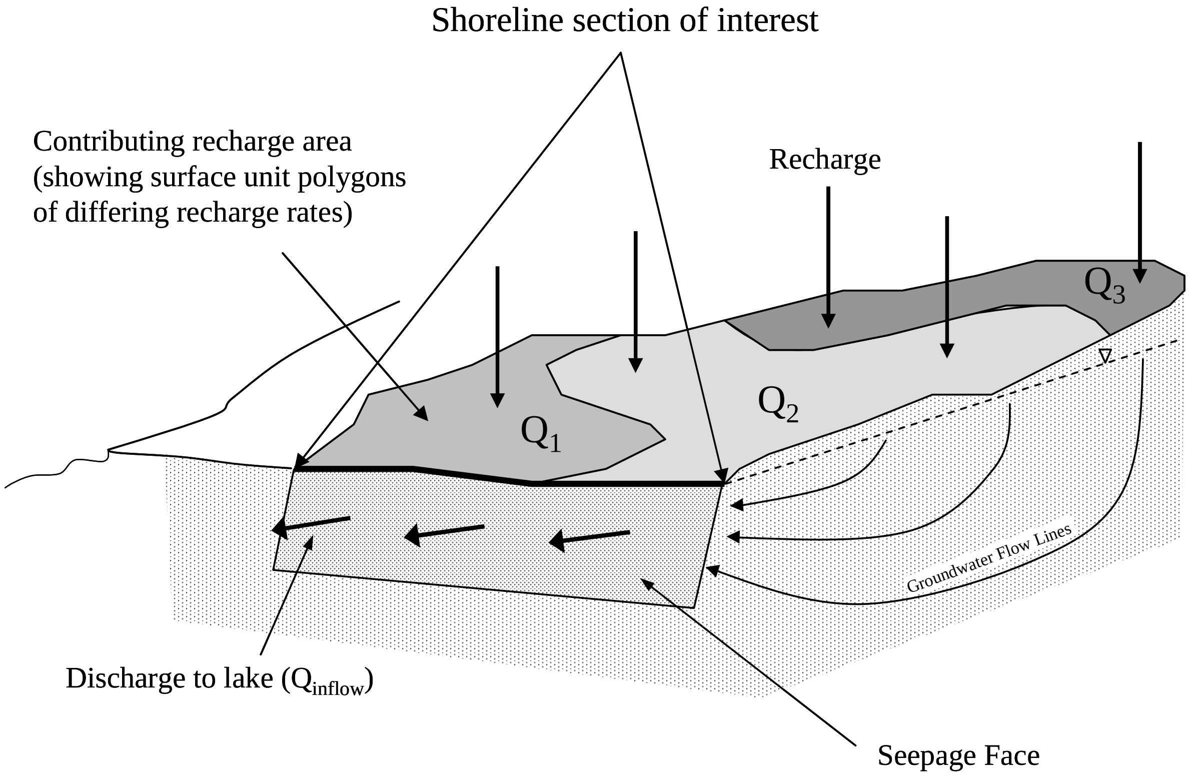
- Since it is impractical to determine the presence and specific locations of stratigraphically or structurally controlled deeper inflow on the lake bottom, it is assumed that the large majority of the groundwater discharging from the aquifer system to the lake flows through a narrow, hydraulically homogeneous, and isotropic seepage face lying immediately beyond the lake shoreline.
- On an annual basis, it is assumed that groundwater discharge occurring within the study area is in dynamic equilibrium with recharge; i.e., there is no effective change in storage within the aquifer system over the long-term.
- All recharge reaching the aquifer is assumed to discharge directly to the lake.

The final assumption probably biases the inflow estimates high. As discussed above, a portion of the study area annual recharge likely discharges as baseflow to surface streams (or fracture-related springs) upslope of the lake shoreline. Analysis of dry-season, baseflow-condition surface flows reported by Matthews et al. (2003, 2004) indicates that during the summer this assumption may result in an approximately 15-20% overestimate of inflow from the bedrock areas, and a significantly smaller overestimate (<1%) in areas dominated by unconsolidated deposits.

If the regional water table in the bedrock areas is raised during the wet season to a level that significantly increases the intersection with surface drainages, discharge of groundwater prior to reaching the lake could increase significantly, resulting in even larger overestimates of inflow from this terrane. An unknown volume of recharge may also be exiting the basin as deep regional flow. These factors indicate that assuming all recharge is directed towards the lake probably provides an upper-bound estimate of the actual groundwater inflow volume.

Groundwater inflow estimates were developed for three major sections of the lake's shoreline: 1) the shoreline adjacent to the unconsolidated deposits of the Squalicum Valley, 2) the shoreline adjacent to the unconsolidated deposits of the South Bay/Anderson Creek valley areas, and 3) the shoreline for the remainder of the lake (adjacent to the area identified as the *Bedrock Terrane*) (Figure 7).

To quantify inflow volumes for each shoreline section, the lateral boundary of the upgradient recharge area likely contributing flow to each section was first identified (Figure 7). Topographic relief was used as a guide to estimate the area of bedrock adjacent to the valleys that were contributing inflow to the glacial deposits, versus direct groundwater discharge to the lake shoreline.



$$Q = \text{Recharge Rate} * \text{Surface Area}$$

$$\text{Annual Recharge } (Q_{recharge}) = Q_1+Q_2+Q_3$$

$$\text{Assumption: } Q_{recharge} = Q_{inflow}$$

Figure not to scale

Figure 8
Conceptual Diagram - Groundwater Inflow Estimate Method

After mapping the boundary of the recharge area for each shoreline section, a GIS software program was used to calculate the surface area of the individual RASA polygons inside the boundary. The surface area of each polygon was then integrated with an appropriate recharge rate to estimate a recharge volume. The recharge volumes for all polygons in a recharge area were summed to provide a groundwater inflow value to the lake for the given shoreline section (Figure 8).

For comparison purposes, two different scenarios were run for each shoreline section of interest. Due to the fact that the 2003 water year was an exceptionally dry year (i.e., not reflective of mean annual conditions), the *minimum* value of the appropriate RASA recharge range was assumed for each polygon for Scenario 1. For example, if the RASA annual mean recharge rate for a given polygon ranged from 12 to 16 inches, the 12-inch value was selected. Table A-1 in Appendix A presents the detailed data and assumptions used for the Scenario 1 analysis. Since RASA recharge rates are only available for the unconsolidated deposits, an annual recharge rate of 2 inches was assumed for all bedrock polygons.

Because the RASA values are intended to represent mean annual conditions, the lowermost values of the recharge range may still overestimate recharge volumes in a very dry year such as occurred during the study period. To address this concern, a second scenario was run that substitutes the RASA recharge rate values for each polygon with recharge rates estimated as a percentage of the study period precipitation total (Scenario 2; Table A-1). The following assumptions were used for Scenario 2:

- The total precipitation for the 2003 water year over the Agate Bay (A) and Bedrock Terrane (C) recharge areas was assumed to be 31 inches (Mitchell, 2004)(Figure 7).
- The total precipitation for the 2003 water year over the South Bay recharge area (B) was assumed to be 43 inches (Mitchell, 2004).
- The recharge rate for predominantly fine-grained surface deposits was assumed to be 30% of the precipitation total.
- The recharge rate for predominantly coarse-grained surface deposits was assumed to be 50% of the precipitation total.
- The Scenario 2 recharge rate for bedrock units is assumed to be 2 inches for the 2003 water year. No downward adjustment to the recharge rate for bedrock was made due to the assumption that infiltration rate, not precipitation, is the limiting factor for recharge to these units.

Table 1 presents a summary of the results for the inflow analysis. The estimated 2003 water year groundwater inflow volume to the lake predicted by the two scenarios ranges from 1.23E+10 to 1.71E+10 liters. The comparatively transmissive nature of the unconsolidated deposits is reflected in the values presented in Table 1. Despite the limited areal extent of these deposits, the combined estimated groundwater inflow to the Agate Bay and South Bay/Anderson Creek shoreline sections represents between 62-72% of the total groundwater inflow estimates.[3]

The Table 1 groundwater inflow totals represent approximately 21-29% of the total *combined* surface water and groundwater inflow calculated using water balance methods by Western Washington University for the 2003 water year (5.9E+10 liters; Matthews et al., 2004; Mitchell, 2004). This percentage range is equivalent to or higher than previously published estimates of the role of groundwater in lake water budgets in roughly similar geologic settings (e.g., Wagner et al., 1983; Winter, 1989). A higher proportional role for groundwater inflow in the total lake inflow budget would be expected during very dry periods such as the 2003 water year; presumably the relative role of groundwater would decrease under average or above-average precipitation conditions when surface runoff volumes increase.

Table 1. Lake Whatcom Groundwater Inflow Estimate Summary for Water Year 2003

Shoreline Section	Shoreline Length (m)	Estimated Upgradient Recharge Area (m^2)	Scenario 1	Scenario 2
			RASA-Based Recharge/Inflow Volume Q_{RASA} (liters/yr)	WY '03 Precipitation-based Recharge/Inflow Volume Q_{PRECIP} (liters/yr)
A - Agate Bay Shoreline	2796	1.12E+07	3.80E+09	2.30E+09
B - South Bay Shoreline	6201	2.71E+07	8.42E+09	5.32E+09
C - Bedrock Terrane (remaining shoreline)	35468	7.76E+05	4.83E+09	4.63E+09
Total (A+B+C)	44465	3.91E+07	1.71E+10	1.23E+10

Notes

1. Estimates assume equilibrium conditions, i.e., no change in storage over time (annual recharge = annual discharge)
2. Area values assume the outer boundary of the groundwater recharge basin is equivalent to the surface drainage boundary for the Lake Whatcom watershed.
3. Inflow estimates assume all recharge discharges to Lake Whatcom (annual discharge = annual inflow to lake).
4. See Table A-1 and Figure 7 for recharge area and shoreline locations and assumed recharge rates.

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[3] On a volume basis, interception of recharge by pumping wells is considered a negligible factor when calculating the discharge volume estimates. In comparison to the values presented in Table 1, withdrawals probably represent less than 3% of the total discharge volume, and it is assumed that most water withdrawn from such wells is ultimately returned to the aquifer system via re-infiltration. The volume of water imported into the watershed or withdrawn from the lake, used, and released back to the aquifer system is also assumed to represent a negligible fraction of the total estimated discharge to the lake. For example, assuming 750 permitted septic systems within the study area (Arthur, 2002), and an average release rate of 113 gallons/day/system (EPA, 1980), the extra recharge volume to the study area aquifer system would represent less than 1% of the total estimated discharge values in Table 1.

Methods

Monitoring Network

To identify potential locations for sampling the quality of groundwater discharging to the lake, a boat-based reconnaissance survey was conducted in September 2002. In light of typical groundwater seepage patterns to lakes, the survey focused on the near-shore zone. The lake-level elevation at the time of the survey was approximately 312 feet above mean sea level. Large portions of the lake's shoreline and near-shore zone are comprised of exposed or near-surface bedrock, a condition which prevents installation of groundwater monitoring devices. As a result, areas of the lake where unconsolidated sediments overlie the bedrock to an adequate thickness for monitoring equipment installation were favored as sampling locations.

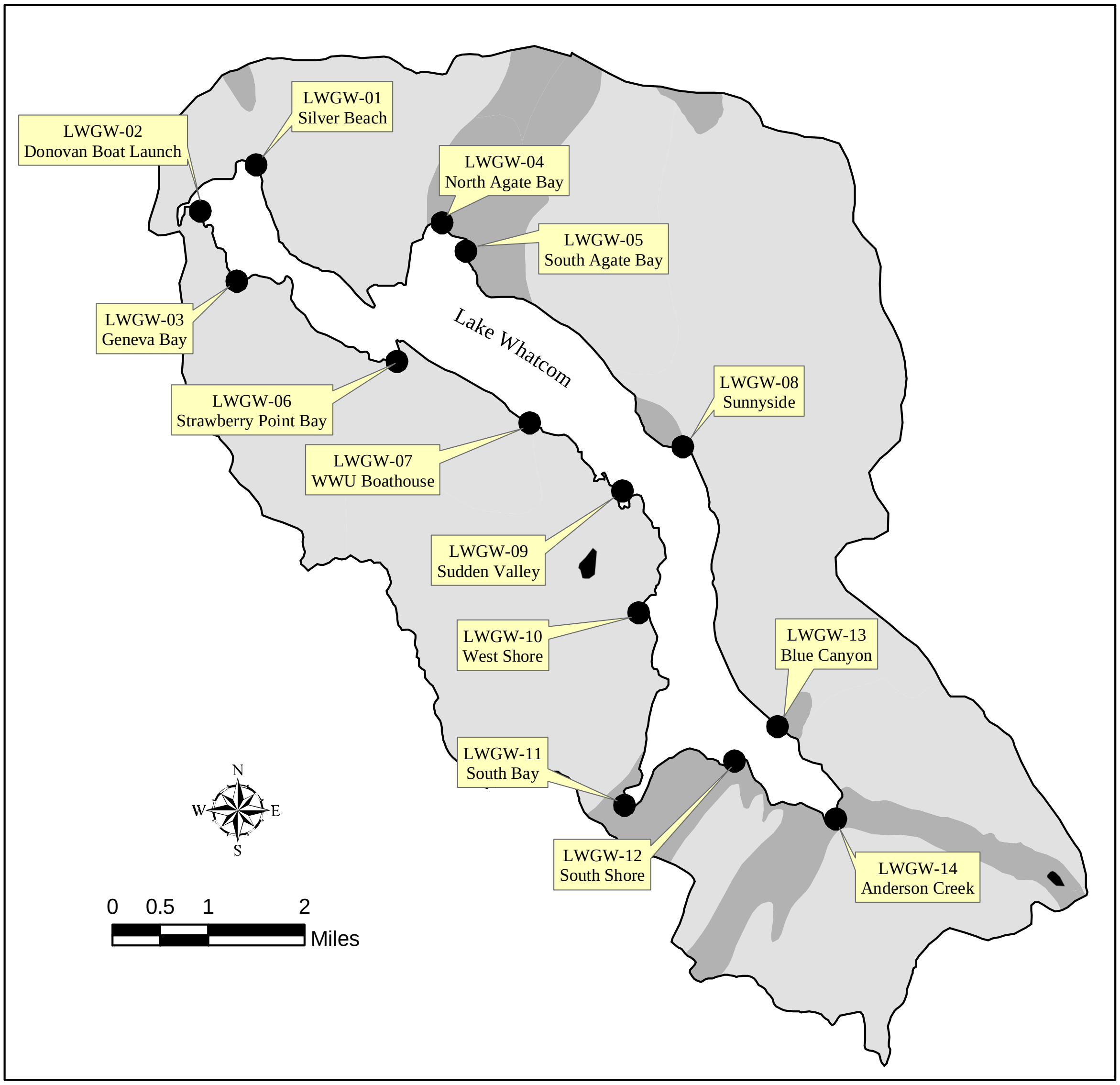
A network of 14 lake-bed piezometers was installed to characterize the concentration and distribution of nutrients in groundwater discharging directly to the lake. Stations were distributed widely around the lake to help characterize the water quality condition downgradient of differing land uses and geologic settings (Figures 4, 5, and 9).

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 the land-use patterns upgradient of each station. Particle-size data and classifications of the surficial sediments occurring at each station are summarized in Table C-1 in Appendix C.

Piezometers were constructed and installed using methods modified from Lee and Harvey (1996). To construct the piezometers, a steel drive point was securely attached to a 10 to 12 foot length of clean ¼" I.D. rigid, translucent polyethylene tubing via a barbed fitting (Figure 10). 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 a clean, 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 selected locations in the lake bed during October 7-9, 2002. 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 shoulder of the drive point. The pipe and tubing assembly was then driven to a depth of approximately 4 to 5 feet below the sediment/water interface, using a tripod and drop hammer. Piezometers were normally installed between 20 and 100 feet from the lake shoreline (partly dependent on the lake bottom slope) in water up to 4.5 feet deep.

Once the piezometer was installed to the appropriate depth, the drive pipe was back-hammered and carefully 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 surface sediment adjacent to the tubing was compacted to improve the seal at the sediment/water interface. Between site visits, the upper end of the



● Piezometer Location

▓ Unconsolidated valley-fill deposits

░ Bedrock terrane

Figure 9
Piezometer Location Map

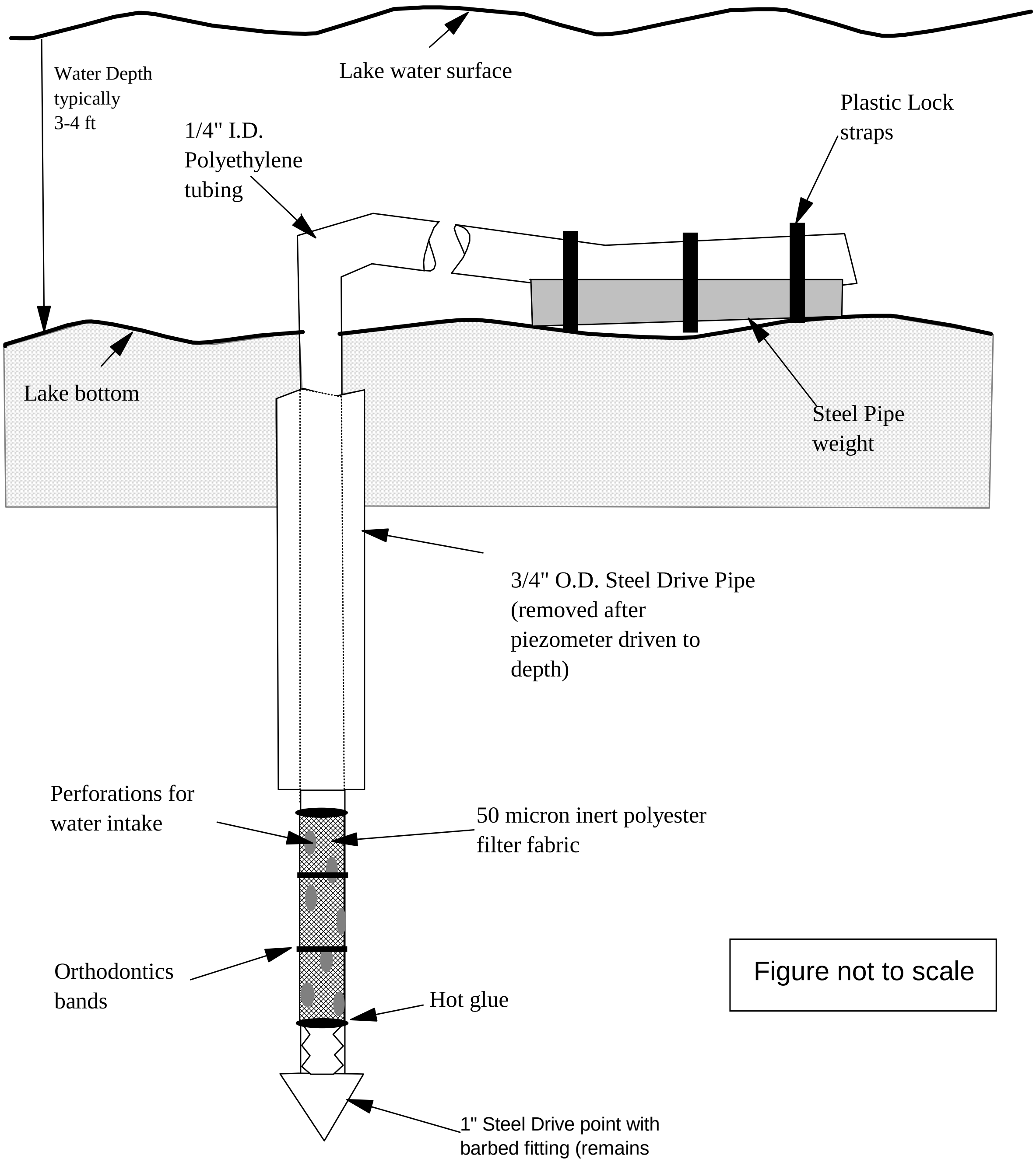


Figure not to scale

Figure 10
Construction Diagram
for Lake-bed Piezometers

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 suspended sediment through several surge cycles (typically 1-2 hours). All piezometers were allowed to equilibrate after development for one week prior to sampling.

Piezometers LWGW-01, -06, -07, -09, -10, and -11 produced limited water for sampling purposes, most likely due to the low permeability of the sediments adjacent to the piezometer intake. As a result, analyses for these stations (particularly LWGW-06) were limited to varying degrees during the course of the 2002-2003 study period. Piezometer LWGW-08 could not be located during the third round of sampling, so no water quality data were collected for that round from the station. LWGW-08 was subsequently relocated and sampled during the final monitoring round. Piezometer LWGW-03 was permanently lost after the third round of sampling, so no water quality data were collected from the station during the final monitoring round.

Sampling and Analysis Procedures

The piezometer network was sampled on four occasions (October 14-17, 2002; January 27-30, 2003; April 28-May 1, 2003, and August 25-28, 2003) to characterize seasonal changes in groundwater quality and vertical hydraulic gradient. Figure 3 identifies these sampling periods in comparison to the study period precipitation data.

Samples were submitted for analysis of target nutrients: total dissolved phosphorus (TDP), orthophosphate as P (orthophosphate-P or OP), nitrate+nitrite as nitrogen (referred to as nitrate-N; nitrite is infrequently detected in shallow groundwater), and ammonia as nitrogen (ammonia-N). In addition, several diagnostic analytes were evaluated to help characterize geochemical controls on nutrient transport (dissolved oxygen, iron, manganese), or serve as source indicator parameters: total dissolved solids (TDS), chloride, dissolved organic carbon (DOC), boron, and methylene blue active substances (MBAS).

Complete details regarding test methods, preservation requirements, sample containers, holding times, and practical quantitation limits for the field and laboratory analytes are presented in Pitz, 2002. Table D-1 in Appendix D summarizes the project analysis methods and reporting limits.

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 11). This comparison provided an estimate of the relative direction and magnitude of the hydraulic potential between the aquifer and 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 a groundwater discharge condition to the lake. A water level in the piezometer below the lake indicates downward infiltration of lake water to the aquifer. A short section of tubing of the same diameter as the piezometer was used as a stilling

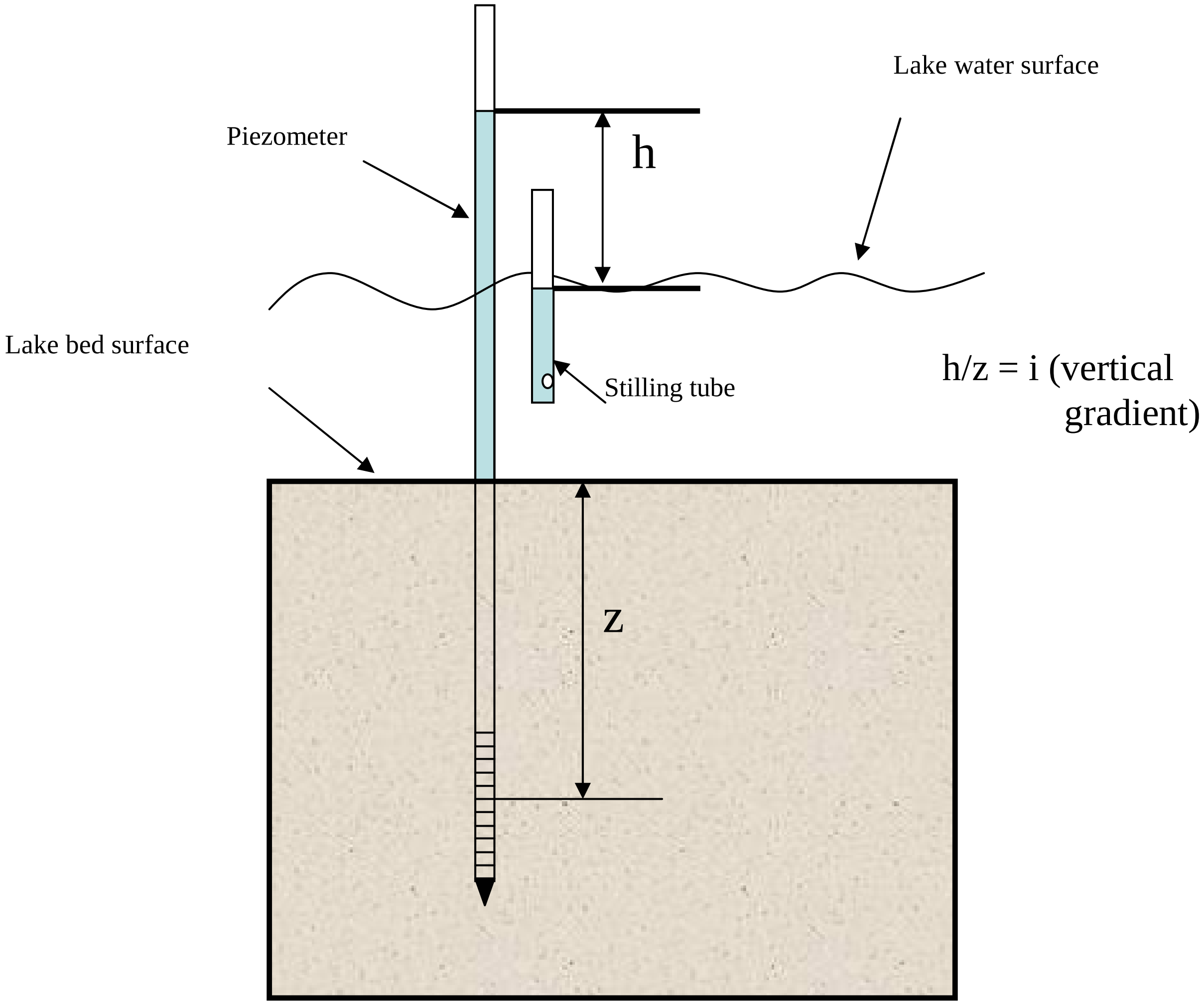


Figure 11
Measurement Method to Estimate Vertical Hydraulic Gradient In Study Piezometers

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 short length of flexible, medical-grade silicon tubing. The intake was placed immediately adjacent to the piezometer just above 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, specific conductance, pH, and dissolved oxygen (DO). The field meters and probes were calibrated daily against known standards, as described in the project quality assurance plan (Pitz, 2002). Temperature measurements were collected for comparative purposes, but are not considered accurate due to atmospheric influences on the flow cell chamber.

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 within the flow cell chamber at five-minute intervals during purging until all parameters had stabilized per the guidelines in Pitz, 2002.

At the end of the purge period, probe-measured temperature, pH, specific conductance, and DO were recorded and compared to the initial lake parameter measurements. During the January, April, and a portion of the August sampling rounds, confirmation analysis of the DO concentration was performed using a field photometer immediately after the end of purging. During the October and August sampling rounds, colorimetric field analysis for MBAS was also performed after completion of the purge. The field measurements for each piezometer (and adjacent lake water) are presented by sample round in Table E-1 in Appendix E.

The piezometer field measurements typically matched lake water conditions at the beginning of purging and then progressively diverged from the lake water results. The conductivity of the groundwater was routinely higher than the lake at the end of the purging period, while the DO concentration was routinely lower. Ongoing comparison of the measured lake values to those from the piezometer served as a field quality assurance check against pumping-induced annular leakage of lake water into the piezometer intake. Upon completion of purging and field measurement, sample collection was initiated.

Sample pumping rates were maintained at less than 0.5 L/min. A clean 0.45 micron in-line filter was attached to the pump outlet. The filter was presoaked with formation water pumped from the piezometer, and the first 200-500 ml of filtrate was discarded prior to filling the sample container. Where applicable, samples were either collected into pre-preserved containers, or preservative was added immediately after sample collection. All samples were quickly placed on ice and transferred to Ecology’s Manchester Environmental Laboratory for analysis.

Quality Assurance

Field

A primary concern for this project was the assurance that the piezometer water samples accurately represent the groundwater in the aquifer. 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 greater than 4 feet) further helped to isolate the intake from the lake.

The field parameter values reported in Table E-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 inflow 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 was also conducted for most of the piezometers as an additional measure of annular leakage of lake water. 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.

Unless otherwise noted, the sampling procedures and equipment 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.

Collectively, these controls suggest that the piezometer samples are representative of groundwater quality conditions immediately beneath the lake.

Analytical

All results reported by Manchester Laboratory were subject to an initial quality assurance review by the lab. Based on this review, the large majority 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 F and are designated by the appropriate qualifiers in the data tables.

A variety of field-based quality assurance test samples were also collected and analyzed during this project. These tests, which included the analysis of field blanks and blind duplicates, are described in detail in Appendix F and were used by the author to further qualify the laboratory data.

With the exception of DOC, field blank 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 field blanks exhibited a significant positive bias for DOC (Table F-1; the source of the contamination was not isolated during the study). Consequently, the DOC analytical results for all piezometers were either: 1) rejected if the sample result was below the respective filter blank value, or 2) qualified as potentially biased high if the sample result was above the respective blank value. DOC values from January and April are considered the most reliable values for the study period. DOC concentrations reported for this study are best used in discerning relative concentration differences between stations.

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 F. 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 (2002), with the exception of OP. The mean %RSD for all duplicate pairs for OP was 11.7%, a value greater than the target %RSD (10%) identified in the project plan. While the data values are still considered acceptable for the purposes of the study, all reported OP concentrations have been qualified as estimates by the author.

The precision for the DOC duplicate pairs was typically below the established target, with the exception of a very high error reported for one of the October sampling round pairs (>81%). As a result of this error, and the high level of DOC blank interference during the same round, all DOC data for October were rejected by the author.

The quality assurance testing and analyses implemented for this study collectively indicate that, with the exceptions noted, the overall data quality was excellent and met the data quality objectives established in the project plan.

Results

Hydraulic Gradient

Table G-1 in Appendix G presents the relative head measurements and vertical gradient estimates for each of the piezometers, by sampling round. Figure 12 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 vertical gradient was observed on only one occasion during the study period, at station LWGW-13 in August.

A progressive increase in the magnitude of the upward gradient was observed between October and April for many of the piezometers measured, and the highest gradients measured were typically recorded during the April round. A notable exception to this pattern occurred at station LWGW-14 (near the mouth of Anderson Creek), where the gradient greatly increased during the January measurement event (Figure 12). Gradient magnitudes measured during the August round were consistently among the lowest of the study period.

Groundwater Quality Results

A statistical summary of the groundwater water quality results for the study is presented in Table 2. Complete laboratory data results are presented in Table H-1 of Appendix H.

Table 2. Summary Statistics for Groundwater Water Quality Results

Constituent	Number of Samples	Minimum	Mean[1]	Median[1]	Maximum
pH	51	5.8	7.0	6.9	8.4
Specific conductance, field (µS/cm)	52	48	305	275	926
Dissolved oxygen, field (mg/L)[2]	31	<0.05	1.0	0.9	2.6
MBAS, field (mg/L)	28	No Detections above 0.025 mg/L			
TDS (mg/L)[3]	51	38	194	164	531
Chloride, dissolved (mg/L)	42	1.8	9.8	3.1	61.2
Ammonia-N, dissolved (mg/L)	50	<0.01	0.45	0.21	1.78
Nitrate-N, dissolved (mg/L)	49	<0.01	<0.01	<0.01	0.04
OP, dissolved (mg/L)[4]	51	0.01	0.09	0.06	0.35
TDP (mg/L)	50	0.02	0.32	0.14	2.18
DOC (mg/L)	Data Qualified				
Boron, dissolved (µg/L)	50	<50	71	60	270
Iron, dissolved (mg/L)	50	<0.05	8.7	1.9	38
Manganese, dissolved (mg/L)	50	0.018	0.358	0.160	1.63

[1] Non-detects assumed to be ½ the detection limit for calculation purposes.
[2] Only DO measurements collected using field photometer included in table.
[3] Includes values estimated by correlation to field specific conductance (SC).
[4] Mean duplicate %RSD precision is above target goal; concentrations considered estimates by author.

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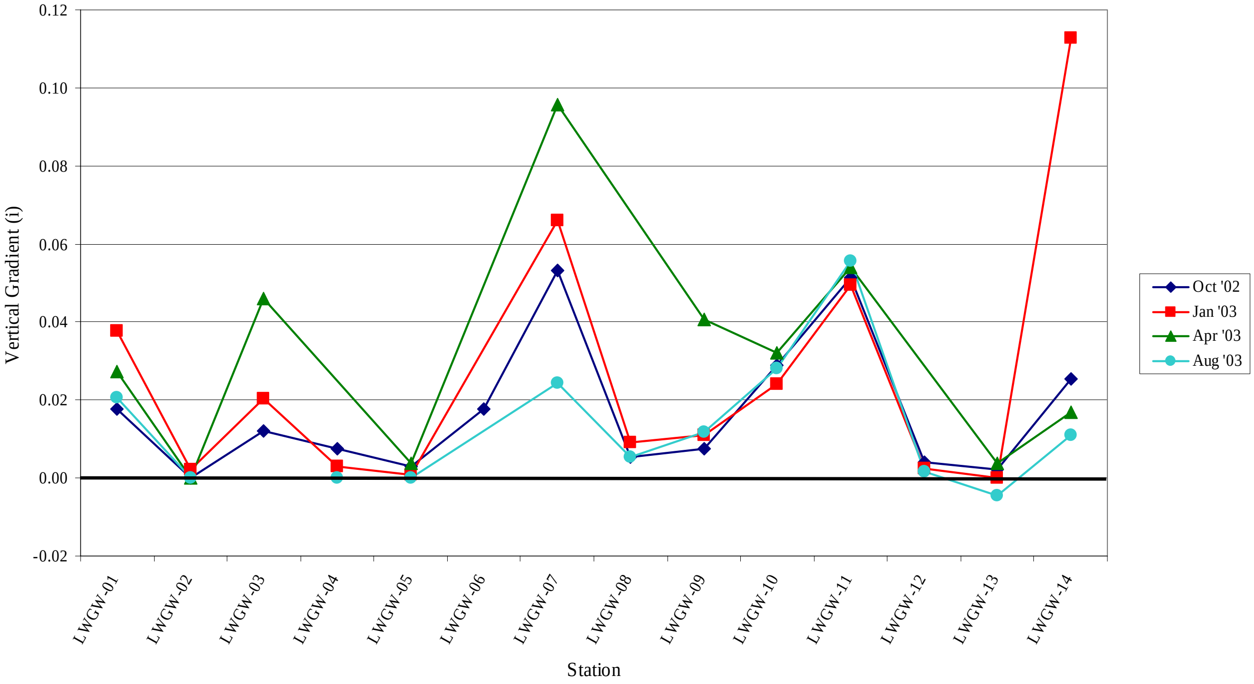

Figure 12
Estimated Vertical Hydraulic
Gradient at Study Piezometers

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Due to the difficulty of obtaining adequate sample volume from a number of the piezometers, TDS (and chloride) analytical results were limited for some of the stations during the study period. Estimated concentrations for TDS are presented in Table H-1 for most of these stations on the basis of a strong correlation between field-measured specific conductance (SC) and TDS concentration ($n = 38$; $TDS=0.5426(SC)+31.829$; $r^2 = 0.97$).

Colorimetric field analysis for methylene blue active substances (MBAS), an indicator of detergent residue, was conducted for all stations during the October sampling round, and again during the August round (Table E-1). No MBAS were detected with this technique at any of the piezometer stations.

Figures 13 through 18 map the geographic distribution of the average concentrations for TDS, chloride, nitrate-N, ammonia-N, OP, and TDP, respectively. Figures 19 and 20 present graphs of reported concentrations of DOC, boron, iron, and manganese during the January sampling round. The relative concentration distributions shown on Figures 19 and 20 are similar to those observed during other quarterly sampling rounds.

With the exception of station LWGW-09, TDS concentrations (Figure 13) generally decrease from the northern, more urbanized end of the lake to the southern, less developed end. The highest TDS concentrations were consistently observed in the piezometer adjacent to Bloedel-Donovan Park (LWGW-02; >500 mg/L); the lowest concentrations were consistently observed in the groundwater adjacent to the mouth of Anderson Creek (LWGW-14; <50 mg/L).

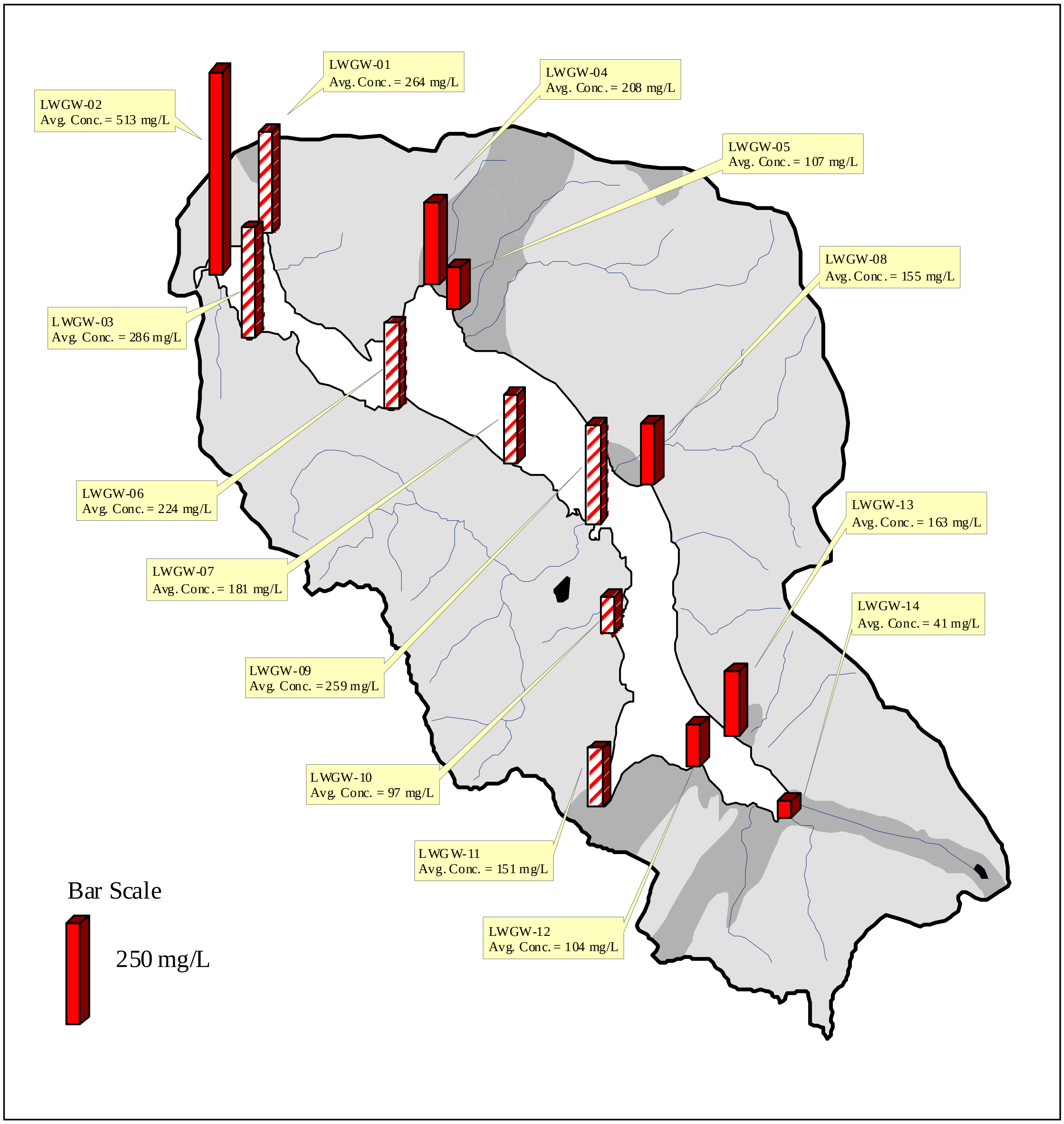
Chloride concentrations were routinely observed below 10 mg/L at all stations with the exception of piezometers LWGW-01 and LWGW-02 at the far northern end of the lake (Figure 14). These stations typically exhibited chloride conditions that were a factor of five times greater than the other locations tested. Concentrations for both TDS and chloride were normally consistent between sampling rounds.

Dissolved nitrogen in groundwater occurred predominantly as ammonia (Figures 15 and 16). The highest ammonia-N concentrations (>1 mg/L) were noted at stations LWGW-11 at the far southern end of South Bay, and station LWGW-02 adjacent to Bloedel-Donovan Park. Stations LWGW-03 and -05 also exhibited moderately elevated ammonia-N concentrations in comparison to other stations. Concentrations of ammonia-N were, for the most part, consistent between sampling rounds, although a small overall increase between October and August was observed at many of the stations. Nitrate-N was occasionally detected, but never at concentrations above 0.04 mg/L. Where detected, nitrate-N concentrations progressively increased from the October to April/May period, and then were consistently below the detection limit throughout the study area during the August sampling round.

Dissolved OP was detected above 0.050 mg/L in half of the piezometer stations tested (Figure 17). The highest OP concentrations were observed at stations LWGW-09 adjacent to the Sudden Valley golf course (~0.35 mg/L), and LWGW-12 in the South Bay area (~0.3 mg/L). A distinct increase in the OP concentration was noted at station LWGW-02 at Bloedel-Donovan Park between the January and May sampling events (January - 0.031 mg/L, May - 0.108 mg/L). Very high concentrations of TDP were consistently noted at the LWGW-09 station (>2 mg/L)

(Figure 18). Stations LWGW-02, -05, and -12 also exhibited moderately elevated concentrations of TDP in comparison to other stations. Orthophosphate-P typically comprised between 40-50% of the TDP concentration during the study.

Figure 19 illustrates DOC results for the January sampling round, the period indicated by field blank data to have the least amount of bias in the concentration results. Station LWGW-09 (Sudden Valley area) exhibited a significantly higher DOC concentration then all other stations (>20 mg/L); DOC values at stations LWGW-02 and LWGW-05 were also higher in comparison to other locations around the lake. The chart of dissolved boron concentrations for January indicates the highest values occurred at stations LWGW-13 (Blue Canyon area) and LWGW-01 (Silver Beach area). Boron was not detected at approximately half of the stations tested. For comparison, Figure 20 illustrates January results for dissolved iron and manganese. Stations LWGW-09 and -02 (Donovan Boat Launch) exhibited significantly higher dissolved metals than the remainder of the study area, with maximum values of iron exceeding 35 mg/L.



Note: Hachured bars indicate one or more field SC values were used to estimate TDS average.

Figure 13
Average Groundwater
Total Dissolved Solids
Concentration

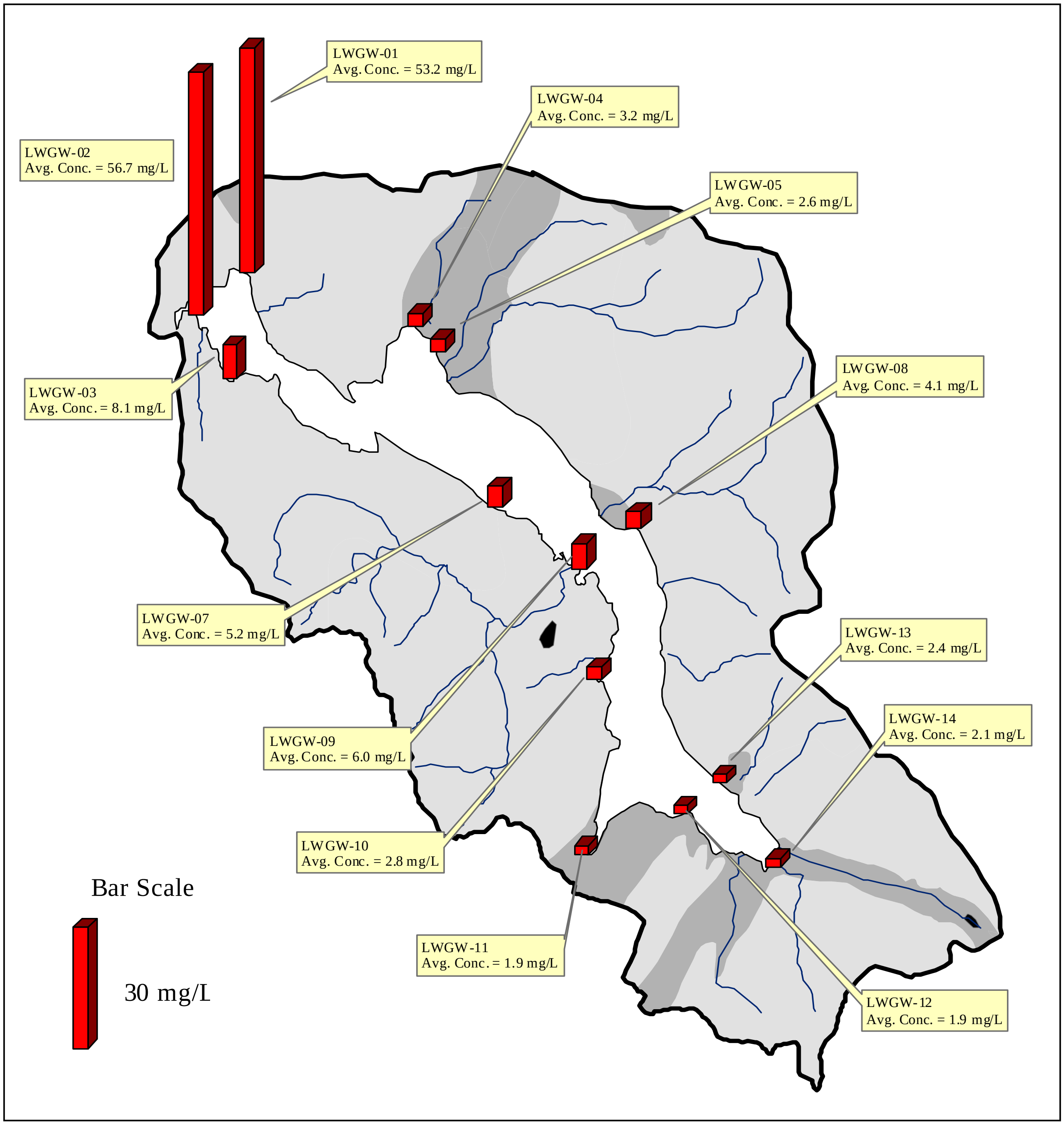


Figure 14
Average Groundwater
Chloride
Concentration
(Dissolved)

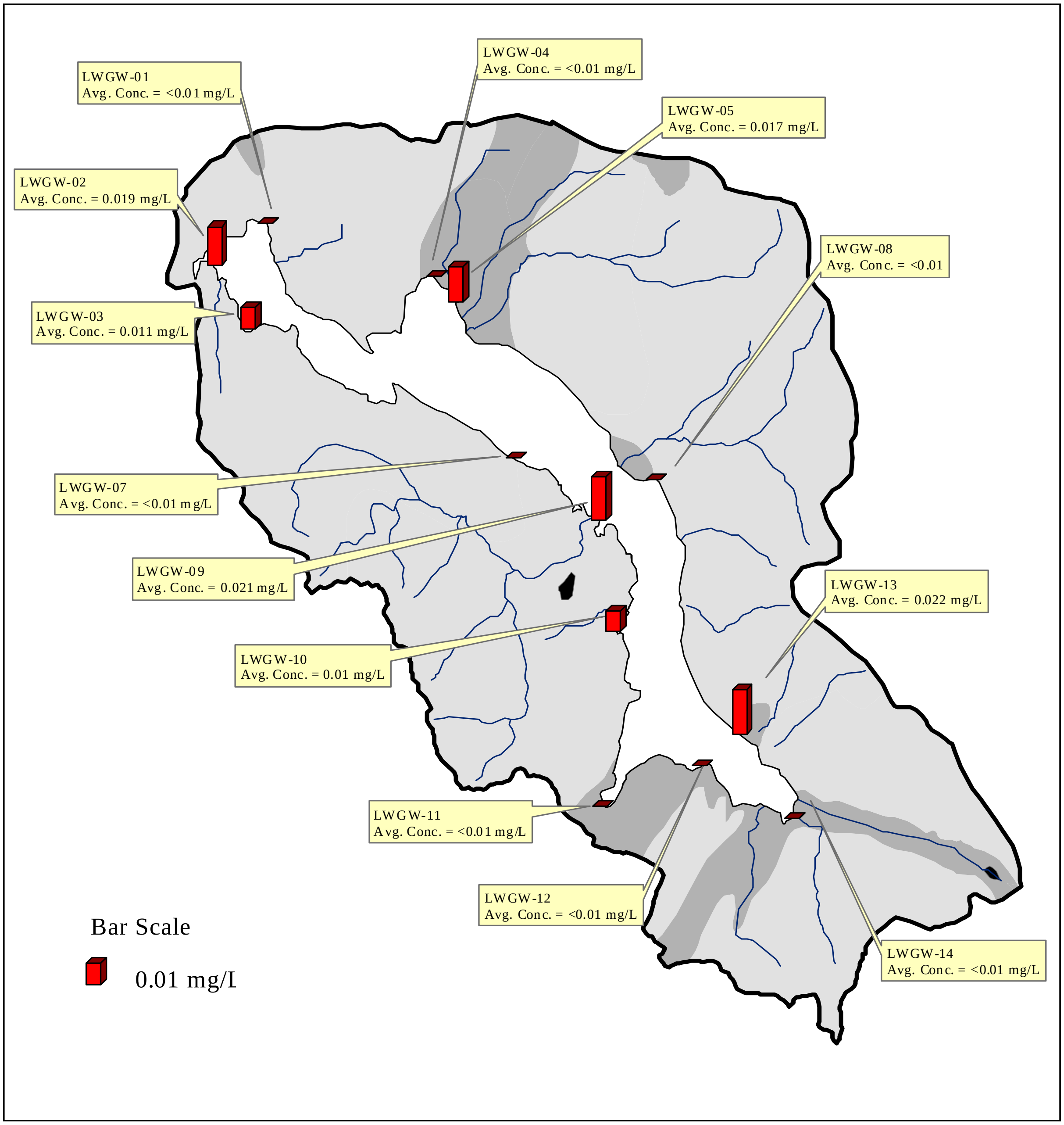


Figure 15
Average Groundwater
Nitrate-N
Concentration
(Dissolved)

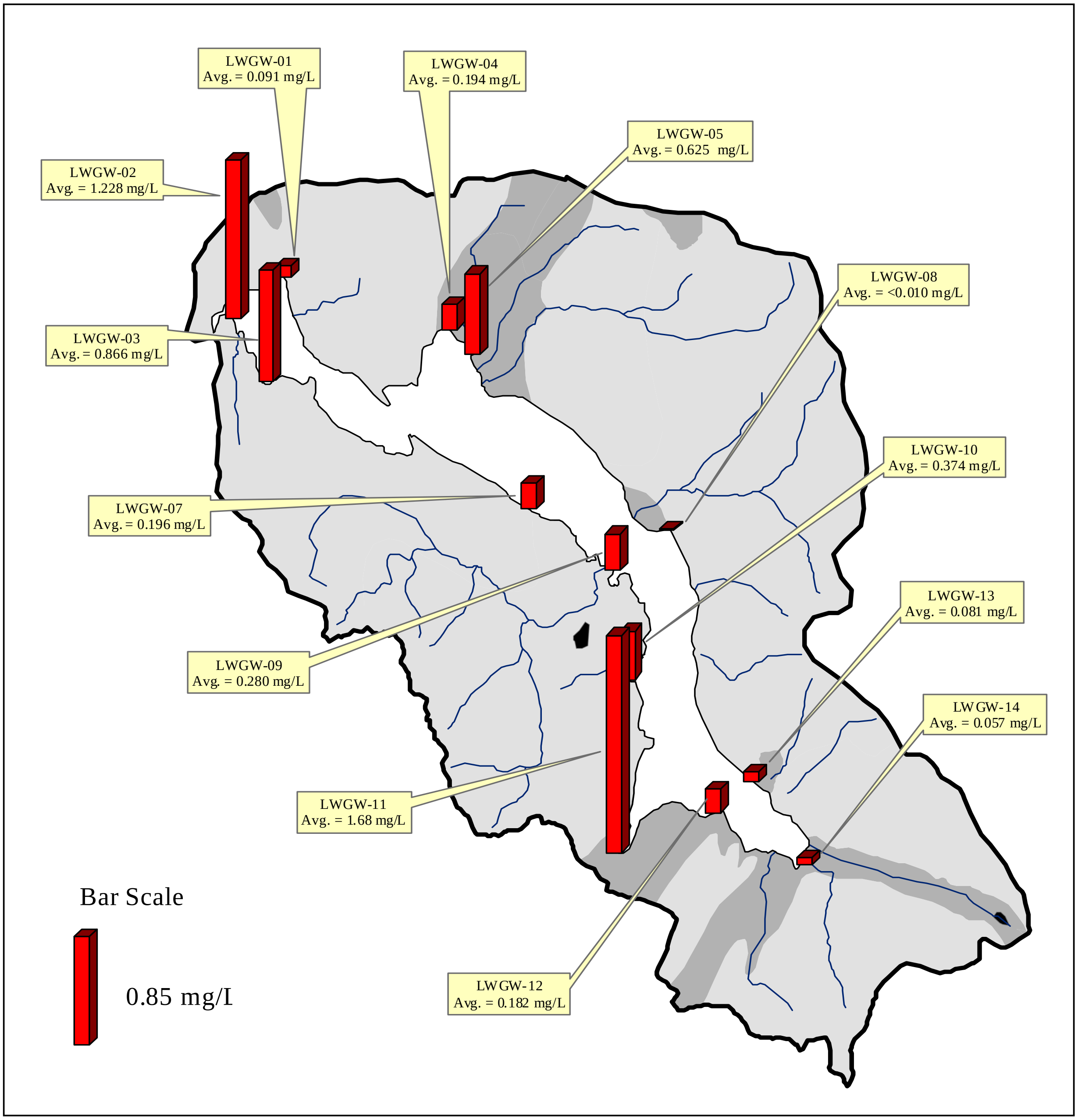


Figure 16
Average Groundwater
Ammonia-N
Concentration
(Dissolved)

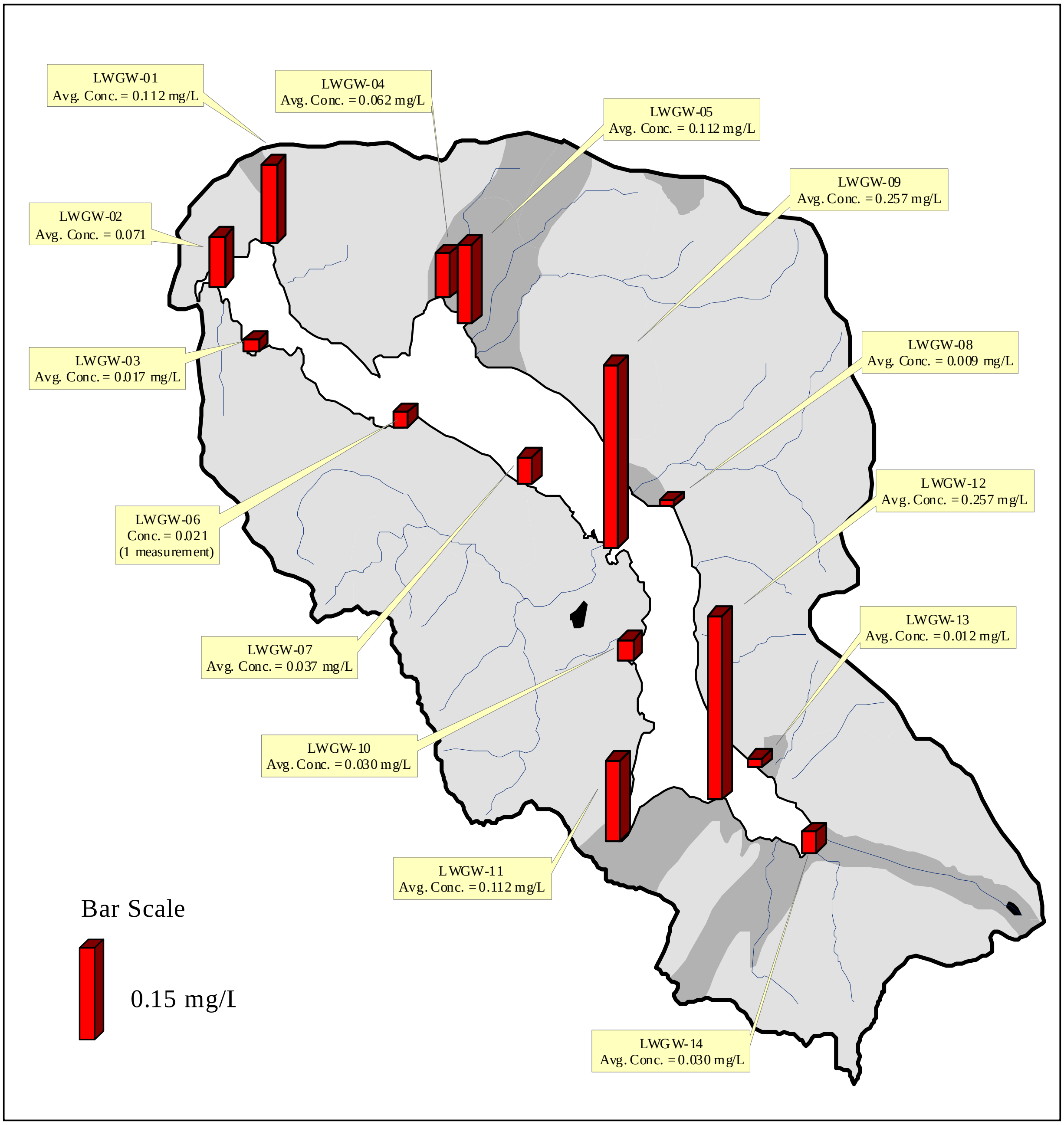


Figure 17
Average Groundwater
Orthophosphate-P
Concentration
(Dissolved)

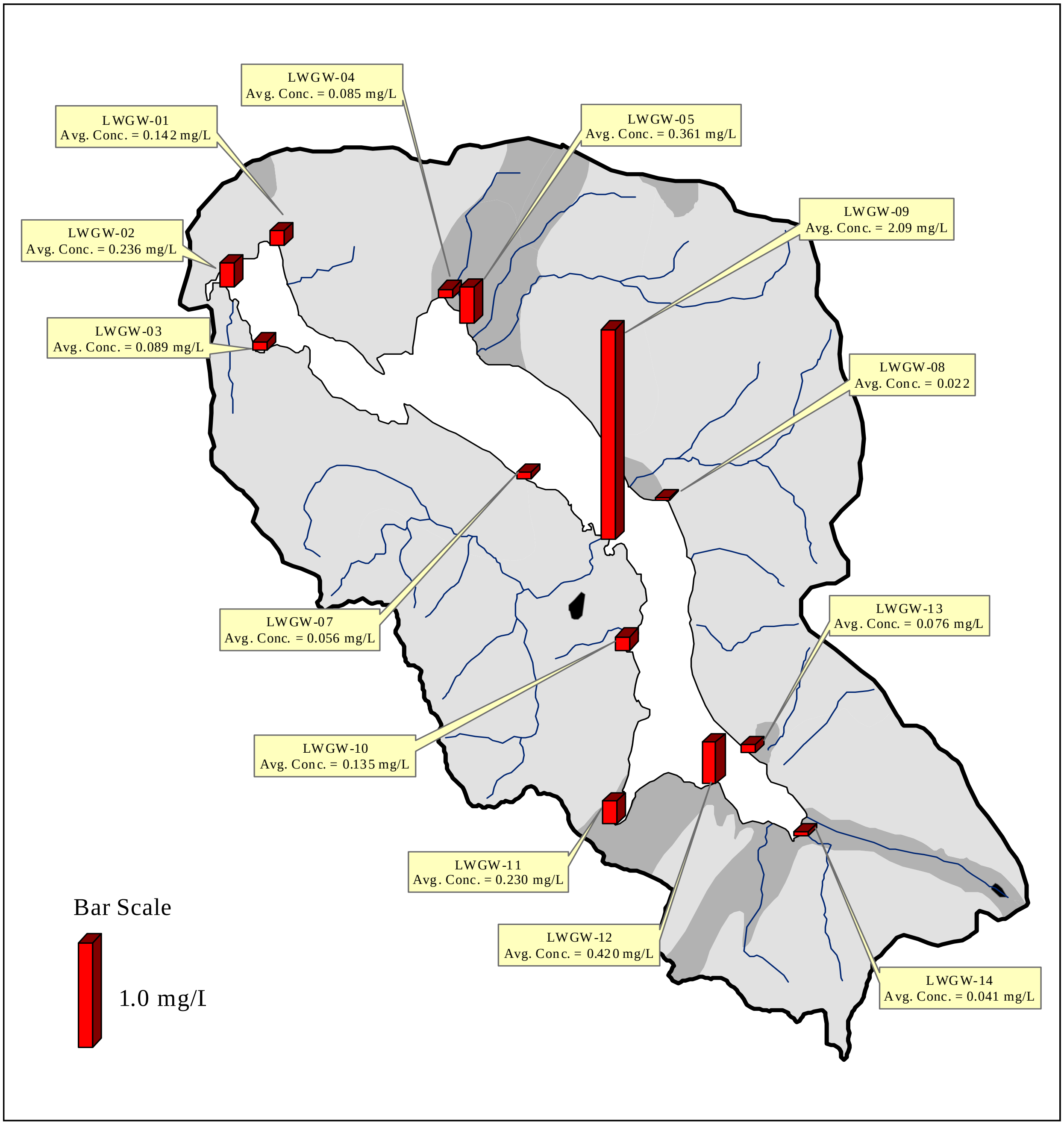
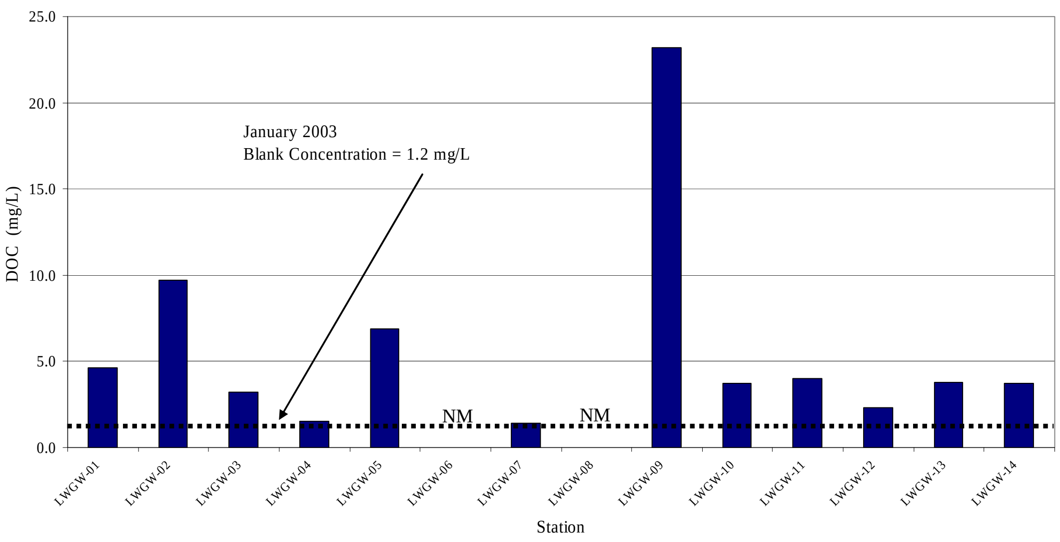
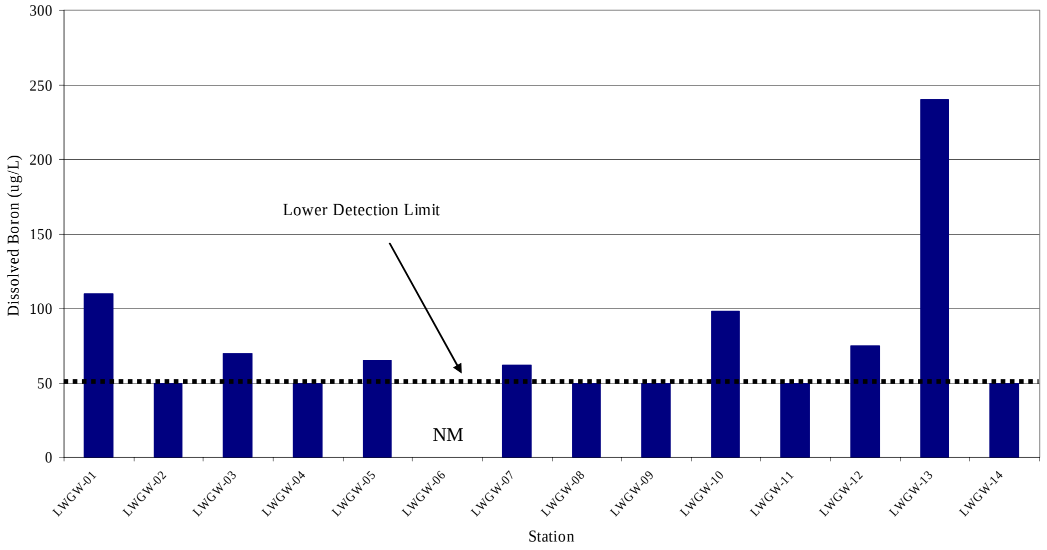


Figure 18
Average Groundwater
Total Dissolved Phosphorus
Concentration

Dissolved Organic Carbon
January 2003

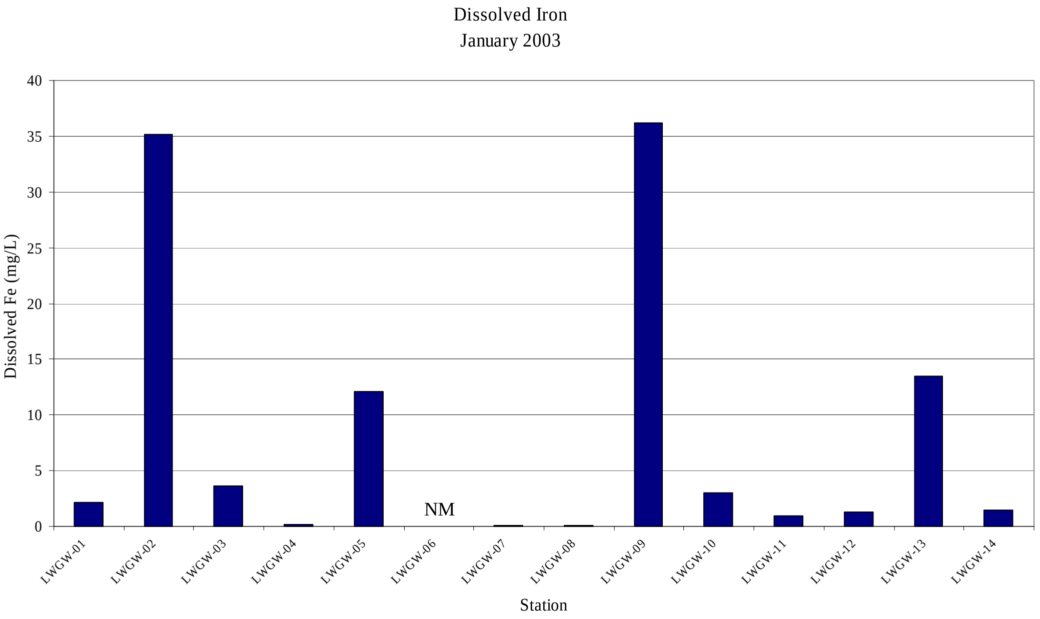


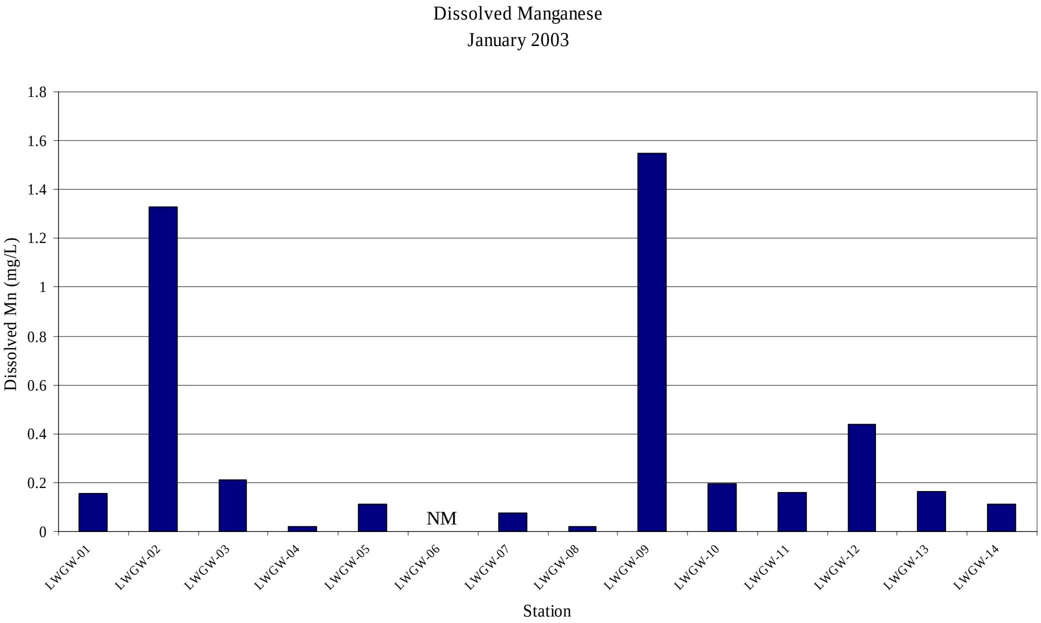
Dissolved Boron
January 2003



Note: Stations on the lower axis of the graphs are arranged in approximate geographic order from north to south (see Figure 8)
NM – Not measured

Figure 19
Dissolved Organic Carbon and Boron Concentrations In Groundwater – January 2003





Note: Stations on the lower axis of the graphs are arranged in approximate geographic order from north to south (see Figure 8)
NM = Not measured

Figure 20
Dissolved Iron and Manganese Concentrations In Groundwater – January 2003

Discussion

Hydraulic Gradient

The data presented in Table G-1 and Figure 12 indicate that the selection of the station locations for the study was successful in providing representative water quality data for inflowing groundwater. As seen on Figure 12, the majority of the hydraulic gradient measurements demonstrated an upward vertical gradient between the aquifer and the lake, indicating groundwater inflow is occurring to some degree around most of the lake shoreline.

Gradient variations between stations are interpreted to be a function of variations in the hydraulic conductivity of the lake bed sediments, and upgradient aquifer conditions. The increase in the magnitude of the vertical gradient between October and April, and the subsequent decrease in gradient between April and August, are interpreted to be an aquifer system response to the local seasonal precipitation and recharge pattern. Presumably regional groundwater levels were near their annual high point during April, and the gradients measured in the piezometers at this time reflect this regional condition. The specific cause for the strong increase in gradient at the LWGW-14 station during the January round is unclear, but may be related to the high streamflows reported by Matthews et al. (2004) in Anderson Creek during this portion of the year.

Groundwater Quality Results

The sampling methods used during this study were intended to characterize the nutrient content of groundwater in the final meters of the groundwater flow path through the aquifer system, just prior to its discharge to 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 constituent concentrations, while also improving the chances of accounting for near-shore nutrient sources influencing groundwater quality.

Interpretation of the potential origin of the nitrogen and phosphorus present in study area groundwater is complicated by the fact that these nutrients can have both natural and human sources. For example, the decomposition (and mineralization) of organic material deposited with the aquifer sediments can release nutrients to the dissolved phase. This process can result in nitrogen and/or phosphorus pore-water concentrations moderately elevated above levels normally observed in mineral sediments with a low organic content. It is best, therefore, to use a variety of factors, such as the overall water quality, presence or absence of constituents routinely associated with wastewater sources, and upgradient land-use patterns to help interpret the results.

At Lake Whatcom, the piezometers located in the most remote, undeveloped areas of the lake (presumably the closest representation of natural conditions) showed good water quality and relatively low nutrient concentrations. The nutrient concentrations that were observed at these stations are interpreted to be the result of natural processes. In contrast, the water quality profile and land-use development patterns upgradient of several of the other piezometers tested suggest that anthropogenic sources account for a significant percentage of the nutrient concentrations reported. In yet other cases, the origin of the nutrients is unclear, possibly the result of a combination of both natural and anthropogenic influences.

Figure 21 presents a series of radar charts illustrating the water quality profile of several stations located downgradient of comparatively undeveloped areas (LWGW-08, -10, -13, and -14; the *maximum* value reported during the 2002-2003 study period for each constituent was used to develop the charts). These background stations typically show suboxic to anoxic redox conditions, with relatively low dissolved iron and manganese concentrations. Concentrations of the parameters used as indicators of anthropogenic influence on water quality (TDS, chloride, and DOC) are typically also very low at these sites. Dissolved ammonia concentrations at these stations are typically low to moderate (< 0.4 mg/L), OP concentrations are normally well below 0.05 mg/L, and TDP concentrations do not exceed 0.15 mg/L.[4]

By contrast, the remaining stations shown on Figure 22 (LWGW-01, -02, -03, -04, -05, -07, -09, -11, and -12) are located downgradient of varying densities and types of land-use development. In comparison to the background stations, the water quality profiles presented on Figure 22 show variably elevated concentrations of TDS, chloride, DOC, ammonia-N, iron, manganese, OP, and TDP (manifested as larger and more spiked radar polygons than those shown on Figure 21).

In the case of station LWGW-02, the elevated concentrations of all of these constituents over background collectively suggest an anthropogenic influence on the quality of the local groundwater (Figure 22). At this station, the maximum reported OP concentration (0.116 mg/L) is more than five times higher than the average OP measured from the background stations (0.021 mg/L), and chloride is over 22 times higher than the background average (61.2 vs. 2.74 mg/L). The TDS concentration exceeded the state groundwater water quality standard of 500 mg/L during three of the four measurement rounds at the station.

The data support the interpretation that at least a portion of the nutrients observed at the LWGW-02 station are related to an upgradient wastewater source (this is similarly suggested for the elevated OP concentration measured at station LWGW-01, which additionally shows an elevated boron content, a possible indicator in this setting of detergent residue). Piezometer LWGW-02 is just downgradient of a public restroom facility located in Bloedel-Donovan Park, representing one possible source for the elevated nutrients. The significant increase noted at the LWGW-02 station for OP concentration between the January and May sampling periods (0.031 vs. 0.108 mg/L) could potentially be related to use patterns at the facility.

Station LWGW-09 also shows concentrations of TDS, dissolved metals, DOC, and nutrients elevated relative to background conditions, but the station lacks the high concentrations of chloride observed at the northern end of the lake (Figure 22). In this case, the maximum measured OP concentration (0.349 mg/L) is over 16 times higher than the background station average, and TDP is over 30 times higher than the background average (2.18 vs. 0.071 mg/L).

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[4] The upgradient land use and geologic setting at LWGW-13 indicate that the elevated boron concentration observed at the station is likely a natural condition, possibly related to the influence of connate water or mineral dissolution during weathering of bedrock (see Cox and Kahle, 1999).

The data suggest that the nutrients observed at this station may be the result of an anthropogenic influence unrelated to a septage source. The LWGW-09 piezometer is located immediately downgradient of a golf course operated by the Sudden Valley Community Association. Turf management practices employed at the golf course over the last 30 years have included routine surface application of phosphorus- and nitrogen-bearing fertilizer to the fairways and greens of the course, as well as regular irrigation during the dry season (SVCA, 2004).

While it is possible that the dissolved phosphorus measured at this site is related, in part, to the decomposition of natural organic material adjacent to the piezometer intake, the concentrations are well above those normally reported in organic-rich natural settings (e.g., in comparison to LWGW-11).

The chronic application of fertilizer to the golf course grounds adjacent to the lake could result in the downward migration of dissolved nutrients through the soil column to the water table, with subsequent lateral transport and submarine discharge via groundwater flow. Nutrient transport, particularly for phosphorus, would be enhanced by the regular infiltration of irrigation water, by the short distance to the water table (<10 feet), and by the anoxic conditions of the local groundwater system (itself potentially accentuated by nutrient-rich leachate consuming a portion of the available oxygen). Alternatively, the elevated nutrients observed at the LWGW-09 station could be related to land-use practices in place prior to the establishment of the golf course facility (e.g., livestock farming).

The origin of the elevated nutrient concentrations observed at some of the other piezometers sampled during the study (e.g., LWGW-03, -05, -11 and -12) is more difficult to interpret (Figure 22). While each of these stations is located downgradient of developed areas[5], the absence of elevated indicator parameters such as TDS, chloride, or DOC could suggest a nutrient source unassociated with septic leachate (e.g., fertilizer), or alternatively, that the ammonia and phosphorus are associated with the decomposition of naturally occurring organic matter present in the lake bed sediments adjacent to the piezometer intake.

The sediments at station LWGW-11 are comprised of organic-rich muck, suggesting the high ammonia-N concentrations at this station are probably in large part a natural condition. High organic content sediments were not observed at LWGW-03, -05 or -12. Stations LWGW-04 and 07 both exhibit profiles similar to the background stations, suggesting little anthropogenic influence on local groundwater quality.

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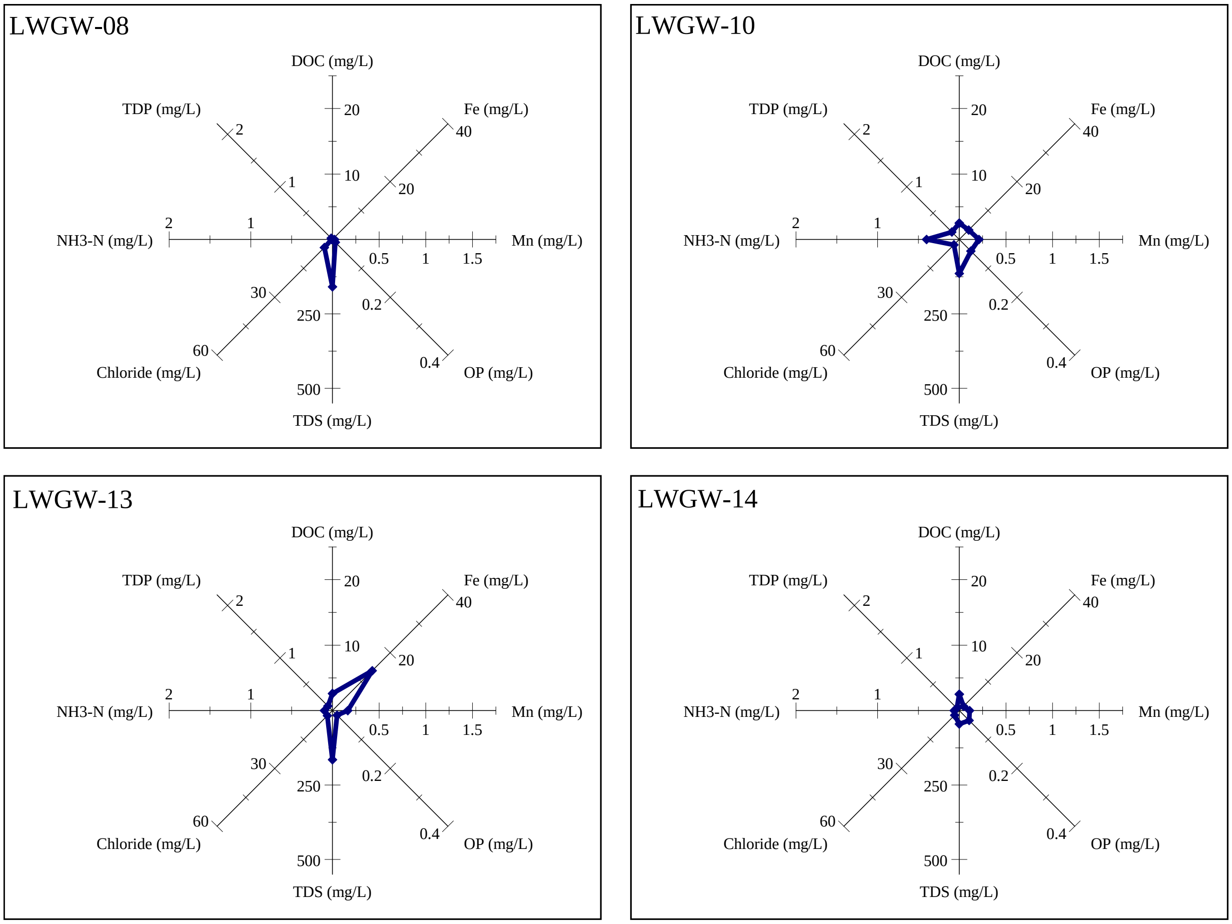
[5] LWGW-03 and -05 are downgradient of higher density residential development served by a mix of city sewer services and single household septic systems; LWGW-11 and -12 are downgradient of moderate density residential development served by single household septic systems; Figure 4.



Note: Charts show maximum concentrations reported for study period for each constituent. DOC concentrations have blank interference removed.

Figure 21
Water Quality Profile
Radar Charts -
“Background” Piezometers

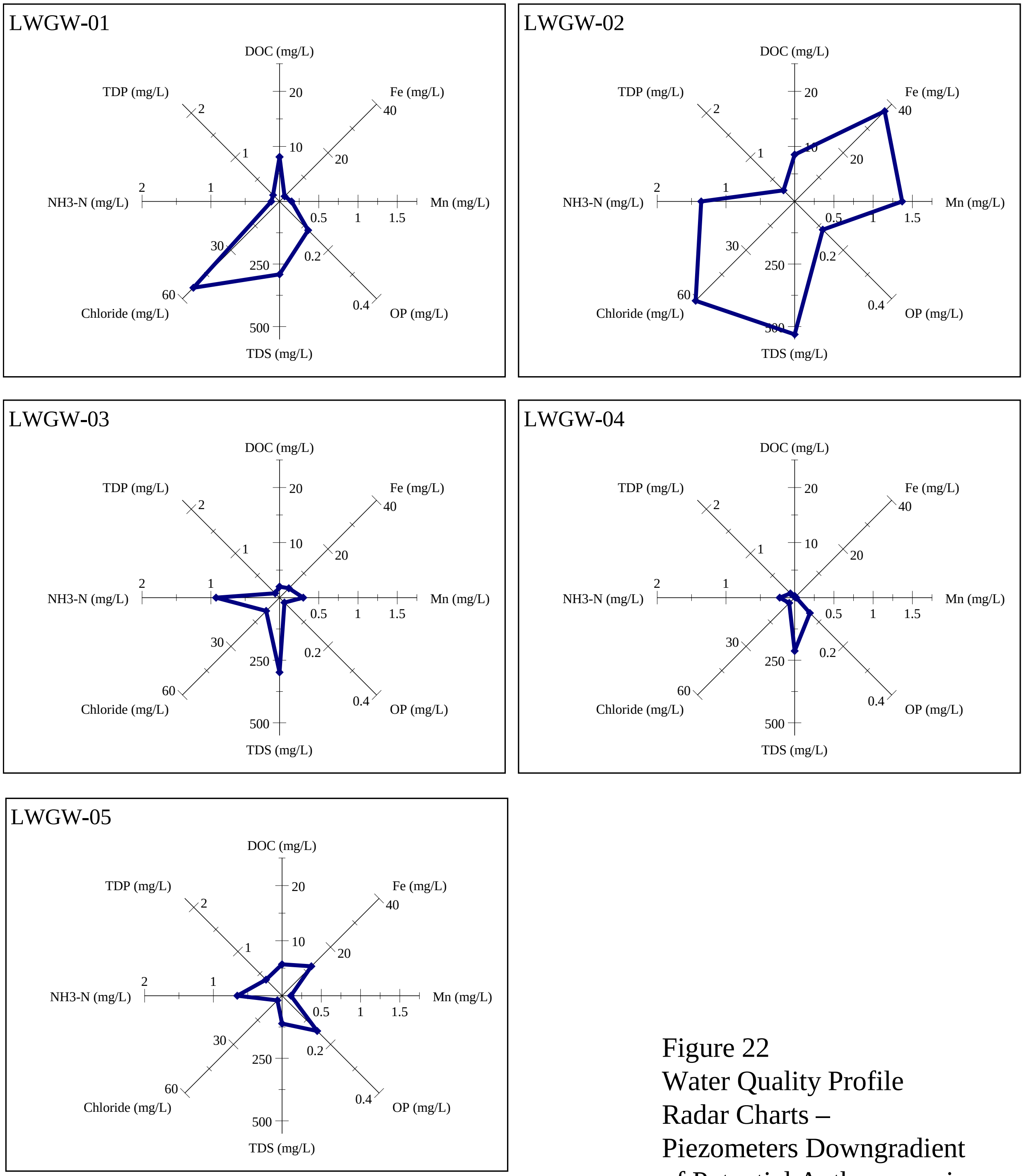


Figure 22
Water Quality Profile
Radar Charts –
Piezometers Downgradient
of Potential Anthropogenic
Nutrient Sources

Note: Charts show maximum concentrations reported for study period for each constituent. DOC concentrations have blank interference removed.

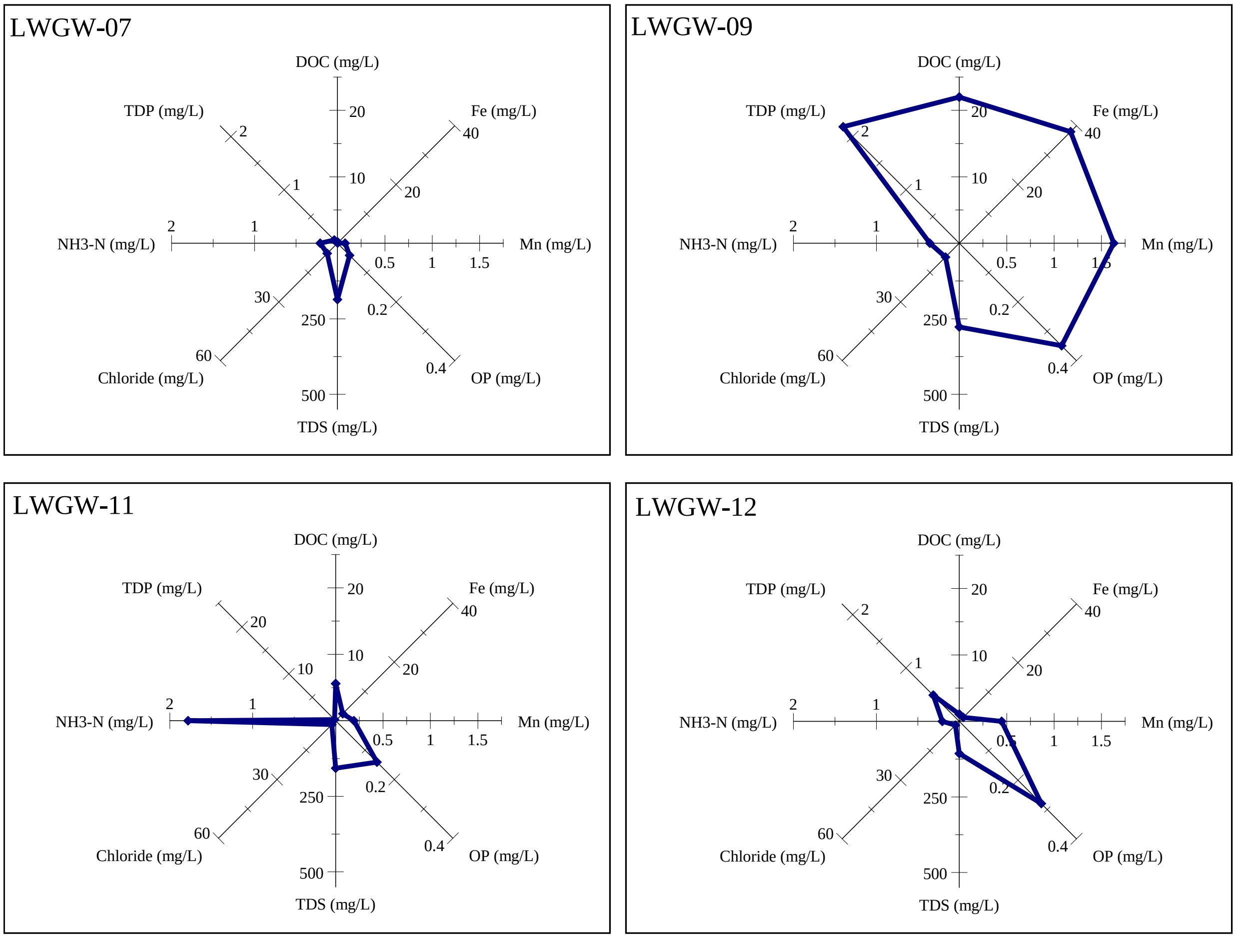


Figure 22 (cont.)
Water Quality Profile
Radar Charts –
Piezometers Downgradient
of Potential Anthropogenic
Nutrient Sources

Note: Charts show maximum concentrations reported for study period for each constituent. DOC concentrations have blank interference removed.

Loading Analysis

The water quality results presented above were integrated with the inflow volume estimates summarized in Table 1 to develop estimates of the nutrient loading to the lake via direct groundwater seepage during the 2003 water year. For the loading analysis, orthophosphate-P (OP) was chosen as the key dissolved constituent of interest in groundwater inflow due to its role as a limiting nutrient in lake eutrophication processes.

To calculate an OP mass flux from the aquifer to the lake, a representative OP concentration of the inflowing groundwater was developed for the three primary shoreline sections by calculating the geometric mean of all of the OP data from piezometers located within that section. The resulting concentration was then integrated with the range of estimates of inflow volume. The results of this analysis are presented in Table 3.

Table 3. Lake Whatcom Water Year 2003 Groundwater Orthophosphate-P Flux Estimates

Shoreline Section	RASA-Based Groundwater Inflow Q_{RASA} (liters/yr)[a]	WY '03 Precipitation-Based Inflow Q_{PRECIP} (liters/yr)[a]	Geomean OP Conc. C_{op} (mg/L)	RASA-Based Estimated OP Mass Flux $F_{op}=Q_{RASA}*C_{op}$ (kg_{op}/yr)	WY '03 Precipitation-Based OP Mass Flux $F_{op}=Q_{PRECIP}*C_{op}$ (kg_{op}/yr)
A - Agate Bay Shoreline	3.80E+09	2.30E+09	0.082[b]	310	190
B - South Bay Shoreline	8.42E+09	5.32E+09	0.095[c]	800	510
C - Bedrock Terrane (remaining shoreline)	4.83E+09	4.63E+09	0.038[d]	180	180
Total (A+B+C)	1.71E+10	1.23E+10		1290	880

[a] Groundwater inflow volumes derived from Table 1.
[b] Orthophosphate-P concentration of inflowing groundwater derived from field results for stations LWGW-04 and LWGW-05.
[c] Orthophosphate-P concentration of inflowing groundwater derived from field results for stations LWGW-11, -12, and -14.
[d] Orthophosphate-P concentration of inflowing groundwater derived from field results for stations LWGW-01, -02, -03, -06, -07, -08, -09, -10, and -13.
[e] Orthophosphate-P concentration of inflowing groundwater derived from field results for stations LWGW-01, -02, and -03.

The loading analysis suggests an upper-bound total of approximately 900 to 1300 kg_{op} entered the lake via direct groundwater inflow during the 2003 water year. The majority of the loading is predicted to occur adjacent to the shoreline of the South Bay area (Figure 7, Area B). Despite the limited lateral extent of the seepage faces of the South Bay and Agate Bay areas, the estimates indicate the combined OP load from these areas is significantly greater than the entire remainder of the lake. This reflects both higher overall OP concentration, and high annual groundwater inflow volumes from these portions of the groundwater basin, which in turn likely reflect differences in upgradient land use, geologic setting, infiltration capacity, and vulnerability to surface loading of nutrients.

The hydraulic gradient data presented in Figure 12 suggest that the highest rate of inflow (and therefore solute loading) occurs near the end of the wet season during the spring, while the lowest rate occurs at the end of the dry season during the late summer. This temporal pattern is consistent with previously predicted annual patterns of inflow to the lake (Matthews et al., 2003).

To judge the sensitivity of the load prediction to overestimates in the volume of groundwater discharging directly to the lake, versus discharge as baseflow to bedrock terrane streams, the Area C bedrock terrane inflow volume was reduced by 50%, and the OP mass load was recalculated. The annual OP mass flux from the bedrock terrane under this assumption decreased to approximately 90 kg_{op}/yr, reducing the total groundwater annual mass flux to a range between approximately 800 to 1200 kg_{op}/yr. This indicates that the mass estimates are not highly sensitive to over-predictions in the inflow rate from the bedrock.

It is important to note, however, that orthophosphate mobility in water is significantly influenced by prevailing redox conditions. For example, under reducing (anoxic) conditions, ferric oxyhydroxide precipitate solids that serve as sorption sites for phosphorus are dissolved. If there is a change from reducing to oxidizing conditions (e.g., the discharge of anoxic groundwater into an oxic surface sediment or lake environment), the dissolved iron can rapidly precipitate out of solution, providing additional sites for the sorption and immobilization of the phosphorus (Robertson et al., 1998; Walter et al., 1995; Carlyle and Hill, 2001; Hendricks and White, 2000; DiToro, 2001). Since the majority of the groundwater inflow to Lake Whatcom is likely to occur in the shallower, near-shore portions of the lake where the water is consistently well oxidized, much of the phosphorus delivered from the aquifer system may not be readily available in the water column, but instead bound up in the final centimeters of the flow path sediments.

The potential for phosphorus to be sequestered in the lake bed sediments mantling the seepage face, and the use of conservative values for inflow volume to the lake, both suggest that the annual mass flux numbers presented in Table 3 are best considered upper-bound estimates for future water quality modeling purposes. The procedures used for the development of the loading estimates also do not account for year-to-year variability or long-term time trends in water quality or groundwater inflow volume and are therefore best considered unique to the study period.

Conclusions

The key conclusions from this study include:

- The geologic setting strongly dictates the distribution of groundwater inflow to Lake Whatcom.
- A significant portion of the groundwater inflow to the lake is focused adjacent to the Agate Bay and South Bay/Anderson Creek Valley shoreline areas.
- Measurements of groundwater inflow to the lake downgradient of relatively undeveloped, area-background locations indicate suboxic to anoxic redox conditions, with low to moderate nutrient concentrations. Little anthropogenic influence on water quality was observed in these areas.
- Nutrient concentrations in groundwater inflow to the lake downgradient of developed areas are frequently elevated above background conditions. In several cases, the evidence suggests that the origin of the elevated nutrients is related to an anthropogenic influence on the groundwater quality. In other cases, the elevated nutrient condition is interpreted as a natural condition resulting from the decomposition of organic matter in the aquifer sediments, or from a combination of natural and anthropogenic origins.
- The reducing condition that prevails in area groundwater causes nitrogen, when present, to occur predominantly as ammonia. Reducing conditions also allow phosphorus inputs to the aquifer system to remain in the dissolved phase. This means that dissolved concentrations and travel distances away from phosphorus source areas may be greater than normally expected.
- Estimates of groundwater inflow volumes and measured water quality conditions were integrated to predict an annual phosphorus contribution to the lake from the adjacent aquifer system. The results of a loading analysis suggest an estimated range of approximately 900 to 1300 kg_{op} was input to the lake during the 2003 water year via direct groundwater inflow. Changes in the redox condition occurring in the final centimeters of the flow path may attenuate a significant portion of the phosphorus load. Due to the use of conservative assumptions for groundwater inflow, and the potential for phosphate attenuation near the sediment/lake interface, the mass flux predictions presented here should be considered upper-bound estimates for future water quality modeling purposes.

Recommendations

In light of the data collected during this study, the following recommendations are made:

- The prevalence of geochemical conditions in the local aquifer system that facilitate phosphorus transport by groundwater flow suggests that land use development that could result in high levels of phosphorus loading to the subsurface should be minimized, particularly in near-shore or high-inflow areas.

- Additional focused field investigations should be considered at the golf course located adjacent to piezometer LWGW-09 to assess the effectiveness of current turf management practices in limiting the downward loss of phosphorus to the shallow groundwater system.

- Additional focused field investigations should be considered to assess the impacts of the restroom facility or other activities at Bloedel-Donovan Park on the local groundwater quality.

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Appendices

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Appendix A. Recharge Estimate Data

Table A-1. Lake Whatcom Water Year 2003 Groundwater Recharge Estimates

				Scenario 1		Scenario 2	
Location	Recharge Polygon Area (ft^2)	Recharge Polygon Area (m^2)	RASA Recharge Group(1)	Mean Annual RASA Recharge Rate Minimum (in/yr)(1)(2)	RASA Annual Recharge Volume Minimum Q_{RASA} (liters/yr)	WY '03 Precipitation-Based Recharge Estimate (in/yr)(2)(3)(4)	WY '03 Precipitation-Based Recharge Estimate Q_{PRECIP} (liters/yr)
Agate Bay Recharge Area (A)	463512	43062	3	7	7.66E+06	9.3	1.02E+07
	606684	56363	3	7	1.00E+07	9.3	1.33E+07
	3050341	283386	4	12	8.64E+07	9.3	6.69E+07
	928187	86231	4	12	2.63E+07	9.3	2.04E+07
	166885	15504	4	12	4.73E+06	9.3	3.66E+06
	24376873	2264685	5	17	9.78E+08	9.3	5.35E+08
	38328174	3560802	7	27	2.44E+09	15.5	1.40E+09
	29811503	2769578	Bedrock	2	1.41E+08	2	1.41E+08
	23007865	2137500	Bedrock	2	1.09E+08	2	1.09E+08
Subtotals	120740024	11217110			3.80E+09		2.30E+09
South Bay Recharge Area (B)	85733	7965	5	17	3.44E+06	12.9	2.61E+06
	555204	51580	5	17	2.23E+07	12.9	1.69E+07
	796941	74038	5	17	3.20E+07	12.9	2.43E+07
	1928729	179185	5	17	7.74E+07	12.9	5.87E+07
	316833	29435	8	37	2.77E+07	21.5	1.61E+07
	27633952	2567277	8	37	2.41E+09	21.5	1.40E+09
	55887052	5192075	8	37	4.88E+09	21.5	2.84E+09
	100364305	9324145	Bedrock	2	4.74E+08	2	4.74E+08
	25159407	2337384	Bedrock	2	1.19E+08	2	1.19E+08
	60907591	5658498	Bedrock	2	2.87E+08	2	2.87E+08
	692037	64292	Bedrock	2	3.27E+06	2	3.27E+06
	17253340	1602887	Bedrock	2	8.14E+07	2	8.14E+07
Subtotals	291581124	27088761			8.42E+09		5.32E+09
Bedrock Terrane Recharge Area (C)	700133	65044	3	7	1.16E+07	9.3	1.54E+07
	697948	64841	3	7	1.15E+07	9.3	1.53E+07
	1994265	185273	4	12	5.65E+07	9.3	4.38E+07
	122282	11360	5	17	4.91E+06	9.3	2.68E+06
	1367375	127033	5	17	5.49E+07	9.3	3.00E+07
	319506	29683	5	17	1.28E+07	9.3	7.01E+06
	3150021	292646	8	37	2.75E+08	15.5	1.15E+08
	426350664	39609256	Bedrock	2	2.01E+09	2	2.01E+09
	507121604	47113118	Bedrock	2	2.39E+09	2	2.39E+09
Subtotals	8351530	775882			4.83E+09		4.63E+09

(1) Recharge rates for unconsolidated deposit polygons are assumed to be the minimum of mean annual range reported by Vaccaro et al., 1998.

(2) Recharge rate for bedrock polygons is an assumed value.

(3) Recharge assumed to be 30% of annual precipitation for predominantly fine-grained units, 50% of annual precipitation for predominantly coarser-grained units.

(4) WY '03 annual precipitation for Agate Bay Area and Bedrock Terrain assumed 31"; for South Bay area assumed 43" (R. Mitchell, personal comm., 2004).

Appendix B. Station Location Data

Table B-1. Piezometer Location and Construction Data

Station Name	Decimal Latitude[(a)]	Decimal Longitude[(a)]	Location Description	Dominant Upgradient Land Uses	Total Depth of Piezometer below Sediment Surface (ft)	Screen Length (ft)	Depth of Screen Mid-point below Sediment Surface (ft)	Appr. Distance from Shore (ft)
LWGW-01	48.767799	122.405276	Silver Beach area	suburban/ forest land	5.3	1	4.8	54
LWGW-02	48.760703	122.417733	Bloedel-Donovan boat ramp	city park/ suburban	5.1	1	4.6	34
LWGW-03	48.750265	122.408976	Geneva Bay	forest land/ suburban	4.1	1	3.6	62
LWGW-04	48.759772	122.362617	North Agate Bay	shoreline residential/ forest land/rural	4.9	1	4.4	28
LWGW-05	48.755633	122.356973	South Agate Bay	shoreline residential/ forest land/rural	4.8	1	4.3	18
LWGW-06	48.738766	122.372074	Strawberry Point Bay	shoreline residential/ forest land	5.0	1	4.5	100
LWGW-07	48.730049	122.341555	WWU Boathouse	mixed suburban and forest land	4.3	1	3.8	30
LWGW-08	48.727000	122.306570	Sunnyside	shoreline residential/ forest land	4.1	1	3.6	60
LWGW-09	48.720165	122.320126	Sudden Valley	golf course/ suburban	4.9	1	4.4	80
LWGW-10	48.701857	122.315733	West Shore	forest land	4.6	1	4.1	45
LWGW-11	48.672898	122.317953	South Bay	suburban/ forest land	4.2	1	3.7	50
LWGW-12	48.679916	122.293269	South Shore	suburban/ forest land	4.7	1	4.2	33
LWGW-13	48.685233	122.283643	Blue Canyon	forest land/ shoreline residential	4.8	1	4.3	30
LWGW-14	48.671566	122.269918	Anderson Creek	pasture/ forest land/rural	4.4	1	3.9	43

[(a)] NAD 27
WWU – Western Washington University

Appendix C. Grain-Size Data – Station Surface Sediments

Table C-1. Surface Sediment Grain Size Distribution Data

	Percent Finer Than Indicated Size, ASTM D422														ASTM D2488
Station	3"	2"	1.5"	1"	3/4"	1/2"	3/8"	#4	#10	#20	#40	#60	#100	#200	Soil Classification
LWGW-01	100.0	100.0	100.0	100.0	100.0	100.0	100.0	99.5	98.9	97.2	95.0	92.8	90.9	85.5	CL, Lean clay
LWGW-02	100.0	100.0	100.0	100.0	96.8	87.9	79.5	55.5	8.6	0.9	0.3	0.2	0.1	0.1	SP, Poorly graded sand w/ gravel
LWGW-03	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0	99.9	99.8	98.7	51.9	9.7	1.3	SP, Poorly graded sand
LWGW-04	100.0	100.0	100.0	100.0	96.2	84.7	71.8	43.7	28.1	24.8	16.6	3.1	1.0	0.5	GP, Poorly graded gravel w/ sand
LWGW-05	100.0	100.0	100.0	92.7	64.2	48.8	45.3	44.1	43.5	42.8	41.1	35.0	18.6	2.9	GP, Poorly graded gravel w/ sand
LWGW-06	Not sampled														-
LWGW-07	100.0	100.0	100.0	100.0	100.0	94.8	93.1	83.8	75.6	66.6	54.5	33.0	17.2	6.9	SW, Well graded sand w/ gravel
LWGW-08	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0	99.9	99.8	98.7	51.9	9.7	1.3	SP, Poorly graded sand
LWGW-09	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0	99.5	98.0	96.1	92.7	74.2	26.2	SP, Poorly graded sand w/ silt
LWGW-10	100.0	100.0	100.0	100.0	96.0	88.7	82.3	71.1	60.4	52.9	42.5	21.5	7.2	2.2	SW, Well graded sand w/ gravel
LWGW-11	100.0	100.0	100.0	100.0	100.0	100.0	100.0	99.9	98.9	94.8	87.2	71.5	51.5	21.9	SP, Poorly graded sand w/ silt, organic rich
LWGW-12	100.0	100.0	100.0	94.3	78.7	62.7	55.5	47.8	40.8	27.9	13.3	4.9	2.4	1.1	GW, Well graded gravel w/sand
LWGW-13	100.0	100.0	100.0	100.0	97.0	95.6	91.5	76.7	61.7	52.6	45.9	32.8	19.9	8.5	SW, Well graded sand w/ gravel
LWGW-14	100.0	100.0	100.0	100.0	100.0	100.0	100.0	99.9	99.6	99.3	97.4	83.8	48.5	9.9	SP, Poorly graded sand w/ silt

Notes:
Analyses performed on grab samples from top 1 foot of lake sediments.
Analyses performed by Rosa Environmental.

Appendix D. Analytical Procedures

Table D-1. Summary of Project Analysis Methods

Constituent	Analytical Method[A]	Reporting Limit
Field		
pH	Field meter	NA
Specific conductance	Field meter	NA
Temperature	Field meter	NA
Dissolved Oxygen	Chemetrics® Colorimetric Indigo Carmine	~0.2mg/L
"	Chemetrics® Colormetric RhodazineD™	~0.010 mg/L
MBAS	Chemetrics® colormetric	~0.025 mg/L
Laboratory[B]		
TDS	EPA 160.1	1 mg/L
Chloride	EPA 300.0	0.1 mg/L
Nitrate-N	SM 4500-NO3 I	0.010 mg/L
Ammonia-N	SM 4500-NH3 H	0.010 mg/L
OP	SM4500-P G	0.010 mg/L
TDP	SM 4500-P I	0.010 mg/L
DOC	EPA 415.1	1 mg/L
Boron	EPA 200.7 ICP	0.050 mg/L
Iron	EPA 200.7 ICP	0.020 mg/L
Manganese	EPA 200.7 ICP	0.010 mg/L
Grain Size	ASTM-D422	NA

(A) SM - Standard Methods for the Examination of Water and Wastewater, 20th Edition, American Public Health Assoc., Wash. DC.

(B) All lab analyses performed by Ecology's Manchester Environmental Lab with the exception of grain size, which was performed by Rosa Environmental.

MBAS - methylene blue active substances
TDS - total dissolved solids
OP – orthophosphate-P
TDP – total dissolved phosphorus
DOC - dissolved organic carbon

Appendix E. Field Water Quality Data

Table E-1. Field Water Quality Data

October 2002		Lake			Groundwater			
Station	Date	pH	SC (uS/cm)	DO (mg/L)[a]	pH	SC (uS/cm)	DO (mg/L)[a]	MBAS (mg/L)
LWGW-01	10/17/2002	6.3	61	7.98	8.2	391	4.13	<0.025
LWGW-02	10/16/2002	8.6	61	11.37	6.4	926	3.98	<0.025
LWGW-03	10/16/2002	7.8	61	12.40	6.8	433	4.39	<0.025
LWGW-04	10/17/2002	8.5	60	7.00	7.5	329	0.10	<0.025
LWGW-05	10/17/2002	8.3	60	8.30	6.3	136	0.18	<0.025
LWGW-06	10/16/2002	8.0	58	14.19	6.9	354	8.48	<0.025
LWGW-07	10/16/2002	6.6	59	12.57	8.3	282	3.79	<0.025
LWGW-08	10/15/2002	7.4	58	13.10	8.1	254	1.77	<0.025
LWGW-09	10/15/2002	7.6	59	11.20	6.0	350	2.54	<0.025
LWGW-10	10/15/2002	7.7	59	11.90	6.5	114	3.72	<0.025
LWGW-11	10/15/2002	7.0	59	11.70	6.9	197	5.30	<0.025
LWGW-12	10/14/2002	7.5	59	10.38	7.4	146	1.90	<0.025
LWGW-13	10/14/2002	7.0	59	10.34	6.7	274	1.35	<0.025
LWGW-14	10/14/2002	7.6	59	11.63	6.1	53	1.68	<0.025

January 2003		Lake			Groundwater			
Station	Date	pH	SC (uS/cm)	DO (mg/L)[a]	pH	SC (uS/cm)	DO (mg/L)[b]	MBAS (mg/L)
LWGW-01	1/30/2003	7.8	62	NM	7.8	415	**1.7**	NM
LWGW-02	1/29/2003	8.4	61	10.35	7.9	915	**0.03**	NM
LWGW-03	1/29/2003	7.8	57	11.10	7.7	429	**1.4**	NM
LWGW-04	1/29/2003	6.3	57	9.70	7.3	332	**0.13**	NM
LWGW-05	1/30/2003	7.6	57	8.85	5.8	119	**0.03**	NM
LWGW-06	NM	NM	NM	NM	NM	NM	NM	NM
LWGW-07	1/29/2003	7.5	56	9.38	7.5	253	**2.6**	NM
LWGW-08	1/28/2003	7.1	57	10.17	8.1	250	**0.13**	NM
LWGW-09	1/28/2003	8.3	58	9.55	6.0	355	**0.86**	NM
LWGW-10	1/28/2003	7.6	58	9.42	6.6	113	**2.38**	NM
LWGW-11	1/28/2003	6.5	61	9.24	6.9	208	**0.7**	NM
LWGW-12	1/27/2003	7.2	57	9.29	7.4	143	**0.12**	NM
LWGW-13	1/27/2003	6.8	61	9.27	6.7	299	**0.1**	NM
LWGW-14	1/27/2003	6.5	54	10.51	5.8	60	**0.2**	NM

SC - specific conductance; DO - dissolved oxygen; MBAS - methylene blue active substances

[a] Dissolved oxygen measurements collected using a membrane probe. Membrane probe DO concentrations <2 mg/L are not considered accurate.

[b] Dissolved oxygen measurements in **bold** font derived by colorimetric analysis with field photometer
NM - Not measured

Table E-1 (cont.). Field Water Quality Data

April 2003		Lake			Groundwater			
			SC	DO		SC	DO	MBAS
Station	Date	pH	(uS/cm)	(mg/L)	pH	(uS/cm)	(mg/L)[b]	(mg/L)
LWGW-01	5/1/2003	7.7	63	NM	8.1	429	**2.3**	NM
LWGW-02	5/1/2003	8.5	60	NM	6.5	879	**0.29**	NM
LWGW-03	5/1/2003	6.5	60	NM	6.8	415	**1.58**	NM
LWGW-04	4/30/2003	7.1	58	NM	7.5	334	**0.39**	NM
LWGW-05	4/30/2003	8.6	58	NM	6.4	131	**0.23**	NM
LWGW-06	NM	NM	NM	NM	NM	NM	NM	NM
LWGW-07	4/30/2003	7.8	58	NM	8.4	275	**1.88**	NM
LWGW-08	NM	NM	NM	NM	NM	NM	NM	NM
LWGW-09	NM	NM	NM	NM	6.1	359	**1.13**	NM
LWGW-10	NM	NM	NM	NM	6.6	116	**1.98**	NM
LWGW-11	4/29/2003	7.2	58	NM	6.9	213	**0.2**	NM
LWGW-12	4/28/2003	7.5	57	NM	7.6	143	**0.13**	NM
LWGW-13	4/28/2003	6.9	57	NM	6.7	295	**0.18**	NM
LWGW-14	4/28/2003	7.4	58	NM	6.1	54	**0.19**	NM

August 2003		Lake			Groundwater			
			SC	DO		SC	DO	MBAS
Station	Date	pH	(uS/cm)	(mg/L)[a]	pH	(uS/cm)	(mg/L)[a,b]	(mg/L)
LWGW-01	8/28/2003	8.2	61	7.42	8.0	475	4.00	<0.025
LWGW-02	8/28/2003	8.7	60	8.82	6.3	910	0.21	<0.025
LWGW-03	NM	NM	NM	NM	NM	NM	NM	<0.025
LWGW-04	8/27/2003	8.2	59	8.05	7.5	331	0.24	<0.025
LWGW-05	8/27/2003	8.3	58	8.38	6.3	118	0.2	<0.025
LWGW-06	NM	NM	NM	NM	NM	NM	NM	<0.025
LWGW-07	8/27/2003	8.1	58	8.34	8.3	279	**0.9**	<0.025
LWGW-08	8/25/2003	8.4	59	8.16	8.0	250	0.18	<0.025
LWGW-09	8/26/2003	9.5	59	8.58	6.1	346	1.44	<0.025
LWGW-10	8/26/2003	8.5	59	7.22	6.5	115	**1.3**	<0.025
LWGW-11	8/26/2003	7.9	59	7.14	6.9	203	**2.2**	<0.025
LWGW-12	8/25/2003	8.1	59	8.18	7.4	139	**1.92**	<0.025
LWGW-13	8/25/2003	8.3	59	8.02	6.6	275	**1.3**	<0.025
LWGW-14	8/25/2003	8.0	59	8.05	6.0	48	**1.17**	<0.025

SC - specific conductance; DO - dissolved oxygen; MBAS - methylene blue active substances

[a] Dissolved oxygen measurements collected using a membrane probe. Membrane probe DO concentrations <2 mg/L are not considered accurate.

[b] Dissolved oxygen measurements in **bold** font derived by colorimetric analysis with field photometer
NM - Not measured

Appendix F. Project Quality Assurance

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 method blanks, duplicate samples, spiked samples, and control standards. Manchester Laboratory's quality control procedures are discussed in detail in MEL (1988). Quality assurance reviews were completed for each round of sampling. The laboratory reviews revealed that the data were of generally excellent quality, meeting or exceeding the data quality objectives established in the project plan. The laboratory indicated that the data can be used without qualification with the following exceptions:

- The October sample for LWGW-02 for dissolved organic carbon (DOC) analysis was not field preserved. The sample was preserved upon arrival at the laboratory.
- The October samples for LWGW-01, -02, and -05 DOC analysis were analyzed past their holding time due to the need for reanalysis due to dilution requirements, and are qualified as estimates.
- The January samples for LWGW-01, -04, -05, -12, -13, -14, -05 duplicate, and -12 duplicate DOC analysis, were all analyzed past their proper holding time and are qualified as estimates.
- The April sample for LWGW-02 for DOC was not field preserved. The sample was preserved upon arrival at the laboratory.

Analytical Quality Assurance – Field

Filter Blanks

Clean, laboratory-supplied de-ionized (DI) water was pumped through the sample collection and filtering system once per sampling round to determine if any component of the sampling system was contributing a positive bias to the analytical results. All of the components used to collect these samples were new.

During one of the sampling rounds (April), an additional filter blank was collected to test the potential for cross contamination between stations. A new filter was used to collect the sample, but all other components of the collection system had been previously used at one of the study piezometer stations. For this sample, the collection system was purged with clean de-ionized water for approximately five minutes prior to attaching the filter, mimicking the normal sample collection procedure. This sample was primarily intended to determine if there was significant adsorption/desorption of target nutrients on the sample contact surfaces of the collection system between stations.

The results for the project filter blanks are presented in Table F-1. The results indicate that the sample collection and filtration system did not introduce significant bias or cross contamination into the study analytical results for most of the study parameters; no significant difference was observed between the filter blank results for new versus used sampling components. Significant positive bias was, however, observed in all five filter blank samples for DOC, particularly during the first and last sampling rounds. As a result, the DOC analytical results for all piezometers were either: 1) rejected if the sample result was below the respective filter blank value, or 2) qualified if the sample result was above the respective blank value.

Field Duplicates

Blind field duplicate samples were submitted to the Manchester Laboratory during each sampling round. Duplicate samples were collected by splitting the pump discharge between two sets of samples bottles. First round duplicate samples were selected randomly; duplicate locations selected for subsequent rounds were chosen on the basis of previous 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 F-2 presents the reported concentration data for each of the duplicate pairs grouped by parameter, and shows the %RSD calculated for each pair. Figure F-1 presents a chart for each of the target analytes showing the %RSD for each duplicate pair vs. the mean of the pair concentration. These charts indicate the duplicate precision for all of the parameters was normally within acceptable limits in the concentration ranges of interest (see Pitz, 2002).

The mean %RSD for all duplicate pairs for orthophosphate-P was 11.7%, a value greater than the target %RSD identified in the project plan. The precision for the duplicate pairs with mean concentrations below 0.1 mg/L was within acceptable limits, but the %RSD (using the pooled standard deviation) of the five duplicate pairs with mean concentrations above 0.1 mg/L was >21%. While the data values are still considered acceptable for the purposes of the study, all orthophosphate-P concentrations reported above 0.1 mg/L have been qualified as estimates by the author.

The mean %RSD for DOC was also above the established target, but this was largely due to a large value (>81%) reported for one of the duplicate pairs collected in October. As a result of the poor duplicate performance in October (in addition to the high level of blank interference), all DOC data for that round were rejected by the author.

Precision estimates for several of the parameters were occasionally above the target %RSD 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.

Field Replicate

As an additional test of the influence of heterogeneity in aquifer conditions, a field replicate sample was collected during the final sampling round from station LWGW-05. After collection of the normal sample, the piezometer was allowed to re-equilibrate to the initial head condition recorded at the station, and a second sampling event occurred. Table F-3 presents the estimated %RSD for each replicate pair for each parameter analyzed. The mean %RSD was less than 2% and no parameter exceeded 10% RSD, indicating good consistency between sampling events.

Table F-1. Filter Blank Data

Analyte	Oct '02(1)			Jan '03(1)			Apr '03(2)			Apr '03(1)			Aug '03(1)		
	Value	Units	Qualifier	Value	Units	Qualifier	Value	Units	Qualifier	Value	Units	Qualifier	Value	Units	Qualifier
Orthophosphate-P	0.003	mg/L	U	0.003	mg/L	U	0.003	mg/L	U	0.003	mg/L	U	0.0030	mg/L	U
TDP	0.010	mg/L	U	0.010	mg/L	U	0.010	mg/L	U	0.010	mg/L	U	0.010	mg/L	U
TDS	**3**	mg/L		1	mg/L	U	**6.3**	mg/L		1	mg/L	U	10	mg/L	U
Chloride	**0.16**	mg/L		0.1	mg/L	U	0.10	mg/L	U	0.10	mg/L	U	0.10	mg/L	U
Boron	50	ug/L	U	50	ug/L	U	50	ug/L	U	50	ug/L	U	50	ug/L	U
Nitrate+nitrite-N	0.010	mg/L	U	0.010	mg/L	U	0.010	mg/L	U	0.010	mg/L	U	0.010	mg/L	U
Ammonia	0.010	mg/L	U	0.010	mg/L	U	0.010	mg/L	U	0.010	mg/L	U	0.010	mg/L	U
DOC	**8.8**	mg/L		**1.2**	mg/L		**1.3**	mg/L		**1.8**	mg/L		**3.0**	mg/L	
Iron	50	ug/L	U	50	ug/L	U	50	ug/L	U	50	ug/L	U	50	ug/L	U
Manganese	10	ug/L	U	10	ug/L	U	10	ug/L	U	10	ug/L	U	10	ug/L	U

U - not detected at or above the reporting limit

Shaded values denote a detection

(1) Blank was collected through fresh collection equipment.

(2) Blank was collected through collection equipment used at a previous sampling station.

Table F-2. Field Duplicate Data

Total Dissolved Solids

Sample Round	Station	Concentration	Units	Qual.	Abs. Diff. [D]	Std. Dev. s	Mean x	RSD% s/x*100
1	LWGW-07	186	mg/L					
1	LWGW-07 Dup	184	mg/L		2.0	1.41	185.0	0.8
1	LWGW-09	277	mg/L					
1	LWGW-09 Dup	276	mg/L		1.0	0.71	276.5	0.3
2	LWGW-05	111	mg/L					
2	LWGW-05 Dup	112	mg/L		1.0	0.71	111.5	0.6
2	LWGW-12	101	mg/L					
2	LWGW-12 Dup	101	mg/L		0.0	0.00	101.0	0.0
3	LWGW-04	213	mg/L					
3	LWGW-04 Dup	215	mg/L		2.0	1.41	214.0	0.7
3	LWGW-14	42.6	mg/L					
3	LWGW-14 Dup	37.5	mg/L		5.1	3.61	40.1	9.0
4	LWGW-02	NA	mg/L					
4	LWGW-02 Dup	NA	mg/L		NA	NA	NA	NA
4	LWGW-09	271	mg/L					
4	LWGW-09 Dup	275	mg/L		4.0	2.83	273.0	1.0
						Mean RSD%		1.8

Table F-2. Field Duplicate Data (cont.)

Chloride

Sample Round	Station	Concentration	Units	Qual.	Abs. Diff. [D]	Std. Dev. s	Mean x	RSD% s/x*100
1	LWGW-07	5.02	mg/L					
1	LWGW-07 Dup	5.03	mg/L		0.01	0.007	5.03	0.1
1	LWGW-09	3.27	mg/L					
1	LWGW-09 Dup	6.51	mg/L		3.24	2.291	4.89	46.9
2	LWGW-05	2.57	mg/L					
2	LWGW-05 Dup	2.47	mg/L		0.10	0.071	2.52	2.8
2	LWGW-12	1.9	mg/L					
2	LWGW-12 Dup	1.8	mg/L		0.10	0.071	1.85	3.8
3	LWGW-04	3.27	mg/L					
3	LWGW-04 Dup	3.22	mg/L		0.05	0.035	3.25	1.1
3	LWGW-14	2.16	mg/L					
3	LWGW-14 Dup	2.14	mg/L		0.02	0.014	2.15	0.7
4	LWGW-02	55.6	mg/L					
4	LWGW-02 Dup	57.5	mg/L		1.90	1.344	56.55	2.4
4	LWGW-09	6.88	mg/L					
4	LWGW-09 Dup	6.99	mg/L		0.11	0.08	6.94	1.1
						Mean RSD%		7.4

Table F-2. Field Duplicate Data (cont.)

Ammonia-N

Sample Round	Station	Concentration	Units	Qual.	Abs. Diff. [D]	Std. Dev. s	Mean x	RSD% s/x*100
1	LWGW-07	0.193	mg/L					
1	LWGW-07 Dup	0.190	mg/L		0.003	0.002	0.192	1.1
1	LWGW-09	0.272	mg/L					
1	LWGW-09 Dup	0.267	mg/L		0.005	0.004	0.270	1.3
2	LWGW-05	0.637	mg/L					
2	LWGW-05 Dup	0.649	mg/L		0.012	0.008	0.643	1.3
2	LWGW-12	0.167	mg/L					
2	LWGW-12 Dup	0.168	mg/L		0.001	0.001	0.168	0.4
3	LWGW-04	0.178	mg/L					
3	LWGW-04 Dup	0.174	mg/L		0.004	0.003	0.176	1.6
3	LWGW-14	0.052	mg/L					
3	LWGW-14 Dup	0.051	mg/L		0.001	0.001	0.052	1.4
4	LWGW-02	1.36	mg/L					
4	LWGW-02 Dup	1.43	mg/L		0.070	0.049	1.395	3.5
4	LWGW-09	0.355	mg/L					
4	LWGW-09 Dup	0.334	mg/L		0.021	0.01	0.345	4.3
						Mean RSD%		1.9

Table F-2. Field Duplicate Data (cont.)

Nitrate-N

Sample Round	Station	Concentration	Units	Qual.	Abs. Diff. [D]	Std. Dev. s	Mean x	RSD% s/x*100
1	LWGW-07	0.01	mg/L	U				
1	LWGW-07 Dup	0.01	mg/L	U	0.000	0.000	0.010	0.0
1	LWGW-09	0.016	mg/L					
1	LWGW-09 Dup	0.017	mg/L		0.001	0.001	0.017	4.3
2	LWGW-05	0.017	mg/L					
2	LWGW-05 Dup	0.018	mg/L		0.001	0.001	0.018	4.0
2	LWGW-12	0.010	mg/L	U				
2	LWGW-12 Dup	0.010	mg/L	U	0.000	0.000	0.010	0.0
3	LWGW-04	0.010	mg/L	U				
3	LWGW-04 Dup	0.010	mg/L	U	0.000	0.000	0.010	0.0
3	LWGW-14	0.010	mg/L	U				
3	LWGW-14 Dup	0.010	mg/L	U	0.000	0.000	0.010	0.0
4	LWGW-02	0.010	mg/L	U				
4	LWGW-02 Dup	0.010	mg/L	U	0.000	0.000	0.010	0.0
4	LWGW-09	0.010	mg/L	U				
4	LWGW-09 Dup	0.010	mg/L	U	0.000	0.00	0.010	0.0
							Mean RSD%	1.0

U - Not detected at or above the reporting limit

Table F-2. Field Duplicate Data (cont.)

Orthophosphate-P

Sample Round	Station	Concentration	Units	Qual.	Abs. Diff. [D]	Std. Dev. s	Mean x	RSD% s/x*100
1	LWGW-07	0.0348	mg/L					
1	LWGW-07 Dup	0.0337	mg/L		0.001	0.001	0.034	2.3
1	LWGW-09	0.298	mg/L					
1	LWGW-09 Dup	0.396	mg/L		0.098	0.069	0.347	20.0
2	LWGW-05	0.117	mg/L					
2	LWGW-05 Dup	0.159	mg/L		0.042	0.030	0.138	21.5
2	LWGW-12	0.245	mg/L					
2	LWGW-12 Dup	0.248	mg/L		0.003	0.002	0.247	0.9
3	LWGW-04	0.0585	mg/L					
3	LWGW-04 Dup	0.0597	mg/L		0.001	0.001	0.059	1.4
3	LWGW-14	0.030	mg/L					
3	LWGW-14 Dup	0.0300	mg/L		0.000	0.000	0.030	0.0
4	LWGW-02	0.116	mg/L					
4	LWGW-02 Dup	0.0925	mg/L		0.024	0.017	0.104	15.9
4	LWGW-09	0.245	mg/L					
4	LWGW-09 Dup	0.156	mg/L		0.089	0.06	0.201	31.4
						Mean RSD%		11.7

Table F-2. Field Duplicate Data (cont.)

Total Dissolved Phosphorus

Sample Round	Station	Concentration	Units	Qual.	Abs. Diff. [D]	Std. Dev. s	Mean x	RSD% s/x*100
1	LWGW-07	0.057	mg/L					
1	LWGW-07 Dup	0.057	mg/L		0.000	0.000	0.057	0.0
1	LWGW-09	2.18	mg/L					
1	LWGW-09 Dup	0.706	mg/L		1.474	1.042	1.443	72.2
2	LWGW-05	0.359	mg/L					
2	LWGW-05 Dup	0.359	mg/L		0.000	0.000	0.359	0.0
2	LWGW-12	0.398	mg/L					
2	LWGW-12 Dup	0.401	mg/L		0.003	0.002	0.400	0.5
3	LWGW-04	0.081	mg/L					
3	LWGW-04 Dup	0.083	mg/L		0.002	0.001	0.082	1.7
3	LWGW-14	0.038	mg/L					
3	LWGW-14 Dup	0.039	mg/L		0.001	0.001	0.039	1.8
4	LWGW-02	0.252	mg/L					
4	LWGW-02 Dup	0.254	mg/L		0.002	0.001	0.253	0.6
4	LWGW-09	2.14	mg/L					
4	LWGW-09 Dup	2.13	mg/L		0.010	0.01	2.135	0.3
						Mean RSD%		9.7

Table F-2. Field Duplicate Data (cont.)

Dissolved Organic Carbon								
Sample Round	Station	Concentration	Units	Qual.	Abs. Diff. [D]	Std. Dev. s	Mean x	RSD% s/x*100
1	LWGW-07	7.9	mg/L					
1	LWGW-07 Dup	8.0	mg/L		0.1	0.07	7.95	0.9
1	LWGW-09	9.0	mg/L					
1	LWGW-09 Dup	33.4	mg/L		24.4	17.25	21.20	81.4
2	LWGW-05	6.9	mg/L	J				
2	LWGW-05 Dup	6.9	mg/L	J	0.0	0.00	6.90	0.0
2	LWGW-12	2.3	mg/L	J				
2	LWGW-12 Dup	2.3	mg/L	J	0.0	0.00	2.30	0.0
3	LWGW-04	1.1	mg/L					
3	LWGW-04 Dup	1.3	mg/L		0.2	0.14	1.20	11.8
3	LWGW-14	2.6	mg/L					
3	LWGW-14 Dup	2.4	mg/L		0.2	0.14	2.50	5.7
4	LWGW-02	9.6	mg/L					
4	LWGW-02 Dup	9.8	mg/L		0.2	0.14	9.70	1.5
4	LWGW-09	22.8	mg/L					
4	LWGW-09 Dup	23.2	mg/L		0.4	0.28	23.00	1.2
						Mean RSD%		12.8

J - Reported result is an estimate

Table F-2. Field Duplicate Data (cont.)

Boron

Sample Round	Station	Concentration	Units	Qual.	Abs. Diff. [D]	Std. Dev. s	Mean x	RSD% s/x*100
1	LWGW-07	79	ug/L					
1	LWGW-07 Dup	77	ug/L		2	1.4	78.0	1.8
1	LWGW-09	50	ug/L	U				
1	LWGW-09 Dup	50	ug/L	U	0	0.0	50.0	0.0
2	LWGW-05	65	ug/L					
2	LWGW-05 Dup	65	ug/L		0	0.0	65.0	0.0
2	LWGW-12	75	ug/L					
2	LWGW-12 Dup	79	ug/L		4	2.8	77.0	3.7
3	LWGW-04	50	ug/L	U				
3	LWGW-04 Dup	50	ug/L	U	0	0.0	50.0	0.0
3	LWGW-14	50	ug/L	U				
3	LWGW-14 Dup	50	ug/L	U	0	0.0	50.0	0.0
4	LWGW-02	59	ug/L					
4	LWGW-02 Dup	62	ug/L		3	2.1	60.5	3.5
4	LWGW-09	50	ug/L	U				
4	LWGW-09 Dup	50	ug/L	U	0	0.0	50.0	0.0
							Mean RSD%	1.1

U - Not detected at or above the reporting limit

Table F-2. Field Duplicate Data (cont.)

Manganese

Sample Round	Station	Concentration	Units	Qual.	Abs. Diff. [D]	Std. Dev. s	Mean x	RSD% s/x*100
1	LWGW-07	81	ug/L					
1	LWGW-07 Dup	83	ug/L		2	1.4	82.0	1.7
1	LWGW-09	1500	ug/L					
1	LWGW-09 Dup	1510	ug/L		10	7.1	1505.0	0.5
2	LWGW-05	112	ug/L					
2	LWGW-05 Dup	112	ug/L		0	0.0	112.0	0.0
2	LWGW-12	440	ug/L					
2	LWGW-12 Dup	453	ug/L		13	9.2	446.5	2.1
3	LWGW-04	20	ug/L					
3	LWGW-04 Dup	20	ug/L		0	0.0	20.0	0.0
3	LWGW-14	99	ug/L					
3	LWGW-14 Dup	98	ug/L		1	0.7	98.5	0.7
4	LWGW-02	1360	ug/L					
4	LWGW-02 Dup	1350	ug/L		10	7.1	1355.0	0.5
4	LWGW-09	1570	ug/L	U				
4	LWGW-09 Dup	1560	ug/L	U	10	7.1	1565.0	0.5
							Mean RSD%	0.7

U - Not detected at or above the reporting limit

Table F-2. Field Duplicate Data (cont.)

Iron

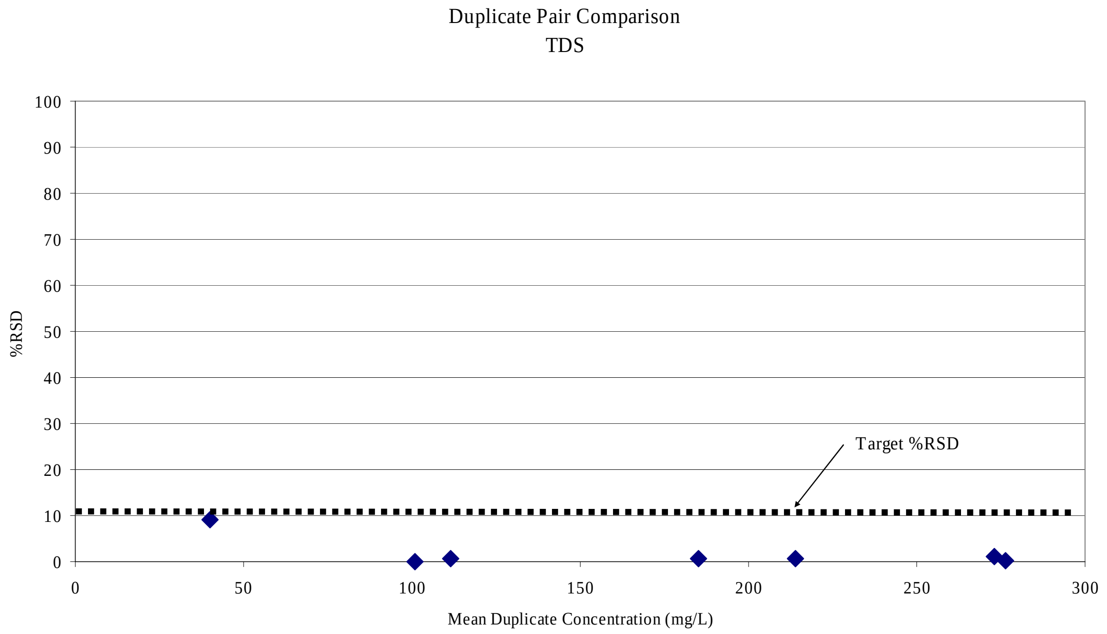
Sample Round	Station	Concentration	Units	Qual.	Abs. Diff. [D]	Std. Dev. s	Mean x	RSD% s/x*100
1	LWGW-07	50	ug/L	U				
1	LWGW-07 Dup	50	ug/L	U	0	0.0	50	0.0
1	LWGW-09	35700	ug/L					
1	LWGW-09 Dup	36000	ug/L		300	212.1	35850	0.6
2	LWGW-05	12100	ug/L					
2	LWGW-05 Dup	12000	ug/L		100	70.7	12050	0.6
2	LWGW-12	1300	ug/L					
2	LWGW-12 Dup	1360	ug/L		60	42.4	1330	3.2
3	LWGW-04	140	ug/L					
3	LWGW-04 Dup	140	ug/L		0	0.0	140	0.0
3	LWGW-14	1310	ug/L					
3	LWGW-14 Dup	1300	ug/L		10	7.1	1305	0.5
4	LWGW-02	37100	ug/L					
4	LWGW-02 Dup	37100	ug/L		0	0.0	37100	0.0
4	LWGW-09	38000	ug/L					
4	LWGW-09 Dup	37900	ug/L		100	70.7	37950	0.2
							Mean RSD%	0.6

U - Not detected at or above the reporting limit

Table F-3. Field Replicate Data

Sample Round	Station	Analyte	Concentration	Units	Qual.	Abs. Diff. [D]	Std. Dev. s	Mean x	RSD% s/x*100
4	LWGW-05	TDS	102	mg/L					
4	LWGW-05 Rep	TDS	102	mg/L		0	0.0	102.0	0.0
4	LWGW-05	chloride	2.82	mg/L					
4	LWGW-05 Rep	chloride	2.84	mg/L		0	0.0	2.8	0.5
4	LWGW-05	ammonia-N	0.652	mg/L					
4	LWGW-05 Rep	ammonia-N	0.663	mg/L		0	0.0	0.7	1.2
4	LWGW-05	nitrate-N	0.010	mg/L					
4	LWGW-05 Rep	nitrate-N	0.010	mg/L		0	0.0	0.0	0.0
4	LWGW-05	orthophosphate-P	0.0764	mg/L					
4	LWGW-05 Rep	orthophosphate-P	0.0709	mg/L		0	0.0	0.1	5.3
4	LWGW-05	TDP	0.363	mg/L					
4	LWGW-05 Rep	TDP	0.407	mg/L		0	0.0	0.4	8.1
4	LWGW-05	boron	50	ug/L	U				
4	LWGW-05 Rep	boron	50	ug/L	U	0	0.0	50.0	0.0
4	LWGW-05	DOC	6.3	mg/L					
4	LWGW-05 Rep	DOC	6.4	mg/L		0	0.1	6.4	1.1
4	LWGW-05	manganese	96	ug/L					
4	LWGW-05 Rep	manganese	97	ug/L		1	0.7	96.5	0.7
4	LWGW-05	iron	10400	ug/L	U				
4	LWGW-05 Rep	iron	10600	ug/L	U	200	141	10500	1.3

U - Not detected at or above the reporting limit



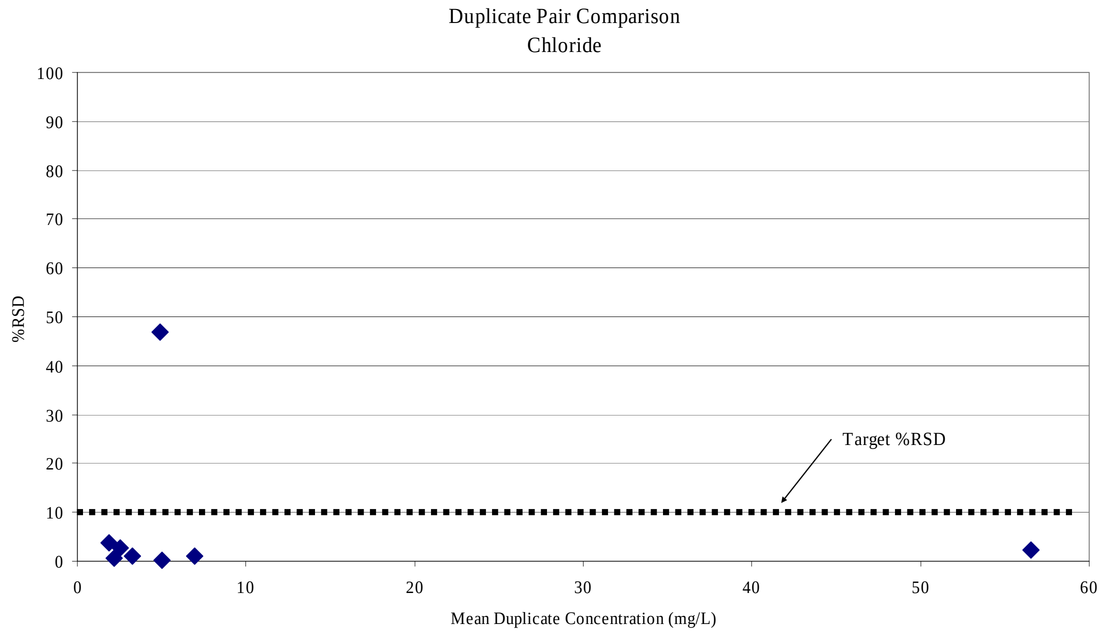
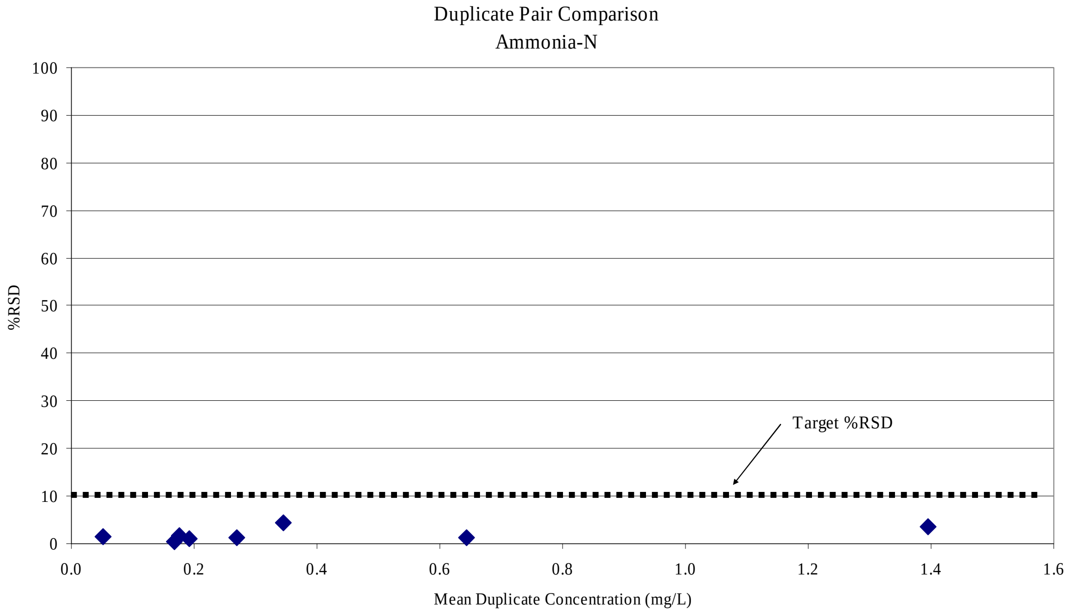


Figure F-1
Blind Field Duplicate Precision (%RSD)
As a Function of Concentration



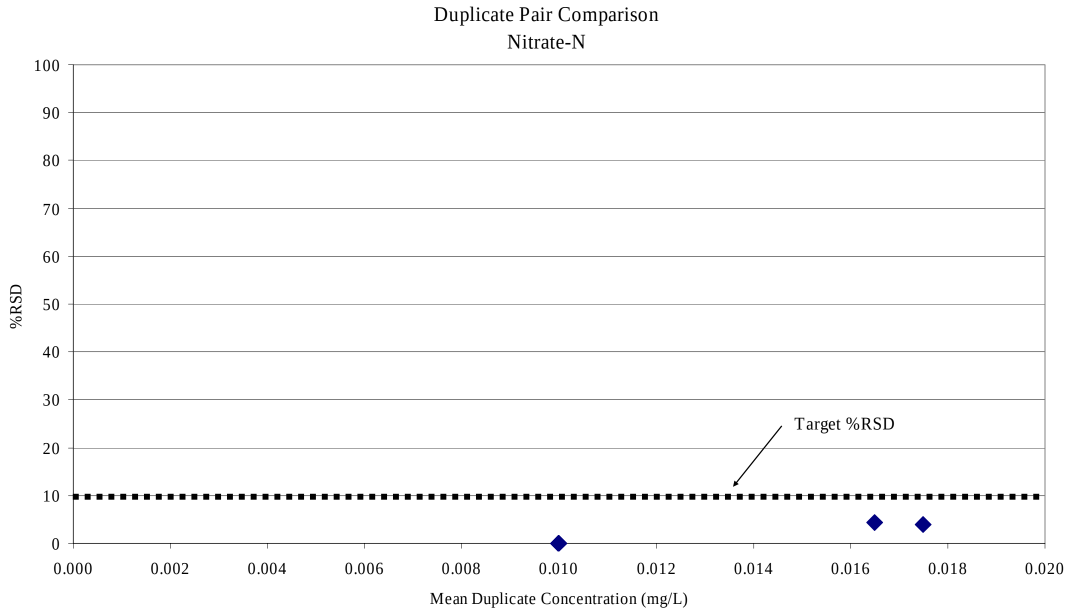
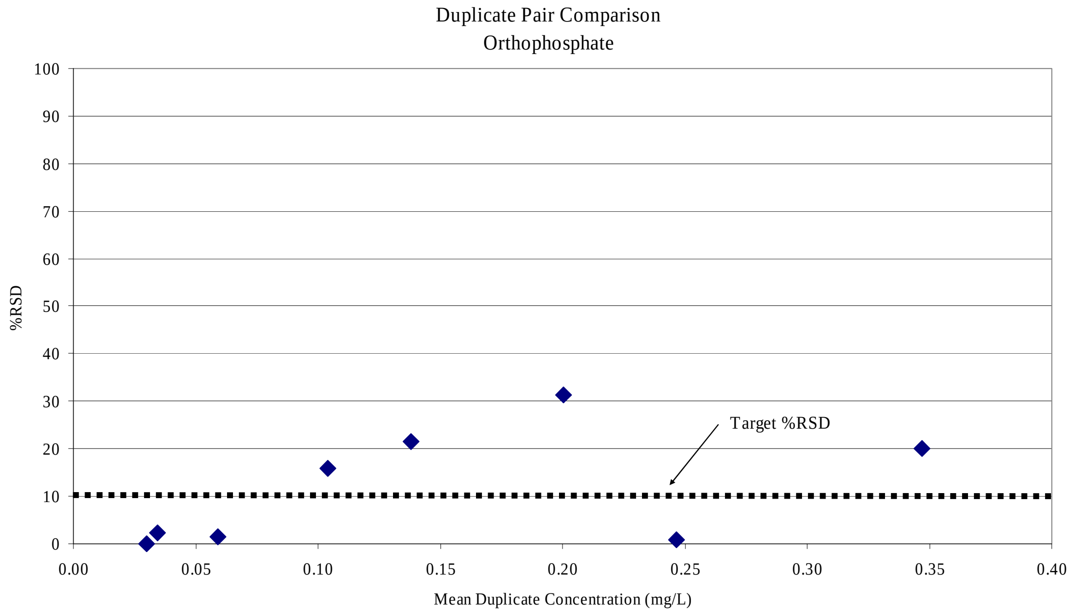


Figure F-1 (cont.)
Blind Field Duplicate Precision (%RSD)
As a Function of Concentration



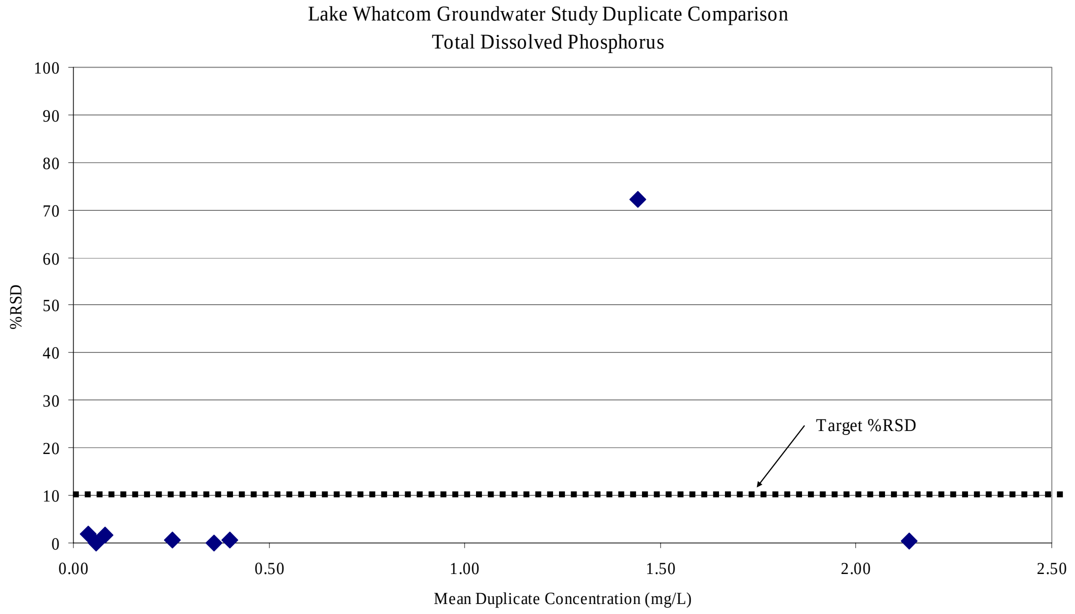


Figure F-1 (cont.)
Blind Field Duplicate Precision (%RSD)
As a Function of Concentration

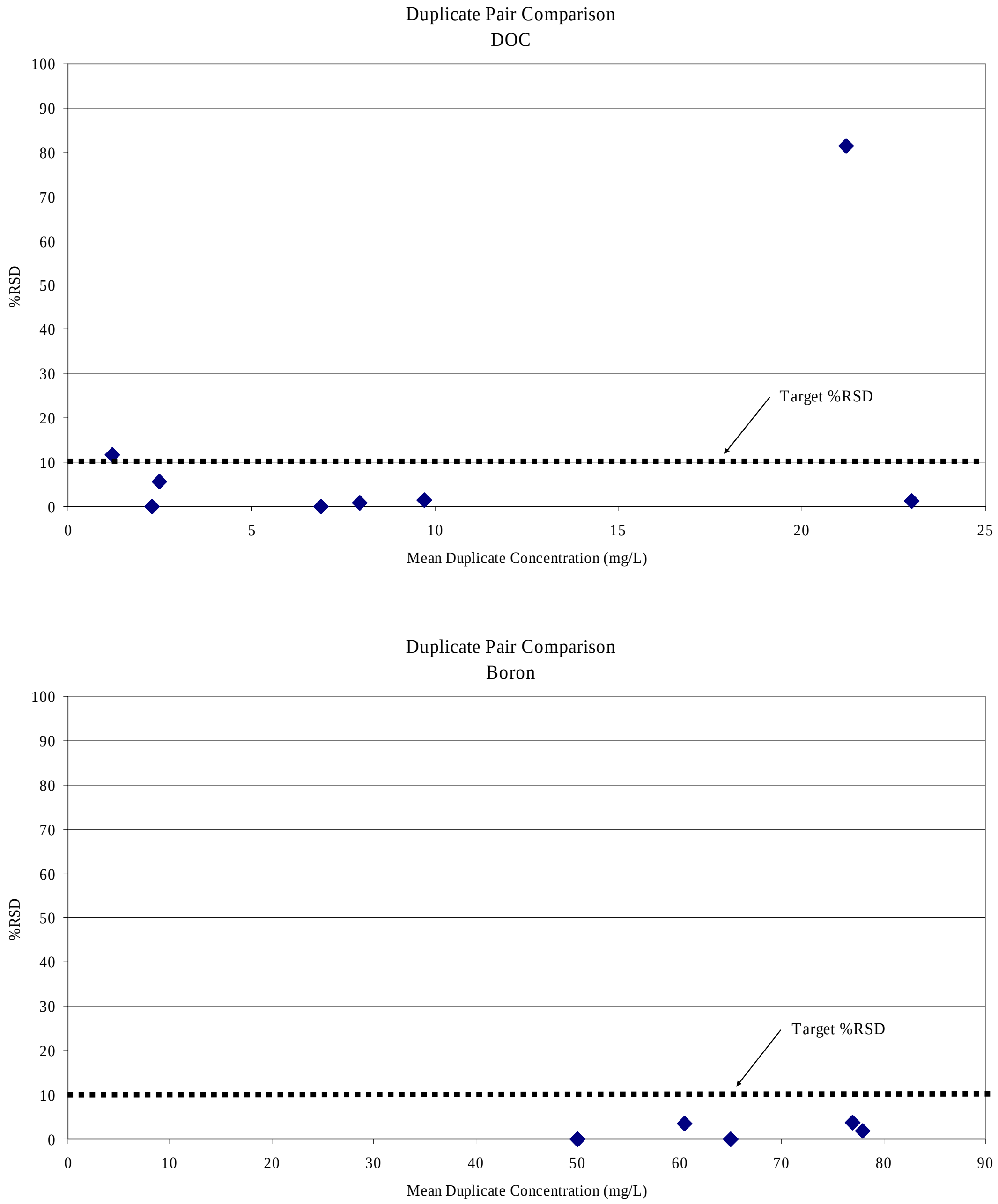
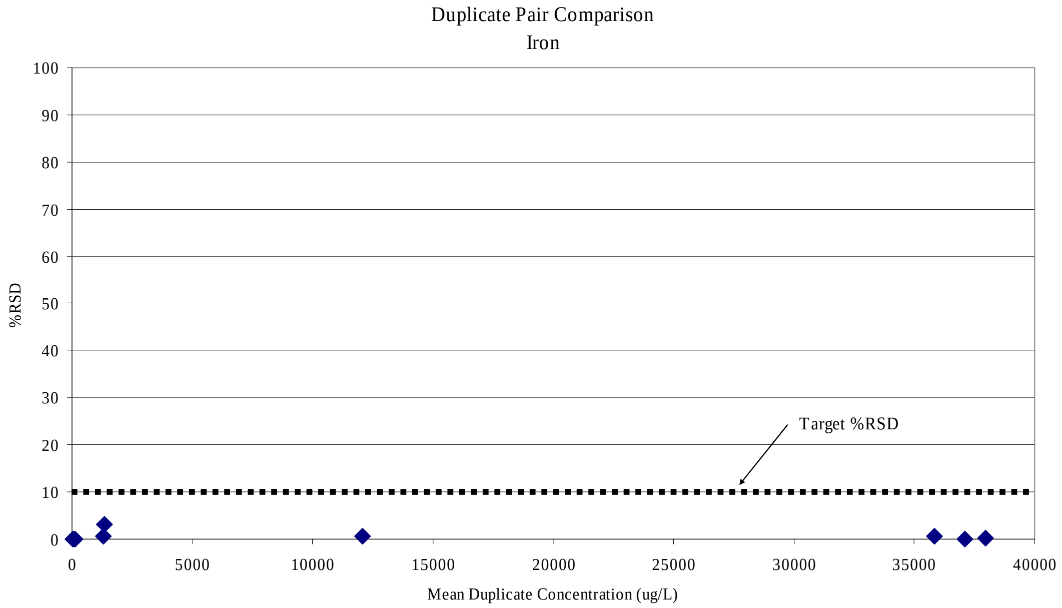


Figure F-1 (cont.)
Blind Field Duplicate Precision (%RSD)
As a Function of Concentration



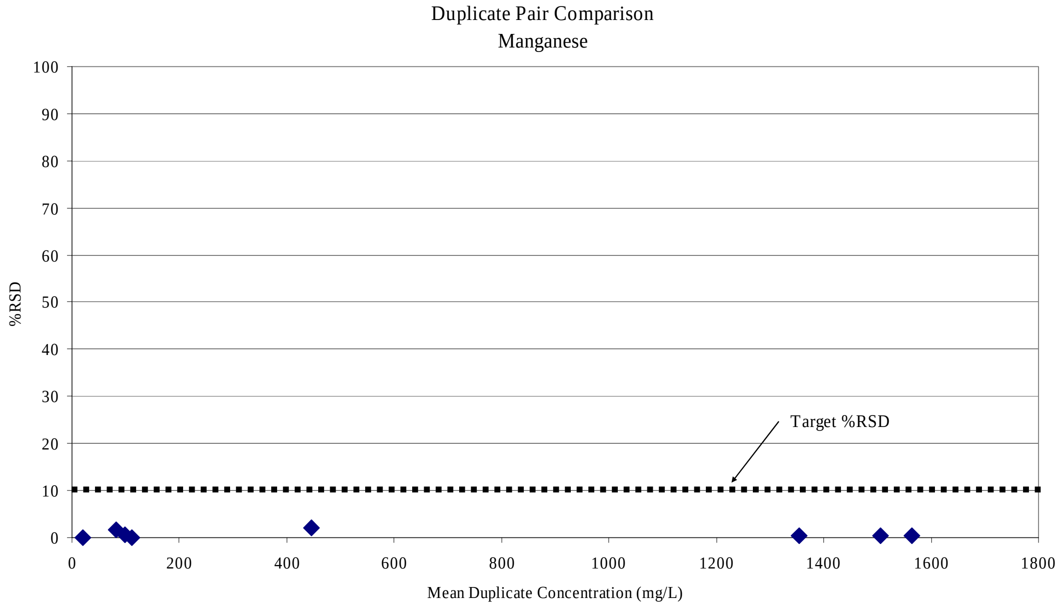


Figure F-1 (cont.)
Blind Field Duplicate Precision (%RSD)
As a Function of Concentration

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Appendix G. Piezometer Head and Gradient Data

Table G-1. Piezometer Head Measurements and Vertical Hydraulic Gradient Estimates

Station Name	Total depth of piezometer below sediment surface (ft)	Depth of screen mid-point below sediment surface *(z)* (ft)	Depth of screen mid-point below sediment surface *(z)* (mm)	Head relative to lake surface Oct '02 *(h)* (mm)	Estimated vertical gradient Oct '02 *(i=h/z)*	Head relative to lake surface Jan '03 *(h)* (mm)	Estimated vertical gradient Jan '03 *(i=h/z)*	Head relative to lake surface Apr '03 *(h)* (mm)	Estimated vertical gradient Apr '03 *(i=h/z)*	Head relative to lake surface Aug '03 *(h)* (mm)	Estimated vertical gradient Aug '03 *(i=h/z)*
LWGW-01	5.3	4.8	1463	26	0.018	55	0.038	40	0.027	30	0.021
LWGW-02	5.1	4.6	1402	0	0.000	3	0.002	0	0.000	0	0.000
LWGW-03	4.1	3.6	1097	13	0.012	22	0.020	50	0.046	NM	NM
LWGW-04	4.9	4.4	1341	10	0.007	4	0.003	NM	NM	0	0.000
LWGW-05	4.8	4.3	1311	4	0.003	1	0.001	5	0.004	0	0.000
LWGW-06	5.0	4.5	1372	24	0.017	NM	NM	NM	NM	NM	NM
LWGW-07	4.3	3.8	1158	61	0.053	76	0.066	110	0.095	28	0.024
LWGW-08	4.1	3.6	1097	6	0.005	10	0.009	NM	NM	6	0.005
LWGW-09	4.9	4.4	1341	10	0.007	15	0.011	55	0.041	16	0.012
LWGW-10	4.6	4.1	1250	36	0.029	30	0.024	40	0.032	35	0.028
LWGW-11	4.2	3.7	1128	57	0.051	55	0.049	60	0.053	62	0.055
LWGW-12	4.7	4.2	1280	5	0.004	3	0.002	NM	NM	2	0.002
LWGW-13	4.8	4.3	1311	3	0.002	0	0.000	5	0.004	**-6**	**-0.005**
LWGW-14	4.4	3.9	1189	30	0.025	133	0.112	20	0.017	13	0.011

NM - Not measured

Shaded values indicate a negative (downward) gradient between the lake and groundwater, suggesting lake recharge of the aquifer at that location. Positive vertical gradient values indicate groundwater discharge to the lake.

Appendix H. Water Quality Results

Table H-1. Groundwater Quality Results

Station	Total Dissolved Solids (TDS) (mg/L)								Chloride (dissolved) (mg/L)							
	Oct '02		Jan '03		Apr/May '03		Aug '03		Oct '02		Jan '03		Apr/May '03		Aug '03	
LWGW-01	**244**		**257**		**265**		**291**		53.2		NM		NM		NM	
LWGW-02	496		531		508		515		53.1		61.2		56.9		55.6	
LWGW-03	298		**265**		296		NM		7.92		NM		8.23		NM	
LWGW-04	203		210		213		207		3.02		3.19		3.27		3.18	
LWGW-05	110		111		104		102		2.64		2.57		2.53		2.82	
LWGW-06	**224**		NM		NM		NM		NM		NM		NM		NM	
LWGW-07	186		175		**180**		183		5.02		5.18		5.24		5.20	
LWGW-08	159		151		NM		155		4.17		3.99		NM		4.15	
LWGW-09	277		261		**226**		271		3.27		7.06		6.88		6.88	
LWGW-10	**90**		**90**		**92**		114		NM		NM		NM		2.77	
LWGW-11	157		**143**		**145**		157		1.86		NM		1.83		1.92	
LWGW-12	106		101		104		104		1.84		1.9		1.85		1.93	
LWGW-13	164		161		165		160		2.40		2.22		2.20		2.62	
LWGW-14	38		45		42.6		40		1.86		2.37		2.16		1.95	

Station	Ammonia-N (dissolved) (mg/L)								Nitrate+Nitrite-N (dissolved) (mg/L)							
	Oct '02		Jan '03		Apr/May '03		Aug '03		Oct '02		Jan '03		Apr/May '03		Aug '03	
LWGW-01	0.083		0.077		0.085		0.119		0.011		0.010	U	0.010	U	0.010	U
LWGW-02	1.02		1.26		1.27		1.36		0.014		0.019		0.033		0.010	U
LWGW-03	0.756		0.923		0.920		NM		0.010	U	0.011		0.012		NM	
LWGW-04	0.187		0.193		0.178		0.217		0.010	U	0.010	U	0.010	U	0.010	U
LWGW-05	0.607		0.637		0.605		0.652		0.016		0.017		0.026		0.010	U
LWGW-06	NM		NM		NM		NM		NM		NM		NM		NM	
LWGW-07	0.193		0.209		0.178		0.204	J	0.010	U	0.010	U	0.010	U	0.010	UJ
LWGW-08	0.010	U	0.010	U	NM		0.010	U	0.010	U	0.010	U	NM		0.010	U
LWGW-09	0.272		0.255		0.236		0.355		0.016		0.020		0.039		0.010	U
LWGW-10	0.381		0.358		0.356		0.400		0.010	U	0.010	U	0.011		0.010	U
LWGW-11	1.67		1.60		1.67		1.78		0.010	U	0.010	U	0.010	U	0.010	U
LWGW-12	0.188		0.167		0.167		0.205		0.010	U	0.010	U	0.010	U	0.010	U
LWGW-13	0.079		0.079		0.068		0.098		0.025		0.024		0.029		0.010	U
LWGW-14	0.059		0.054		0.052		0.063		0.010	U	0.010	U	0.010	U	0.010	U

NM - Not Measured.
J - Result is considered an estimate by the laboratory.
U - Analyte not detected at or above the reporting limit.
Shaded values for TDS concentration are estimated from field specific conductance (see text).

Table H-1. Groundwater Quality Results (cont.)

	Orthophosphate-P (dissolved) (mg/L)								Total Dissolved P (mg/L)							
Station	Oct '02		Jan '03		Apr/May '03		Aug '03		Oct '02		Jan '03		Apr/May '03		Aug '03	
LWGW-01	0.114	D	0.110	D	0.104	D	0.118	D	0.146		0.145		0.134		0.143	
LWGW-02	0.0302		0.0310		0.108	D	0.116	D	0.236		0.215		0.239		0.252	
LWGW-03	0.016		0.020		0.016		NM		0.097		0.081		0.089		NM	
LWGW-04	0.0631		0.0613		0.0585		0.0631		0.095		0.077		0.081		0.088	
LWGW-05	0.145	D	0.117	D	0.111	D	0.0764		0.362		0.359		0.358		0.363	
LWGW-06	0.021		NM		NM		NM		NM		NM		NM		NM	
LWGW-07	0.0348		0.0368		0.0373		0.0410		0.057		0.049		0.055		0.061	J
LWGW-08	0.010		0.0091		NM		0.0074		0.025		0.015		NM		0.025	
LWGW-09	0.298	D	0.349	D	0.136	D	0.245	D	2.18		2.00		2.03		2.14	
LWGW-10	0.040		0.028		0.023		0.029		0.143		0.123		0.133		0.140	
LWGW-11	0.088		0.110	D	0.110	D	0.141	D	0.246		0.187		0.244		0.242	
LWGW-12	0.291	D	0.245	D	0.212	D	0.280	D	0.411		0.398		0.379		0.492	
LWGW-13	0.013		0.016		0.013		0.0068		0.087		0.063		0.069		0.084	
LWGW-14	0.0339		0.028		0.030		0.029		0.047		0.033		0.038		0.046	

	Dissolved Organic Carbon (DOC) (mg/L)								Boron (dissolved) (µg/L)							
Station	Oct '02		Jan '03		Apr/May '03		Aug '03		Oct '02		Jan '03		Apr/May '03		Aug '03	
LWGW-01	R		4.6	JF	5.2	F	9.8	F	136		110		100		140	
LWGW-02	R		9.7	F	8.6	F	9.6	F	63		50	U	50	U	59	
LWGW-03	R		3.2	F	2.8	F	NM		88		70		61		NM	
LWGW-04	R		1.5	JF	R		R		56		50	U	50	U	56	
LWGW-05	R		6.9	JF	5.7	F	6.3	F	50	U	65		50	U	50	U
LWGW-06	NM		NM		NM		NM		NM		NM		NM		NM	
LWGW-07	R		1.4	F	R	F	R		79		62		66		79	
LWGW-08	R		R		NM		R		50	U	50	U	NM		50	U
LWGW-09	R		23.2	JF	16.3	F	22.8	F	50	U	50	U	50	U	50	U
LWGW-10	R		3.7	JF	3.0	F	3.2	F	120		98		88		120	
LWGW-11	R		4.0	F	3.3	F	6.4	F	50	U	50	U	50	U	50	U
LWGW-12	R		2.3	JF	R	F	R		91		75		71		110	
LWGW-13	R		3.8	JF	2.3	F	3.6	F	270		240		160		270	
LWGW-14	R		3.7	JF	2.6	F	3.6	F	50	U	50	U	50	U	50	U

NM - Not Measured.
D - %RSD precision at this concentration range is above target goal; value considered an estimate by the author.
J - Result is considered an estimate by the laboratory.
U - Analyte not detected at or above the reporting limit.
F - Filter blank indicates reported concentration may be biased high.
R - Laboratory result rejected due to high filter blank concentration or poor duplicate precision.

Table H-1. Groundwater Quality Results (cont.)

	Manganese (dissolved) (µg/L)								Iron (dissolved) (µg/L)							
Station	Oct '02		Jan '03		Apr/May '03		Aug '03		Oct '02		Jan '03		Apr/May '03		Aug '03	
LWGW-01	132		154		91		87		1640		2120		110		50	U
LWGW-02	1290		1330		1370		1360		35200		35200		35800		37100	
LWGW-03	304		211		211		NM		3670		3600		3850		NM	
LWGW-04	21		20		20		18		120		150		140		140	
LWGW-05	111		112		114		96		11600		12100		12100		10400	
LWGW-06	NM		NM		NM		NM		NM		NM		NM		NM	
LWGW-07	81		75		69		69		50	U	53		50	U	50	U
LWGW-08	18		18		NM		22		50	U	50	U	NM		50	U
LWGW-09	1500		1550		1630		1570		35700		36200		37900		38000	
LWGW-10	199		197		211		205		3220		3040		3280		3160	
LWGW-11	193		160		163		163		2440		939		1220		1040	
LWGW-12	415		440		447		417		1280		1300		1390		1290	
LWGW-13	150		164		159		163		11600		13500		13800		13300	
LWGW-14	94		111		99		87		1310		1490		1310		1190	

NM - Not Measured.
U - Analyte not detected at or above the reporting limit.

Appendix I. Response to Public Comments on the Draft Report

The draft for this report was posted to the web at http://www.ecy.wa.gov/biblio/0403050.html on November 29, 2004, with public comments due to the author by January 3, 2005. All comments and responses by the author are listed in this appendix.

Comment 1

The relative bar heights shown on Figure 13 do not always match the reference scale bar.

Response: The figure has been corrected.

Comment 2

The average concentration displayed on Figure 17 for stations LWGW-06 and LWGW-07 is incorrect.

Response: The figure has been corrected.

Comment 3

The study does not address nutrient inflow to the lake via surface water and precipitation inputs, and does not evaluate the impact of internal loading of nutrients on the lake's water quality. Are there complimentary studies assessing the nutrient contribution from these sources?

Response: Complementary studies of basin surface water conditions and nutrient loading from sources other than groundwater are underway. The text of the report has been revised to mention these additional studies.

Comment 4

All possible sources of groundwater nutrient enrichment should be identified and addressed. Other potential sources could include sewage treatment plant effluent, leaking municipal sewer lines, historic and current use of on site septic systems, storm water infiltration dry wells, leachate from landfills, buried toxic and/or chemical waste sites, agricultural practices in upgradient permeable soils, etc.

Response: The primary goal of the study was to describe the current conditions of groundwater entering the lake, with a particular focus on the nutrient content of the inflowing water. Identifying the potential source of the nutrients measured was a secondary goal, and was addressed only where the water quality data most clearly suggested an unnatural groundwater condition. While it is true that the water quality condition of groundwater nearing the end of the flow path could exhibit the influence of multiple sources, the high reactivity of phosphorus (the key nutrient of interest) in the subsurface suggests that sources immediately upgradient of the piezometers are the most likely or strongest influence. The report recommended several stations where additional upgradient characterization and source identification might be justified. Presumably more detailed studies in these areas would address the possibility of alternative sources of nutrient enrichment further upgradient.

Comment 5

In the geologic setting of Lake Whatcom it appears that geochemical weathering is occurring because of prolonged contact of groundwater with bedrock. This phenomenon may be having far more influence on near shore groundwater geochemistry, even in areas of unconsolidated sediments, than is being recognized by the study's narrative.

Response: The author agrees that the *ambient* reducing condition of the groundwater measured in the study area is a result of normal geochemical weathering along the flow path, although this was never stated explicitly in the report. The anoxic conditions observed at some of the stations may, however, be accentuated by additional anthropogenic inputs of oxygen-consuming nutrients to the aquifer.

Comment 6

The report mentions the possibility that a significant percentage of the predicted groundwater phosphorus load may be attenuated in the final portions of the flow path. A quantitative estimate of this reduction should be presented; otherwise the loading figures stated in the report are likely to be utilized for modeling purposes, thereby creating an unintended bias.

Response: The study design did not provide the information necessary to model the geochemical reactions that can influence the dissolved-phase concentrations of phosphate as groundwater transitions through the redox front, and the attenuation can be difficult to measure accurately in the field. However, published values for phosphate phase partitioning in oxic vs. anoxic conditions suggest that the reduction could be as high as 90-95% in the final centimeters of the sediment column (e.g., DiToro, 2001). The groundwater loading and volume flux estimates presented in the report were intended to provide the developers of the hydrodynamic/ nutrient budget model a reasonable upper-limit on groundwater contributions to the lake. In the end, the best estimate of net groundwater nutrient flux will be derived through model calibration, after other components of the lake's water and nutrient budget are incorporated into the model.