AN ABSTRACT OF THE THESIS OF

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Abstract approved:

Roy D. Haggerty

Groundwater nitrate contamination is a well-documented issue in the Southern Willamette Valley (SWV) of Oregon, as a Groundwater Management Area (GWMA) has recently been declared. As a GWMA, groundwater nitrate monitoring must occur until regional concentrations are below 7 mg/L NO_3-N. However, the presence of temporal variability can make it difficult to determine if contamination exceeds a threshold and if contamination is increasing or decreasing over time. To examine the potential impact of temporal variability on groundwater nitrate monitoring in the SWV, a well network was created and sampled monthly for 15 months. Results indicate that substantial intra-well temporal variability is present, and that spatial variability of groundwater nitrate is greater than temporal variability. Generally, temporal variability was associated with recharge events, which flushed higher concentration soil-water into the aquifer. Though individual wells showed seasonality, network-wide seasonal trends were not statistically significant (which is believed to be caused by a dampening effect due to local heterogeneities). From a monitoring perspective, this implies that less frequent groundwater nitrate sampling (such as quarterly) can capture network-wide seasonal response to the same degree as monthly sampling.

To determine how long-term land management practices are likely to impact regional nitrate leaching and future monitoring trends, a nitrogen loading model was created for the SWV. Present-day data were used to calibrate and validate the Soil and Water Assessment Tool (SWAT) model, with 3 alternative future scenarios then being evaluated. The effects of agrarian Groundwater Best Management Practices (GW-BMPs) were examined with respect to nitrate leaching in present and future scenarios. Modeled

values indicate that agrarian GW-BMP implementation is a more effective agent for reduced nitrate leaching than land use change alone. Together, land use change and the adoption of GW-BMPs were found to decrease nitrate leaching values by 32 to 46% of their present-day rates. These predicted results do not include the impact of denitrification or changes in septic leaching, and therefore should be regarded with caution as they do not completely represent future conditions. Considering this, a conservative conclusion which can be drawn is that GW-BMP implementation is a safer alternative than reliance on projected land use/crop change alone for lessening groundwater nitrate concentrations in the GWMA. This is the first study to successfully apply SWAT as a tool to examine the spatial and temporal variability of nitrate leaching.

Temporal and Spatial Variability of Groundwater Nitrate in the Southern Willamette Valley of Oregon

by

Jeffrey Glenn Mutti

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APPROVED:

Professor, representing Geology and Water Resources Science

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Dean of the Graduate School

I understand that my thesis will become part of the permanent collection of Oregon State University libraries. My signature below authorizes release of my thesis to any reader upon request.

Jeffrey Glenn Mutti, Author

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CONTRIBUTION OF AUTHORS

Dr. Roy Haggerty assisted in the editing, design, and writing of Chapters 1-4.

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Temporal and Spatial Variability of Groundwater Nitrate in the Southern Willamette Valley of Oregon

1. General Introduction

1.1. Introduction

Nitrogen (N) is a key component to life processes and its global cycling is possibly the most altered biogeochemical cycle on earth (Vitousek et al., 1997). Human impact on the N cycle is manifested in numerous ways, including a doubling of the transfer of atmospheric N into biologically available N, increased global concentrations of the greenhouse gas N_2O, increased smog and acid rain locally, acidified ecosystems, declines in biodiversity, and increased plant uptakes of CO_2 (Vitousek et al., 1997). Increasing global populations and relatively inexpensive synthetic N fertilizer have caused world agriculture to greatly rely on N fertilizers to increase crop yields (Pierzynski et al., 2005). The manufacture of fertilizer is the single greatest anthropogenic source of fixed N to the environment (Holland et al., 2005). To maintain high agricultural productivity without excessive environmental impacts, efficient crop selection and educated fertilizer management practices must be employed.

Plants uptake N in the form of ammonium (NH_4^+) or nitrate (NO_3^-), with nitrate generally being the form which becomes an environmental concern if it is not consumed by plants or microbially assimilated. Ammonium that is not used in soil biological processes is generally retained on cation exchange sites, volatilized into NH_3, or nitrified into nitrate. Since nitrate is an anion that is highly soluble with virtually no retardation in soil water, it is a major leaching concern and the most commonly observed contaminant in groundwater (Nolan and Stoner, 2000). Nitrate has numerous anthropogenic sources, including fertilizers, septic drain fields, animal feeding operations, and atmospheric deposition. Due to the wide variety and nonpoint distribution of nitrate sources, it is a difficult contaminant to manage.

Nitrate can follow a number of fates after it exits the root zone, including microbial assimilation, denitrification, and dissimilatory reduction of nitrate to ammonium (Korum, 1992), as shown in Figure 1.1. Denitrification, the primary process which breaks down nitrate, is generally anaerobic and should not be expected to remove significant amounts in groundwater systems where dissolved oxygen concentrations are greater than 1 or 2 mg/L (MPCA, 1999).

Heterotropic denitrification, the most frequently observed form of denitrification, is limited by organic carbon availability, and thus in aerobic aquifers or anaerobic aquifers with limited carbon availability, nitrate can be expected to have long aquifer residence times. Therefore, even if Groundwater Best Management Practices (GW-BMP) for nitrate are implemented, high groundwater nitrate concentrations may persist for decades (Bohlke and Denver, 1995).

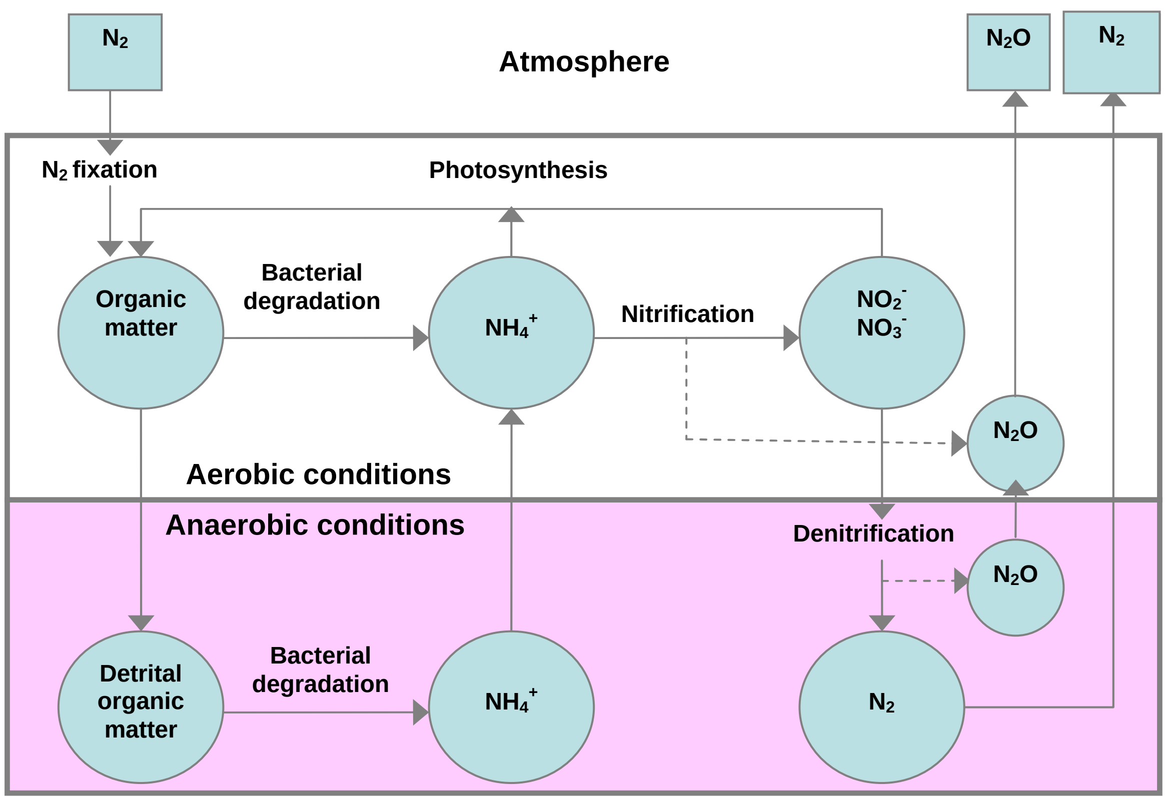

Figure 1.1 Microbial transformation in the nitrogen cycle. Figure is based on Wollast (1981) and drawn by Dan Sobata (used with permission).

Health concerns believed to be associated with drinking-water nitrate include methemoglobinemia (“blue-baby syndrome”), which occurs in infants and is associated with the consumption of water with nitrate concentrations greater than 10 mg/L NO_3-N (Ziebarth, 1991). Additionally, several forms of cancer, negative reproductive outcomes, and diabetes are thought to be associated with consumption of drinking-water nitrate (Weyer et al., 2001; Ward et al., 2005). However, based on a review of current epidemiological research, no definitive conclusions can be drawn regarding the health effects of nitrate on humans (Ward et al., 2005), and therefore nitrate can only be considered a potential health threat.

Based on early methemoglobinemia studies, the US EPA mandated a maximum contaminant level of 10 mg/L NO_3-N for municipal water supply systems (Ward et al., 2005), while no regulations exist for most rural drinking water systems. As 96 % of self-supplied drinking water systems in the United States rely on groundwater (Nolan et al., 1997), high groundwater nitrate concentrations are a cause of concern as many private wells are not monitored frequently for water quality. Additionally, as it is required by law that municipal wells provide drinking water that meets public health standards, local municipalities have a critical interest in the aquifer quality of surrounding regions.

1.2. Study Area

The Southern Willamette Valley (SWV) is an agrarian region located between the Cascade and Coast Range Mountains of Oregon. Major cities include Albany, Corvallis, and Eugene (see Figure 1.2). Outside of urban areas, land use in the region is dominated by coniferous forests in mountainous regions and by agriculture in the valley lowlands. Major crops of the region include grass seed, winter wheat, hay, peppermint, corn, hazelnuts, and assorted orchard crops. Higher intensity crops which require greater N and irrigation applications are generally grown within 5 km of the Willamette River, where coarser, more well-drained floodplain soils are located.

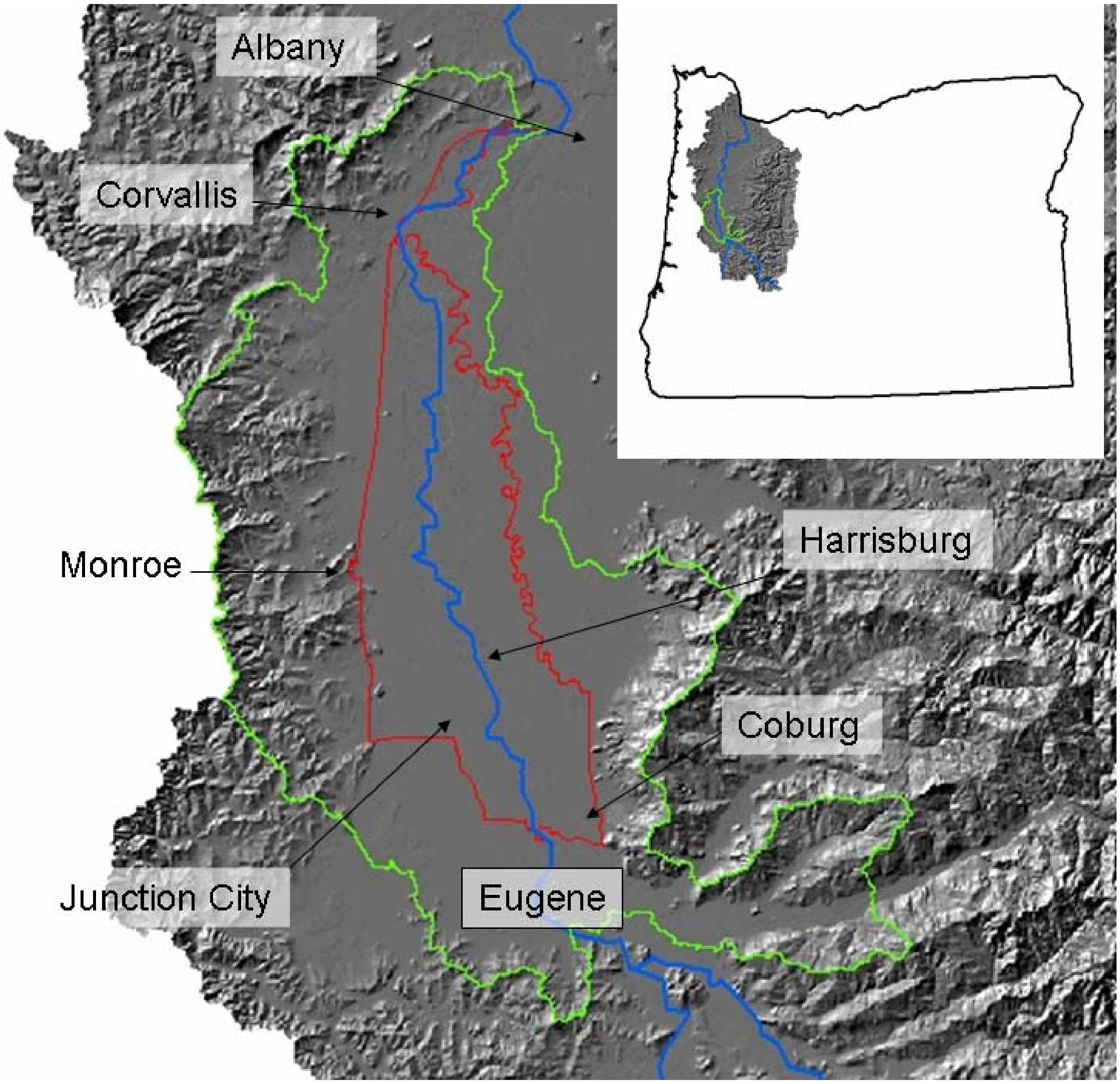

Figure 1.2. The Southern Willamette Valley of Oregon, with communities. The Groundwater Management Area is outlined in red, while the green outline is the study area modeled in Chapter 3. The Willamette River is shown in blue.

The hydrogeology of the SWV is dominated by the Willamette aquifer, a surficial aquifer composed of alluvial sediments originating from the Cascades and deposited during previous glacial maximums and by reworked sediments of the Willamette River (Gannet and Caldwell, 1998; O'Connor et al., 2001). The Willamette silt, a fine-grained glaciolacustrine unit deposited after the last glacial maximum, overlies much of the Willamette aquifer in the SWV and acts as a semi-confining unit on the Willamette aquifer. However, the Willamette silt is not present in the floodplain of the Willamette River, making the aquifer unconfined in the floodplain region.

In the past decade, water quality analyses from the SWV have shown a trend that indicates that nitrate contamination is an increasing problem. In a US Geological Survey study from 1993-1995, nitrate contamination was present in the SWV and 13% of all samples exceeded the EPA's maximum contaminant level of 10 mg/L NO_3-N (Hinkle, 1997). A study of groundwater in the Junction City and Coburg areas, done in 1993 and 1994 by the Oregon Department of Environmental Quality (DEQ), found 40% of the wells sampled exceeded 10 mg/L NO_3-N (Aitken et al., 2003). In 2003, two reports released by the DEQ (Aitken et al., 2003; Eldridge, 2003) revealed the spatial extent of the high nitrate levels within the SWV. Further work by Vick (2004) found statistically significant differences in groundwater nitrate concentrations for areas with different surficial geologic units, further supporting observations made by DEQ that areas where the Willamette aquifer is unconfined typically have higher nitrate contamination, as shown in Figure 1.3. Increased nitrate concentrations have been linked to increased fertilizer application along with a decline in the use of cover crops (Burket et. al, 2003) and the harvest of crops with high N and/or irrigation requirements, such as peppermint and vegetable crops (Feaga et. al, 2004). Septic leachate has also been found to significantly influence groundwater nitrate concentrations in the Coburg and Junction City areas, as indicated by isotopic data from Vick (2004). Groundwater age-dating and a flow model for the SWV indicate found that groundwater ages are approximately 18 years in the unconfined Willamette aquifer near Coburg and approximately 38 years under the Willamette silt near Harrisburg (Craner, 2006).

Due to concerns mentioned above, the DEQ declared the SWV a Groundwater Management Area (GWMA) in 2004. As a GWMA, a committee of citizen-stakeholders is required to advise and help various state agencies formulate an action plan for reducing existing contamination and to minimize future contamination. Additionally, the GWMA mandate requires that groundwater monitoring be carried out to determine when groundwater nitrate levels start to decline, and ultimately when regional groundwater nitrate concentrations drop below 7 mg/L NO_3-N (which is when the GWMA can be rescinded).

1.3. Objectives

There are two objectives of this thesis: 1) collect baseline temporal groundwater nitrate data to determine if seasonality exists and how seasonality may impact the sample frequency and design of a groundwater nitrate monitoring network; and 2) predict the likely impacts of GW-BMPs and future land use change on nitrate leaching in the SWV. Through statistical analyses of temporal well data and interpretations of modeled data, groundwater monitoring objectives and sample design can be refined and potential GW-BMP outcomes can be examined.

The proceeding major chapters (Chapters 2 and 3) have been written as stand-alone papers and may be submitted for publication.

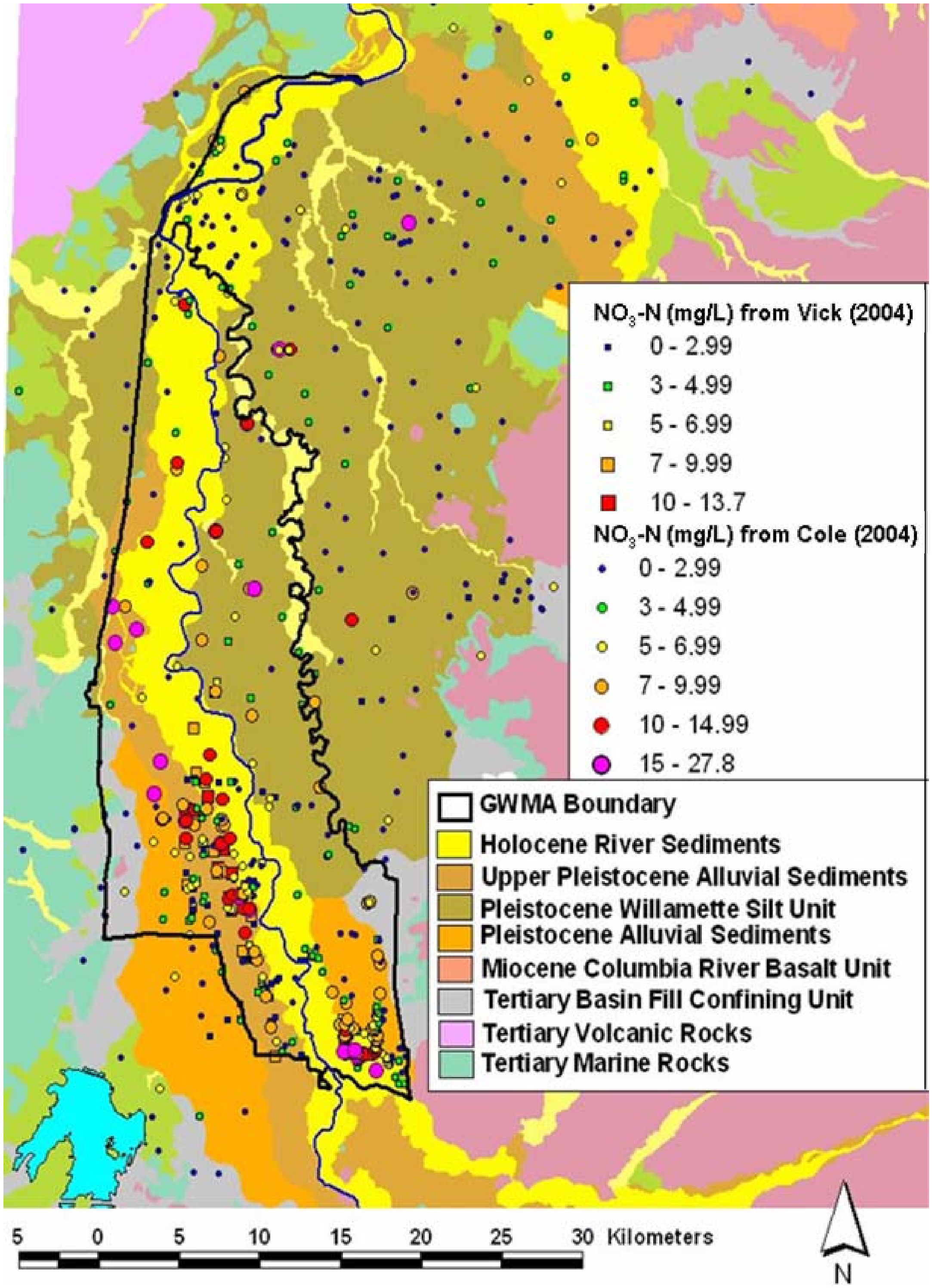

Figure 1.3. Spatial distribution of groundwater nitrate from previous studies and regional geology as mapped by O'Connor et al. (2001).

2. Examining temporal variability in groundwater nitrate in the Southern Willamette Valley, Oregon: Implications for detecting long-term change

2.1. Abstract

Groundwater nitrate contamination is an issue facing many parts of the United States and the world. Although nitrate is monitored as a public health concern, temporal variations can make it difficult to determine if an aquifer's contamination exceeds a threshold and if the contamination is increasing or decreasing over time. This study examines temporal variability in groundwater nitrate and its implications for the setup and design of groundwater nitrate monitoring networks. A well network was established in the Southern Willamette Valley of Oregon for 15 months and analyzed for seasonal trends. Large concentration fluctuations were observed, with most sample sites appearing to respond to recharge. However, we found no statistically significant seasonal differences for the network population. We believe spatial heterogeneities in land use, vadose zone properties, and aquifer characteristics cause a dampened network-wide seasonal response, resulting in non-significant seasonal variation at the network level. Well concentration and variability differed significantly depending upon hydrogeologic unit and overlying land use, with areas of high concentration and variability typically in regions of higher agricultural intensity. Findings indicate that with a sufficiently large network (in this case, 19 wells) of spatially distributed wells, less frequent sampling (quarterly) should be sufficient to detect long-term trends in regional nitrate concentrations.

2.2. Introduction

Nitrate is the most commonly detected groundwater contaminant in the United States (Nolan and Stoner, 2000). Nitrate fertilizer use has steadily increased since the 1950s (Holland et al., 2005). As groundwater nitrate is largely present in rural regions, it can be viewed as a byproduct of rural living, due to septic leachate, agricultural fertilization, lawn and garden fertilization, and animal feeding operations. Additionally, as 96% of rural residents depend on groundwater as their source of drinking water (Nolan and Stoner, 2000), it is considered a health concern. Nitrate pollution generally is derived from non-point sources, and is therefore a difficult contaminant to control. Since many locales are now monitoring or considering monitoring groundwater nitrate, understanding its temporal variability is important for regulation, remedial efforts, and real estate transactions.

Several studies have examined seasonal or monthly variability in groundwater nitrate, identifying different forcing mechanisms for variation in different areas. Studies have found recharge water to act as a diluting agent (Pacheco et al., 2000; Wilcox 2003), a concentrating agent (Anderson, 1993; Williams et al., 1998), or both depending on time of year, fertilization dates, and sample location (Katz and Böhlke, 2000; Landon et al., 2000; Harter et al., 2002; Mitchell et al., 2005). All studies note that high monthly or seasonal variabilities exist (wells changing either more than 5 mg/L NO_3-N in the course of a year or varying by greater than 50% of their concentration annually).

Though numerous studies have identified significant monthly variability in groundwater nitrate (Harter et al., 2002; Pacheco et al., 2000; Rajmohan and Elango, 2005; Katz and Böhlke, 2000; Mitchell et al., 2005), most groundwater nitrate monitoring networks sample frequencies range from quarterly to annually (Maila et al., 2004; Williams et al., 1998; Anderson, 1993; Stogner, 1997; Kelly and Ray, 1999; Richerson, 2003). As EPA guidance only applies to public water systems (which requires sampling for groundwater nitrate quarterly if concentrations are above 5 mg/ L NO_3-N, otherwise annually (EPA, 2004)), most monitoring programs determine monitoring frequency based on a "best judgment" approach or fiscal constraints. An unresolved and overlooked issue is whether or not lower frequency monitoring (such as quarterly) sufficiently captures network-wide seasonal variability, or if more frequent monitoring is necessary to adequately address network-wide seasonal variability given high intra-well variabilities.

The Southern Willamette Valley (SWV) of Oregon has high groundwater nitrate concentrations (Hinkle, 1997; Eldridge, 2003; Aitken et al., 2003; Vick, 2004), which are a cause of concern to the local community. In 2004, the region was designated a

Groundwater Management Area (GWMA) by the Oregon Department of Environmental Quality due to a large number of wells testing above 70 % of the US-EPA's 10 mg/L maximum contaminant level for drinking water. Future action plans for the GWMA include drilling monitoring wells and designing a groundwater nitrate monitoring network.

Aside from research focusing on groundwater nitrate, studies of nitrate in the Willamette Valley have addressed leaching from the vadose zone (Feaga et al., 2004; Shelby 1995; Brandi-Dohrn et al., 1997, Young et al., 2000), tile drains (Warren, 2002), surface water (Floyd, 2005; Rinella and Janet, 1998), mineralization (Whalen et al., 2000), and denitrification potential (Iverson, 2002; Arighi, 2004, Rich and Myrold, 2004; Well et al., 2000). Notably, Feaga et al. (2004) observed a seasonal flushing of vadose nitrate, where soil water with high concentrations of nitrate is flushed out of the shallow vadose zone at the onset of winter rains. Specifically, under crops with higher nitrogen (N) demand in the SWV, summer soil water concentrations are found to be high (> 30 mg/L NO_3-N) due to fertilization and minimal dilution. Fall and winter rains then dilute and move much of the nitrate mass past the vadose sampling sites.

We hypothesize that flushing of soil leachate will impact shallow groundwater nitrate concentrations and thus cause a seasonal signature in groundwater nitrate values. We shall examine this as our primary hypothesis, examining fluctuations both qualitatively and statistically. Additionally, we will investigate NO_3-N concentration differences between recharge and non-recharge months and in separate hydrogeologic units. The primary objective of this research is to determine if seasonal nitrate variability is present in SWV wells, and if so, to assess the likely implications of intra-well temporal variability on long-term nitrate monitoring trends.

2.3. Methods

2.3.1. Site Information:

The SWV of Oregon is a structural basin with the Cascade Range to the east and the Coast Range to the west. The area of focus for this study lies between Corvallis and Eugene (refer to Figure 2.1). The hydrogeology of the SWV is characterized by a

Basement Confining unit composed of Tertiary marine sedimentary rocks, volcanic rocks from the Coast Range, and volcanics from the Western Cascades. Above this lies the Willamette Confining unit composed of reduced clays with minor sand lenses (Gannett and Caldwell, 1998). The overlying Willamette aquifer, the primary groundwater source, is composed of several large alluvial fans of sand and gravel deposited by Cacade streams after Pleistocene glaciation. Within the Willamette River floodplain, the Willamette aquifer also includes Holocene alluvial sediments deposited by the Willamette River. Overlying the Willamette aquifer is the locally-present Willamette silt unit, composed of alluvium and fine-grained outburst flood deposits associated with the Glacial Lake Missoula floods. During the last glacial maximum, the lowlands of the Willamette Valley were frequently inundated by jökulhlaups, which formed Glacial Lake Allison and re-deposited fine-grained glacial deposits in the lowlands (Gannett and Caldwell, 1998). In our study area, the Willamette silt is incised by the Willamette River, causing the Willamette aquifer to be unconfined in the corridor along the Willamette River. Soils in the study region are primarily silt loams and silty clay loams, with generally more well-drained soils (including some loams, fine sandy loams, and gravelly sandy loams) in areas where the Willamette silt is not present (Knezevich, 1975; Patching 1987). Major soil groups include Datyon (Vertic Albaqualfs), Malabon (Pachic Ultic Agrixerolls), Bashaw (Xeric Endoaquerts), Coburg (Pachic Ultic Argixerolls), Woodburn (Aquultic Argixerolls), Newburg (Fluventic Halploxerolls), Chehalis (Cumulic Ultic Haploxerolls), Amity (Argiaquic Xeric Argiabolls), and Cloquato (Cumulic Ultic Haploxerolls) (Knezevich, 1975; Patching 1987).

Climate in the SWV is Mediterranean, characterized by cool wet winters and warm dry summers. Mean annual precipitation for the low-relief SWV is 1109 mm, with approximately 80% of the annual precipitation falling between October and March. Monthly temperatures on average range from 4.5^{o} C in December to 19.4^{o} C in August (OCS, 2006). Water year 2005, when most of this study's data were collected, was drier than the mean by 382 mm and was abnormal in that only 72 mm of rain fell from January 1 to March 15.

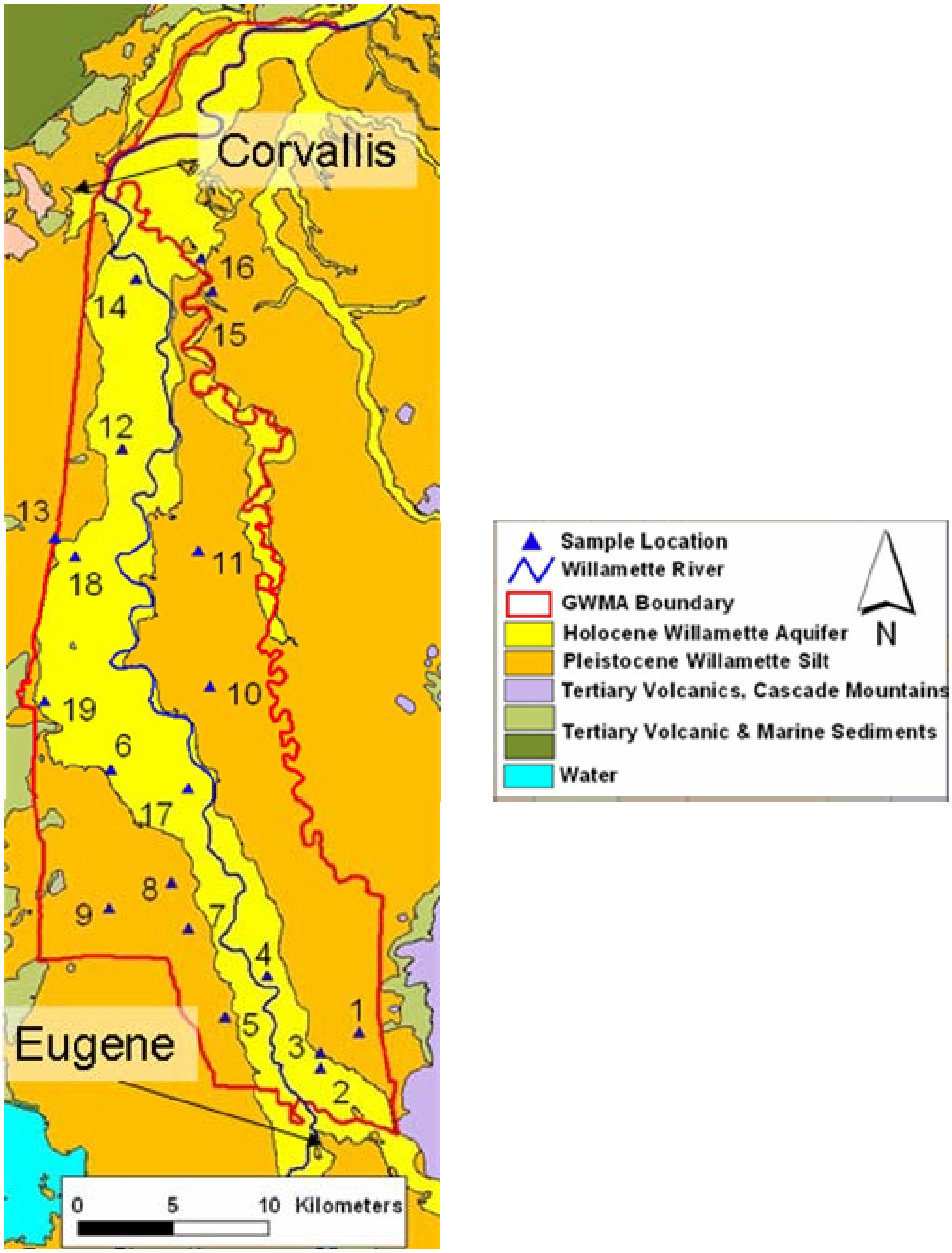

Figure 2.1. Generalized geologic map of the Southern Willamette Valley (adapted from Gannett and Caldwell (1998)). Major hydrogeologic units include the Willamette aquifer (Holocene alluvial deposits) and the Willamette silt (Pleistocene flood sediments and alluvium). Though more detailed geologic and hydrogeologic maps are available for the area (O'Connor et al., 2001; Conlon et al., 2005), the generalized map is used in this study due to statistical sample size concerns. Exact well locations and IDs given are found in Appendix A.

Land use in the lowland SWV is predominantly agricultural, with major crops being grass seed, hay, wheat, filberts, sweet corn, and peppermint. Assorted berries, fruits, and row crops are also grown. Land use in the area overlying the unconfined Willamette aquifer has historically included row crops and peppermint, both of which are relatively high-intensity crops requiring irrigation and high N inputs. However, in the late 1990s, a market shift caused many row crop and peppermint growers to move to grass seed production, which is less intensive with respect to nutrient and irrigation demands. Areas underlain by the Willamette silt generally harvest grass seed and hay. Additional land use practices believed to significantly increase N inputs in the GWMA include septic drainage from rural residences and several confined animal feeding operations (CAFOs). High density septics are believed to be a major cause of contamination for several locales in the GWMA, while CAFOs are believed to have a more localized effect.

2.3.2. Sampling Methods

To determine if seasonal variability in groundwater nitrate exists for the SWV, we sampled from 19 wells at an approximately monthly interval for 15 months (August 2004 through October 2005). Domestic wells accounted for 15 of the 19 wells sampled, while 4 were shallow (<30 ft or 9.14 m) monitoring wells. Sample locations are shown in Figure 2.1. To ensure that the data were most likely to capture seasonal trends, wells chosen for sampling were all 50 ft (15.24 m) or shallower in depth (to ensure that surface impacts would likely be discernable), have screening intervals of 15 ft (4.57 m) or less (to minimize mixing and dilution of shallow groundwater), were drilled within the last 30 yrs (to ensure minimal well deterioration), have an extant well log (to determine well depth, screening interval, and aquifer material), and had no coliform bacteria present at the initiation of sampling (for ascertaining that no conduit allows surface water to enter the well). Given that it would be difficult to characterize a 548 km^2 sample area with just 19 wells, we chose sample regions in an attempt to be as representative as possible with regard to surficial geology, land use, and expected nitrate concentrations (based on previous studies). After specific sample regions were identified, sample sites were determined based on the permission of landowners and the ability of the well to pass the aforementioned quality controls.

Field protocols used for domestic wells for the majority of sampling months were as follows. Wells were purged for a minimum of 15 minutes, with total purge time determined by the stabilization of field parameters (pH, dissolved oxygen (DO), temperature, and specific conductivity). Field parameters were recorded at 3 minute intervals, and sampling occurred after all parameters were stable for 3 consecutive intervals (with stability parameters determined from Koterba et al., 1995). In cases where a field parameter would not stabilize, purging continued until at least 3-4 casing volumes were purged. During initial sampling months (August – October 2004), wells were purged for at least 5 minutes, with samples taken several minutes later when field parameters were considered stable. Prior to December 2004, field parameters used for stability criteria were as follows: temperature and total dissolved solids in August, DO and temperature in September, DO, temperature, and specific conductivity in October, and DO and temperature in November. Analyses of well water collected after 6.5 and 9 minutes compared to those collected between 18 and 15 minutes found that nitrate differences associated with purge times were minimal.

Sample protocols used for monitoring wells included purging 4-5 casing volumes with a peristaltic pump. The pump intake depth at each well was held constant across sampling events. Field parameters were recorded at the time the sample was taken.

Samples were collected in acid-washed bottles and stored on ice throughout the sampling day. At the time of sampling, all bottles were rinsed at least 3 times with fresh purge water before taking the sample. After sample collection, samples were frozen until analysis (for 1-16 days). Samples were analyzed for NO_3-N using the cadmium-reduction method on an Alpkem Flow Solution, a continous-flow autoanalyzer with digital and monochromater detectors.

Sample duplicates, spikes, and blanks were submitted in addition to study samples, and accounted for 10% of all samples run. Methods for field duplicate collection included taking two successive samples from a running tap, or by taking one large sample and splitting it into smaller sample bottles. Spikes were created following the spike protocol 4500-NO_3^- B from Eaton et al. (1995), with concentrations varying so that duplicates were obtained for the entire sample concentration range. Blank preparation was done by rinsing sample containers 3 times with deionized (DI) water before sampling the DI water.

2.3.3. Statistical Methods

All statistical tests employed in data analyses were nonparametric, which do not assume that a data set is normally distributed. We analyzed the data using nonparametric statistics because the sample populations could not be transformed to have normal distributions, had numerous outliers present, and were generally heteroscedastic in nature. Helsel and Hirsch (2002) note that most water resource data sets do not meet the assumptions required in parametric statistical tests. In the following discussion of statistics, seasonality is defined as a network-wide statistically significant difference in groundwater nitrate concentrations for a specific time period. Time periods examined include monthly data for the duration of the study and lumped monthly periods when recharge occurred. The time period used in examining seasonality is specified in each paragraph.

The first suite of hypothesis tests were performed to determine the seasonality of groundwater NO_3-N, and include the Kruskal-Wallace test, the rank-sum test, and the Moses test. The Kruskal-Wallis test, similar to ANOVA, compares several independent groups to determine if their central values (mean for ANOVA, median for Kruskal-Wallis) differ. More generally, the Kruskal-Wallis test can be used to compare if different groups of data have identical distribution shapes. Thus, the Kruskal-Wallis was applied to determine if seasonality was present in the data set, which would be the case if significant differences were found between monthly NO_3-N concentrations. The Kruskal-Wallis test statistic is computed by ranking all data from a population and then comparing the average monthly population ranks to the entire population's average rank. Further information on the test statistic can be obtained in Helsel and Hirsch (2002).

The rank-sum test (also known as the Mann-Whitney test or the Wilcoxon rank-sum test) is similar to the parametric t-test and compares two independent sets of data to see if one data set generally contains larger values than the other. Similar to the Kruskal-Wallis, the entire population is ranked and then the rankings are summed for the subset populations. For examining seasonality, the rank-sum test was used to compare recharge and non-recharge month NO_3-N concentrations.

The final statistical test used for examining seasonal fluctuations was the Moses test, a nonparametric test used to compare differences in variability. It is similar to the parametric F-test. The Moses test statistic is obtained by calculating the average values of randomly grouped data subsets from two populations, summing the squared differences between the values in a group and the group's mean, and then ranking and summing the ranks of the squared differences for each population. The Moses test was applied to test if recharge and non-recharge months have differences in variability. More information on the Moses test is available in Sheskin (2004).

Other test statistics employed were Spearman's rho and Kendall's coefficient of concordance. Both measure the strength of association between variables, and were used in this study to compare groundwater nitrate and monthly precipitation values during periods of recharge. Spearman's rho is used to compare two variables, while Kendall's coefficient of concordance can be used to compare multiple variables. The rho statistic involves ranking the two separate variables independently and then for each pair multiplying the ranks of their variables together (Helsel and Hirsch, 2002). Kendall's coefficient of concordance is calculated similarly to rho, except it compares multiple ranks of variables to see if the rankings are consistent across all populations (i.e. dependent) (Daniel, 1990). Kendall's coefficient of concordance was used in this study to examine intra-site NO_3-N trends for monthly populations during periods of recharge. Rho examined monthly median NO_3-N concentrations against precipitation for recharge months. For both test statistics, a value of +1 indicates perfect positive correlation, -1 indicates a perfect negative correlation, and 0 indicates no correlation.

The rank-sum and Moses test were also employed to examine differences in nitrate concentration and variability between the Willamette silt and Willamette aquifer hydrogeologic units. Additionally, the Spearman's rho and the Kruskal-Wallis test were used to determine if well variability increases with concentration. In the rho test examining dependence between variability and concentration, variability was calculated as part of the nonparametric coefficient of variation. The nonparametric coefficient of variation is similar to the coefficient of variation because they both compare the central 38.3% of a frequency distribution with the distribution's central location. In the case of the coefficient of variation, this is expressed as the population's standard deviation divided by the mean; the nonparametric coefficient of variation is the difference between the population's 69.15 and the 30.85 percentiles divided by the median (Prager and Mohr, 1999).

2.4. Results

Data values are in Table 2.1, while Figures 2.2 and 2.3 examine month to month and intrasite sample distributions via boxplots. In Figure 2.2, interquartile ranges (the range between the 75th and the 25th percentiles) appear to be greater in several high precipitation months, leading to an examination of the variabilities for different time periods as shown in Table 2.2. In Figure 2.2, the outlier/extreme outlier values are associated with one well (well 6), which is 500m down gradient from a confined animal feeding operation. The median concentration of all groundwater nitrate values is 5.6 mg/L NO_3-N, with 39.9% of values above 7 mg/L and 18.0% above 10 mg/L NO_3-N.

All hypothesis and statistical tests performed, along with their results, are listed in Table 2.2. Major findings from Table 2.2 include 1) Seasonality is not manifested by monthly NO_3-N population data. 2) The difference in median values for recharge and non-recharge periods has a low level of significance (two-tailed p-value = 0.30). 3) There is no significant difference in variability between recharge and non-recharge periods. 4) Precipitation and median groundwater nitrate concentrations for recharge months show a strong, but insignificant correlation. 5) The precipitation-nitrate correlation is supported significantly when the entire well population is examined. 6) The Willamette silt and the Willamette aquifer have significantly different medians and variabilities. 7) Wells with higher concentrations are correlated with higher variabilities.

Though seasonality was not found to be statistically significant, substantial seasonal fluctuations were observed in numerous wells. Figure 2.4 displays nitrate concentrations in time with precipitation between sampling events. Though not all wells have as strong seasonality as wells 16 and 19, 15 of the 19 wells appear to be influenced by seasonal precipitation (see Appendix A). Figure 2.5 shows a subsample of wells that exhibit a seasonal relationship with precipitation, but are out of phase with respect to one another. Error bars applied in Figures 2.4b and 2.5 represent a 95% confidence interval of +/- 7.4%, based on 25 duplicates having a mean difference of 6.8% and a standard deviation of 18.9% (see Appendix B). A 0.0% duplicate difference would indicate that the values of a duplicate pair agree exactly, and the presence of several duplicates with large differences (between -22 and 68%) caused considerable uncertainty.

The impact of precipitation between sample events and average monthly groundwater levels is shown in Figure 2.6, with each month's groundwater depths being the average depth observed in four monitoring wells (sites 3, 6, 12, and 16). Recharge months used in all analyses (Figure 2.7 and Table 2.2) are defined as months where groundwater levels increase. No relationship between monthly water level and groundwater nitrate concentrations was observed.

A trend with median groundwater nitrate values increasing with greater precipitation during recharge months is shown in Figure 2.7. Though the Spearman's rho value for the trend is strong, significance is compromised by the low number of recharge months observed in this study. Further investigation of this trend using Kendall's coefficient of concordance found a statistically significant correlation between monthly nitrate concentrations and precipitation values during recharge (see Table 2.2).

Figure 2.8 indicates substantial inter-site variability is present in our data, and notably the data appear to have a greater range of values for wells with higher median NO_3-N concentrations. Using the nonparametric equivalent of a standard deviation (the central 38.3% of the population distribution) to represent variability, Figure 2.8 shows a correlation between median well concentration and well variability, with higher concentration wells generally having higher magnitudes of variability. Kruskal-Wallis tests of log-transformed normalized and non-normalized well concentrations (Table 2.2) also indicate that higher concentration wells generally have greater nitrate fluctuations.

Table 2.1. Monthly NO3-N concentrations (mg/L) by sample site. Hyphens are used for months when samples could not be obtained. Variabilities listed are the nonparametric equivalent of the standard deviation, which is the spread between the 30.85th percentile and the 69.15th percentile of a given population. The Range/Median data indicate that spatial variability is much greater than temporal variability, while the Range and Range/Median data for the monthly median values indicate that monthly fluctuations of the median are significantly less than most other wells.

Site #	8/04	9/04	10/04	11/04	12/04	1/05	2/05	3/05	4/05	5/05	6/05	7/05	8/05	9/05	10/05	Well Median	Variability*	Range	Range/ Median
1	4.2	5.3	5.9	5.5	2.4	5.2	5.0	5.0	4.9	4.8	5.1	5.1	5.6	5.7	5.8	5.1	0.4	3.5	0.7
2	10.9	11.9	13.2	13.0	12.9	-	10.8	12.8	13.0	11.5	12.4	11.7	11.2	12.1	12.5	12.3	1.1	2.4	0.2
3	7.8	9.4	9.8	9.8	9.2	9.7	9.3	9.6	10.0	7.3	8.8	9.5	9.6	9.6	9.5	9.5	0.4	2.7	0.3
4	2.8	2.4	2.7	2.2	2.4	2.3	2.7	3.0	2.7	2.5	2.6	2.8	2.6	2.2	1.9	2.6	0.3	1.2	0.5
5	5.8	7.0	8.6	8.6	8.8	8.7	8.6	8.4	8.5	7.4	7.1	7.2	7.6	7.9	8.4	8.4	1.2	3.0	0.4
6	28.4	26.2	35.1	34.6	31.8	29.5	33.8	35.1	37.7	31.7	33.9	34.9	35.1	34.8	35.3	34.6	3.2	11.5	0.3
7	8.7	8.6	9.2	10.3	10.8	11.1	10.0	9.8	9.8	7.8	7.6	8.6	8.6	9.1	9.6	9.2	1.2	3.5	0.4
8	-	8.9	9.7	10.4	10.8	11.3	10.3	10.3	10.4	9.5	9.1	7.1	9.0	9.5	9.6	9.7	1.1	4.2	0.4
9	3.4	3.9	3.5	3.8	4.1	4.0	3.6	4.1	4.5	4.0	3.3	3.5	4.1	4.0	3.3	3.9	0.5	1.2	0.3
10	3.6	5.2	5.5	5.4	5.5	5.5	4.9	5.7	5.9	3.3	5.5	4.4	5.6	5.7	4.9	5.5	0.7	2.6	0.5
11	4.7	4.1	4.7	5.0	4.6	4.7	4.5	4.7	5.0	4.2	4.1	2.7	4.8	5.1	5.1	4.7	0.4	2.4	0.5
12	0.0	0.5	0.5	0.0	0.0	0.0	0.2	0.9	1.1	1.8	1.9	2.0	1.2	0.1	0.0	0.5	0.9	2.0	4.4
13	0.8	1.1	1.4	1.4	1.4	1.1	0.9	1.1	1.5	1.3	1.9	1.7	1.8	1.9	1.6	1.4	0.4	1.1	0.8
14	3.7	5.8	7.5	7.8	7.0	7.1	6.1	6.1	10.0	8.6	6.1	4.1	6.4	6.5	6.7	6.5	0.9	6.3	1.0
15	4.3	4.4	4.6	4.9	5.2	5.2	5.0	5.1	5.7	3.8	5.0	2.8	5.0	5.2	5.2	5.0	0.6	2.9	0.6
16	2.5	4.5	5.1	5.5	5.9	5.1	4.7	5.7	6.1	4.2	4.1	2.8	3.7	3.5	4.0	4.5	1.2	3.6	0.8
17	5.5	4.5	3.9	3.2	2.7	2.5	3.2	4.0	4.9	4.1	4.3	4.2	5.2	5.5	4.6	4.2	0.9	3.1	0.7
18	11.4	11.1	11.4	10.2	12.2	11.5	11.4	12.0	12.7	6.4	10.3	7.3	9.8	11.7	11.7	11.4	1.3	6.3	0.6
19	6.4	6.8	7.5	7.9	8.1	5.8	6.9	7.1	7.2	4.9	6.6	6.5	6.6	7.0	7.2	6.9	0.6	3.2	0.5
Monthly Median	4.5	5.3	5.9	5.5	5.9	5.4	5.0	5.7	6.1	4.8	5.5	4.4	5.6	5.7	5.8	5.5	0.6	1.8	0.3
Variability*	2.6	2.8	4.2	4.1	4.9	3.4	4.4	3.9	4.9	3.3	3.0	4.3	3.1	3.1	4.0				
Range	28.4	25.7	34.6	34.6	31.8	29.5	33.6	34.2	36.6	30.4	31.9	33.2	33.9	34.7	35.3				
Range/Median	6.3	4.9	5.9	6.3	5.4	5.5	6.7	6.0	6.0	6.3	5.8	7.6	6.0	6.1	6.1				

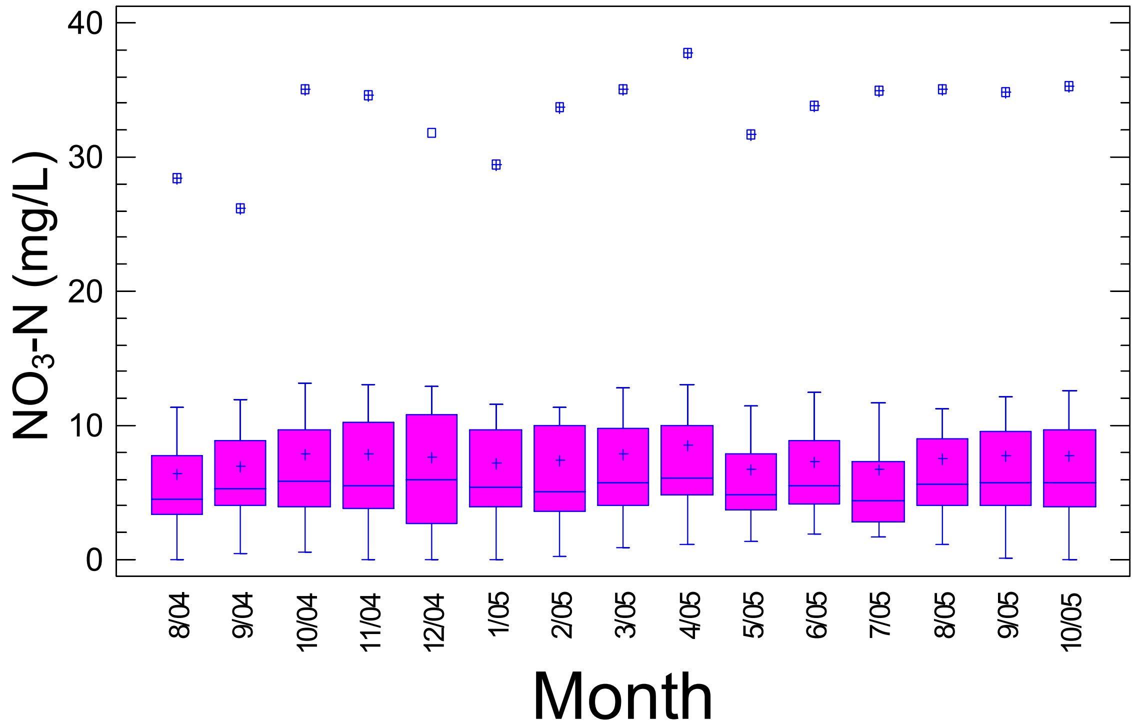

Figure 2.2. Monthly box and whisker plot for all groundwater nitrate data collected. "+" indicates the monthly mean values, the box bounds the 25th and 75th data percentiles, and the middle bar is the median. The interquartile range (IQR) is the difference between the 75th and 25th data percentiles. Whiskers extend to the farthest data point within 1.5 times the IQR. Outliers (data which range from 1.5 to 3 times greater than the IQR) are indicated by a small square, while extreme outliers (greater than 3 times the IQR) are squares with a cross inside. Extreme outliers and the outlier are all values from one well (well 6) across time. Well 6 is 500 m down gradient of a confined animal feeding operation.

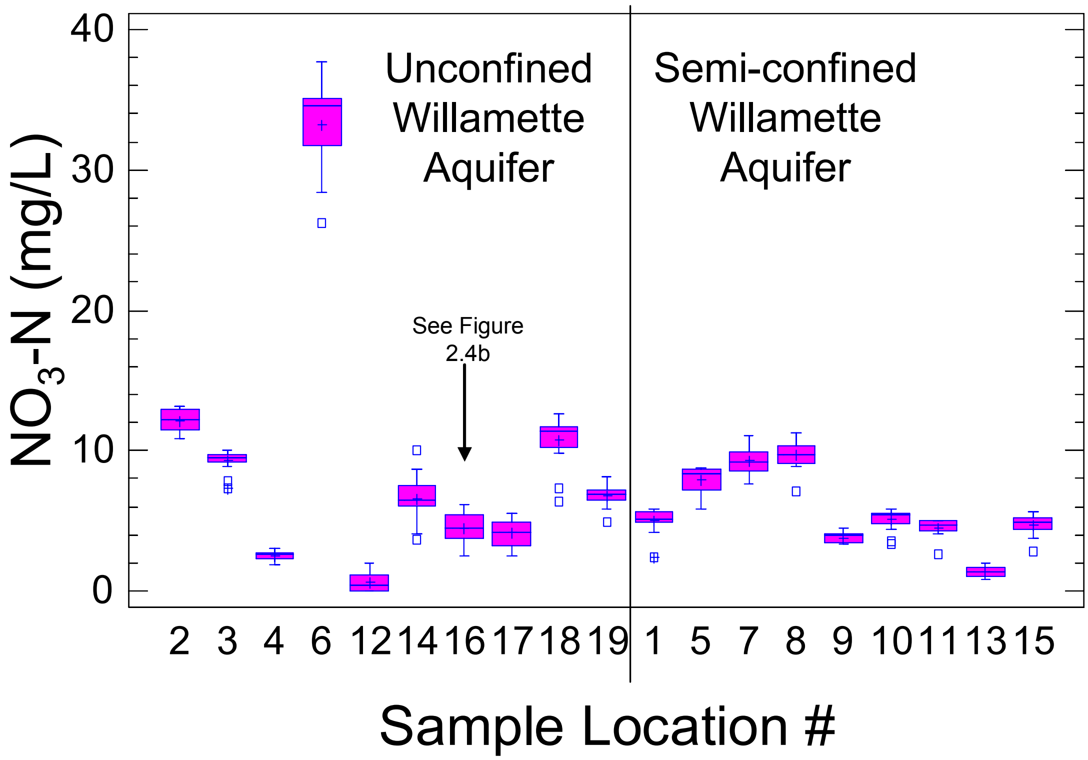

Figure 2.3. Box-and-whisker plot of groundwater nitrate for all wells. Sites to the left of the central line are wells in the Willamette aquifer, while wells to the right penetrate the Willamette silt to reach the Willamette aquifer. Wells 10, 11, and 15 puncture the fine-grained Missoula Flood Deposits (Qff_2) of O'Connor et al. (2001), which are generally finer than the Willamette silt unit applied in this study because they are composed of glaciolacustrine silts exclusively.

Table 2.2. Statistical tests, hypotheses examined, and results and significance. "Population" refers to the pool of NO_3-N concentrations associated with the given grouping. Seasonality was also qualitatively analyzed.

Concept Examined by Test Suite	Null Hypothesis	Test	Significance[+]
Seasonal Fluctuation in NO_3-N	Monthly population distributions are similar	Kruskal-Wallis	n. s.
	The population's median values in recharge and non-recharge months are similar	Rank-Sum	n. s. p = 0.30
	The population's variance between recharge and non-recharge months is similar	Moses Test	n. s.
Influence of Precipitation on NO_3-N during recharge	The monthly median nitrate concentrations are independent of precipitation during recharge months	Spearman's Rho	n. s. ρ =1, p = 0.5
	During recharge months, precipitation and groundwater nitrate values are independent	Kendall's Coefficient of Concordance	W = 0.189, α = 0.05
Hydrogeologic differences in NO_3-N	The Willamette silt and the Willamette aquifer have similar median values	Rank-Sum	p = 0.0068
	The variance between the Willamette silt and Willamette aquifer do not differ	Moses Test	p = 0.001
Variability Scaling with Concentration	Well variability* is independent of concentration	Spearman's Rho	ρ = 0.567, p = 0.02
	The sample distributions of all 19 wells do not differ (using log-transformed data)	Kruskal-Wallis	p = 0.005
	The normalized^ sample distributions of all 19 wells do not differ (using log-transformed data)	Kruskal-Wallis	n. s.

[+] All p and alpha values reported are for two-tailed tests. n.s. indicates "not significant"

* Variability used in these tests is the nonparametric equivalent of the standard deviation, which is the spread between the 30.85th percentile and the 69.15th percentile of a given population.

^ Normalization for each well's population was done by dividing every value by the population's median value.

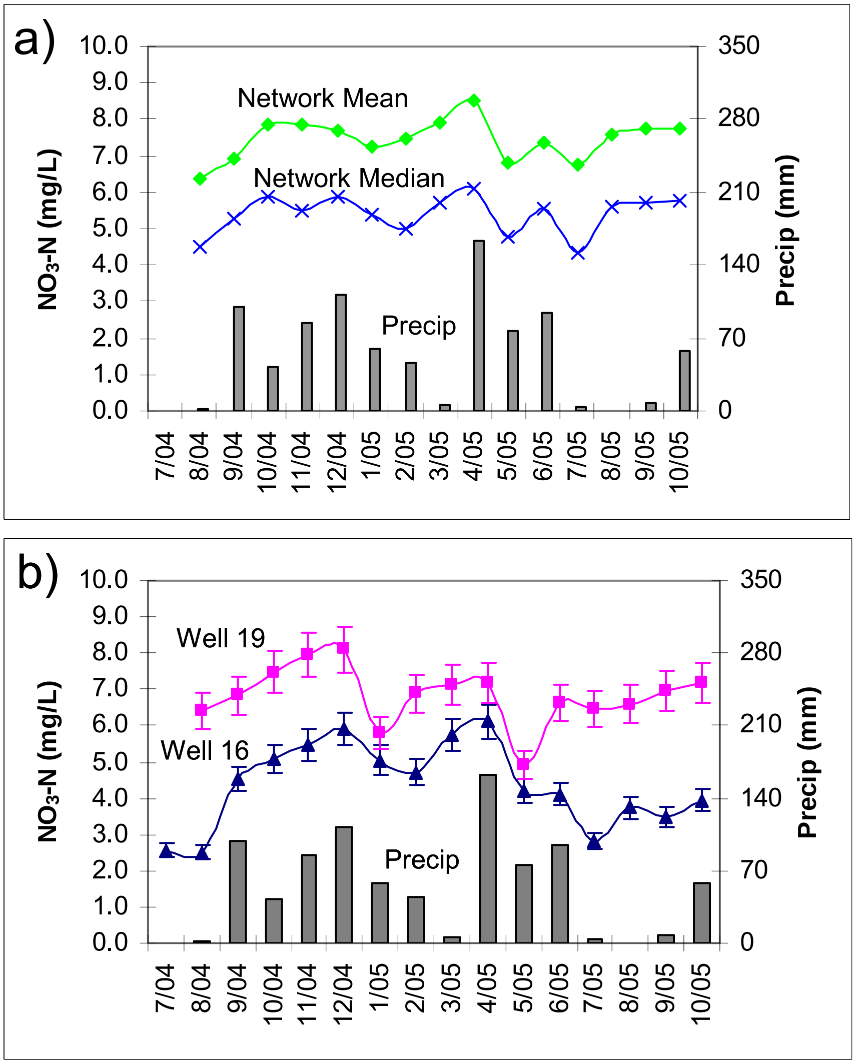

Figure 2.4. Precipitation between sampling events plotted against a) monthly mean and median groundwater nitrate concentrations and b) monthly values for wells 16 and 19. Error bars represent the 95% confidence interval for samples. Lines connecting sample points are for eye guidance and do not represent interpolated values.

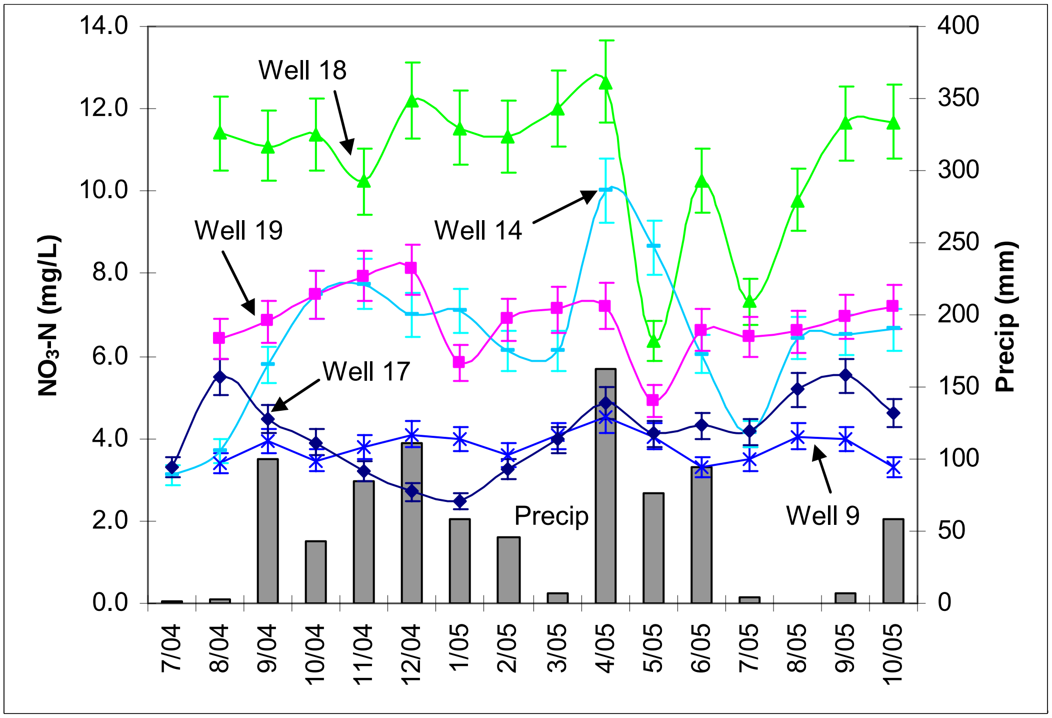

Figure 2.5. Though most wells exhibit seasonal variability, seasonality is not detected across the network. We attribute the out of phase relationship between wells, as shown above, to heterogeneities which effectively dampen network-wide seasonal responses. Error bars represent the 95% confidence interval for samples. Lines connecting sample points are for eye guidance and do not represent interpolated values.

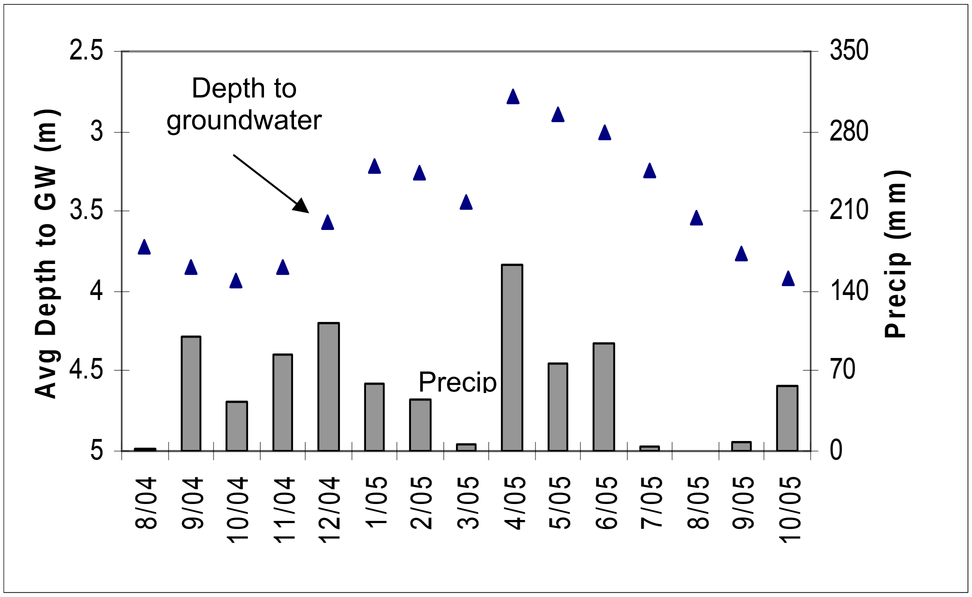

Figure 2.6. Precipitation and depth to groundwater vs. time. The groundwater depth is the monthly average depth between 4 monitoring wells where water level measurements were taken. Recharge months are defined based on this figure, with a recharge month being a month where the depth to groundwater decreased.

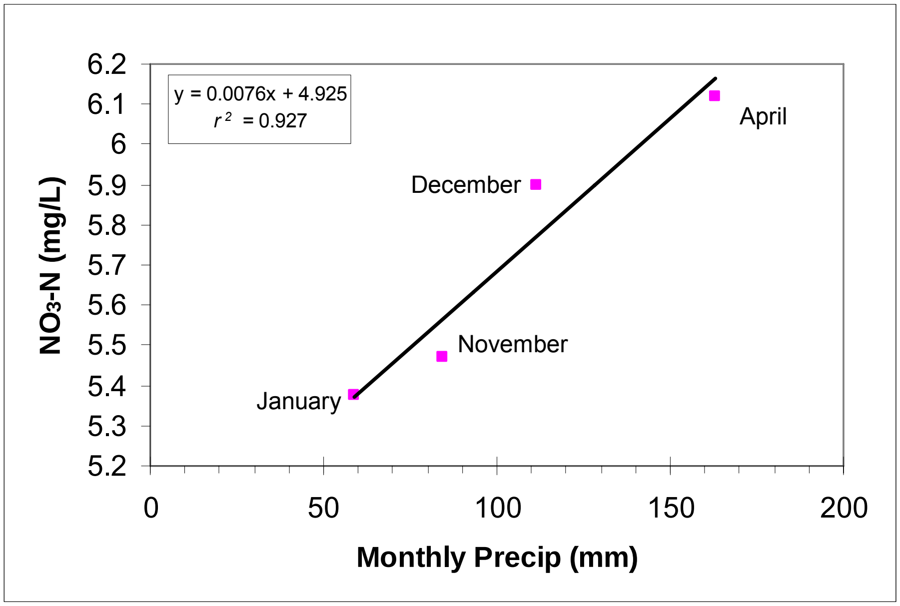

Figure 2.7. Median monthly groundwater nitrate values plotted against precipitation for recharge months. The Spearman's rho value for the association is 1, but is not significant.

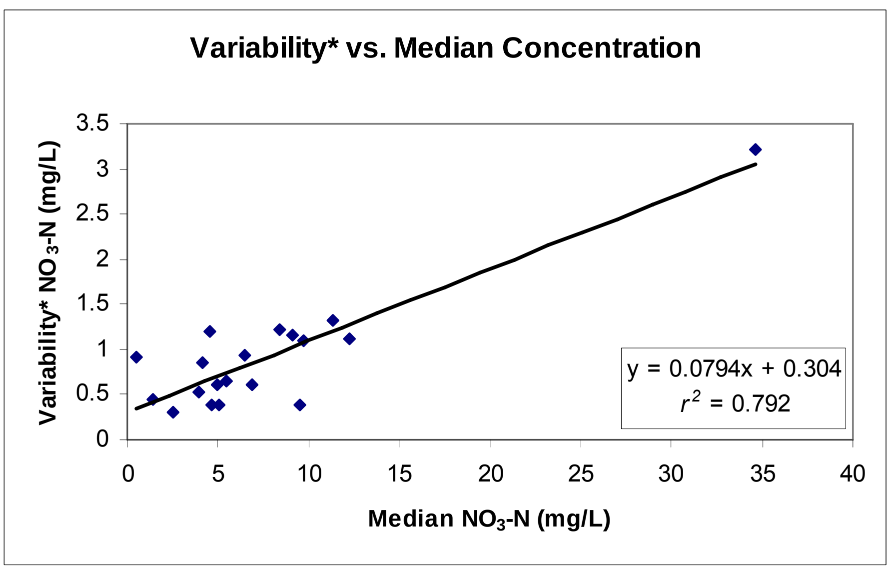

Figure 2.8. Site variability vs. median groundwater nitrate concentration. The Spearman's rho value of 0.567 was found to be significant at $p = 0.02$.

*The variability plotted is the nonparametric equivalent of the standard deviation, which is the spread between the 30.85th percentile and the 69.15th percentile of a given population. The slope of the regression line is therefore the nonparametric coefficient of variation.

2.5. Discussion

2.5.1. Comparison with Previous Data Sets

Data from this study are comparable to previously collected DEQ data (Aitken et al., 2003). DEQ data collected inside the GWMA from 2000-2001 had a median nitrate concentration of 5.7 mg/L NO_3-N , with 35.3% of wells having concentrations higher than 7 mg/L and 11.6% greater than 10 mg/L (based on a population of 258 wells). Data from this study had a median concentration of 5.6 mg/L NO_3-N, with 39.9% of samples higher than 7 mg/L in concentration and 18.0% higher than 10 mg/L. The similarity in data values indicates that data collected in this study are representative of nitrate concentrations in the GWMA

2.5.2. Seasonal Variability

Monthly groundwater nitrate distributions are not statistically different. Additionally, median and variance values are not statistically different between recharge and non-recharge months. Though the overall population distribution shows no significant seasonal differences, substantial intra-site variability exists, where individual wells, monthly median, and monthly mean values appear to be influenced by seasonal precipitation (Figure 2.4 and Appendix A). A visual assessment of the temporal data for all sites found discernable seasonality in 15 of the 19 wells (see Appendix A for individual well trends). Probable reasons why the majority of shallow wells observed show seasonal trends but population-wide seasonality does not exist likely include heterogeneities in the form of vadose zone characteristics, land use history, and aquifer properties. Combinations of these heterogeneities are hypothesized to affect the seasonal response of wells in different ways, causing the network population not to have seasonality since many of the wells exhibiting seasonality are out of phase with one another, as shown in Figure 2.5. Numerous studies have found the aforementioned factors to impact contaminant transport, including Landon et al. (2000), Zinn et al. (2004), Bedient et al. (1997), Mitchell et al. (2005), Williams et al. (1998), Harter et al (2002), Katz and Böhlke (2000), and Wilcox et al. (2005). The dampened network-wide seasonality is helpful from a monitoring network viewpoint because it indicates that local heterogeneities can make network-wide data less prone to seasonal bias.

2.5.2.1. Land Use history influencing seasonal response

Land use practices that have been found to impact the seasonal response include timing of fertilizer application and the intensity and frequency of irrigation (Landon et al., 2000). In a study of recharge rates using isotopic tracers, Landon et al. (2000) found that recharge derived from different times of the growing season had significantly different nitrate concentrations. Because soil water mixing was not pervasive in the study, specific surface management practices (such as pre- and post-fertilization irrigation) were observable in groundwater nitrate fluctuations. Though such data do not exist for the SWV, it is possible that recharge pulses entering the aquifer will not be well-mixed and will vary significantly in concentration for a given location. If this is the case, then the assumption that NO_3-N concentrations in the vadose zone are higher than groundwater NO_3-N concentrations would not always be correct, and hence a well's concentration could either increase or decrease due to recharge.

Irrigation timing and intensity may impact the seasonal response by making irrigated soils wetter than non-irrigated soils before the fall rains start. If a soil has more water in its profile prior to the rainy season, it could reach field capacity more quickly than a non-irrigated soil, leading to a quicker seasonal nitrate response for irrigated regions as the high concentration soil water will begin recharging the aquifer more quickly than in dry areas. Additionally, over-irrigation during the summer months for certain crops has been shown to cause summer leaching, which generally would not occur without anthropogenic control (Faega, 2003). This potentially could diminish annual fluctuations or minimize winter fluctuations since the soil water would be less concentrated in nitrate prior to the rainy months.

2.5.2.2. Vadose properties influencing seasonal response

Vadose zone properties that could cause different NO_3-N seasonal response times include the vadose zone thickness, degree of preferential flow, and transport parameters (saturated hydraulic conductivity and effective porosity) (Zinn et al., 2004; Bedient et al., 1997; Mitchell et al., 2005; Landon et al., 2000). A soil's unsaturated thickness generally determines the total amount of time and water needed for recharge to occur, as the vadose zone will largely act as a reservoir until ample precipitation has fallen for recharge to commence. As the depth to groundwater will vary spatially (Craner 2006; Mitchell et al., 2005; Landon et al., 2000), one should expect the exact timing of recharge to vary. Since samples collected for this study are temporal snapshots of aquifer nitrate concentrations, separate wells sampled on the same date could show different trends because their unsaturated zones are in different stages of wetting or drying.

The degree of preferential flow could likely affect a well's NO_3-N response to seasonality because it effectively makes the depth to groundwater less, which hastens the wetting up time of deep soils and causes recharge to occur sooner than otherwise (Landon et al., 2000; Selker et al., 1997). Additionally, in areas where matrix flow is dominant, basic transport parameters such as the effective porosity and the saturated hydraulic conductivity will be important controls on the rate of water and nitrate movement within soils (Selker et al., 1997, Bedient et al., 1997). Therefore, transport parameters should be expected to cause different recharge response times for different wells. Tracer test data reported by Feaga (2003) from 26 lysimeters in the Willamette Valley found substantially different recharge rates, with breakthrough times varying by a factor of 2.

2.5.2.3. Aquifer properties influencing seasonal response

Aquifer flow rate would be expected to influence the seasonal NO_3-N response in a well because it would directly influence shallow aquifer mixing and dilution rates. Aquifer preferential flow pathways could also affect seasonal NO_3-N concentrations for a well because the presence of a highly permeable unit could also influence dilution rates. In the Willamette aquifer, zones of preferential flow along thalweg gravel deposits have been noted in dewatered gravel pits, where seepage mostly enters the pit from a few well-defined zones (O'Connor et al., 2001). Though these regions of high flow are believed to even out over a scale of several hundred meters (O'Connor et al., 2001), their local impact on wells and nitrate transport are thought to be significant.

Additionally, localized aquifer properties are likely to impact groundwater nitrate seasonality trends for specific wells. Of the 4 wells with no seasonal response discernable for nitrate concentrations, 3 of the wells (1, 12, and 13) are suspected to be influenced by denitrification as all 3 had monthly mean dissolved oxygen values below 1 mg/L. Wells with nitrate present are generally sensitive to denitrification when dissolved oxygen values are lower than 1 mg/L (MPCA, 1999). For further information on dissolved oxygen values, refer to Appendix C.

Well 4, the remaining non-seasonal well, is hypothesized to be affected by hyporheic flow associated with the Willamette River. As the well is only 0.55 km from the river, it may be affected by local reversals in groundwater flow direction caused by high flows in the Willamette River. Flow reversals observed downstream in the Willamette aquifer at similar distances from the river (Hinkle et al., 2001) indicate that this is a possible mechanism that could impact groundwater nitrate concentrations.

As a result of the multiple heterogeneities found within the Willamette aquifer system, it is probable that most wells will show seasonal fluctuations and possible that most wells with no discernable seasonality will have local explanations. Due to the large spatial heterogeneity of the Willamette aquifer and most aquifer systems, muted seasonality trends when examining a large population should be expected.

2.5.3. Influence of Precipitation on Groundwater Nitrate During Recharge

A tentative trend can be identified when population medians are examined for recharge months, where increasing precipitation correlates to increases in groundwater nitrate values (Figure 2.7). A ρ value of one, along with a high r^2 value, indicates that a trend could be present, but due to small sample size, ρ is not significant. However, testing the association between volumetrically successive recharge months with their nitrate values (via Kendall's coefficient of concordance) indicates that groundwater nitrate and precipitation values are not independent during recharge months ($\alpha = 0.05$). Assuming piston flow is dominant, one would expect that during recharge months, when the soil is at or near field capacity, that proportionately more rainfall would displace more water from the lower vadose zone. Additionally assuming that nitrate concentrations in the vadose zone are higher than in groundwater (supported by Shelby (1995), Nelson (2003), and Feaga et al. (2004)), one would expect groundwater nitrate concentrations to increase when greater volumes of high concentration vadose water are expelled into the shallow aquifer. The above hypothesis is tentative, however, and changes in cropping practices could potentially reverse the trend if vadose nitrate concentrations become lower than groundwater nitrate concentrations.

Additionally, significance of the Kendall's coefficient of concordance supporting the hypothesis that greater precipitation correlates with higher groundwater nitrate concentrations during recharge is questionable. Though the hypothesis test did find that November, December, January, and April concentrations are dependent with respect to precipitation, the test is strongly influenced by the April rankings and does not necessarily imply dependence in November, December, and January data. The data collected in the winter of water year 2005 in many ways is comparable to data for two winters because two large flushing events were observed, one lasting from November to January and one in April. Since the uncommonly wet late March and early April of 2005 was preceded by 2.5 months of relative dryness, regional soil matric potentials had decreased (Van Verseveld, unpublished data) and it is likely that soil water nitrate became more concentrated. Therefore, the March-April rains likely caused a second seasonal flush by remobilizing the soil water nitrate. It is hypothesized that the well network showed a stronger seasonal signal after the April flush because it was shorter in duration and more intense than the earlier event, which would explain why the April rankings substantially differ from November, December, and January rankings (peak recharge concentrations were observed in 14 of 19 wells during April). Though modeled data presented in Chapter 3 support the above hypothesis, further work and more data collection during recharge months is necessary to fully determine the validity of the precipitation-groundwater nitrate correlation hypothesis for recharge months.

2.5.4. Hydrogeologic Differences in NO_3-N

Data collected in this study are consistent with those from Aitken et al. (2003), Eldridge (2003), and Vick (2004), which found that wells overlain by the Willamette silt are typically lower in concentration than those where the Willamette aquifer is exposed at the surface. Using the hydrogeologic units of Gannet and Caldwell (1998), the median concentrations and variances between the silt and alluvial aquifer are significantly different (see Table 2.2). Additionally, as shown in Figures 2.2 and 2.3, groundwater nitrate has much greater spatial variability than temporal variability.

The differences in both variance and median values of the silt and Holocene alluvial deposits are not particularly surprising given land use distribution, soil, and aquifer characteristics. As noted earlier, the soils overlying the floodplain alluvium are more permeable than those overlying the silts, and therefore more irrigation (and N) intensive crops have traditionally been cultivated on the alluvium. Therefore, higher groundwater nitrate concentrations in areas where the aquifer is exposed is expected based on the following characteristics. 1) Greater fertilizer inputs and more intense irrigation should increase the amount and concentration of leachate from fields overlying the alluvial sediments. Grass seed, the dominant crop on the Willamette silt (historically and presently), comparatively has little irrigative demand and is extremely efficient at N processing (Young et al., 2000). 2) The Willamette silt unit is more fine grained and has significant field tiling. Tiled fields have less nitrate leachate entering the aquifer than untiled fields, as recharge is diverted by tile export (Warren, 2002). 3) Fine grained sediments of the Willamette silt are more likely to have surface ponding, which can lead to anaerobic soil conditions and biotic denitrification. 4) Abiotic denitrification has been identified as a likely mechanism causing greater nitrate attenuation for sites where the Willamette silt is present with sufficient thickness (Arighi, 2004). 5) Rural population densities are generally lower in regions of the GWMA where the Willamette silt is present (see Appendix D). Thus, in areas where the Willamette aquifer is exposed, higher septic densities are likely to lead to higher groundwater nitrate concentrations.

Of the factors listed above, the difference in variabilities between the geologic units is most likely explained by the total N loading and the differing transport characteristics of the vadose material. Higher N loading and quicker recharge rates for the alluvium should translate into greater inter-monthly variability since recharge waters should be more voluminous and of higher concentration.

2.5.5. Variability Scales with Concentration

Analysis of the variability between different sampling sites indicate that the variance observed scales with concentration, or more explicitly, wells with higher median concentrations typically will have higher variabilities (Figure 2.8). This hypothesis is further supported by the outcome of two Kruskal-Wallace hypothesis tests (Table 2.2), where non-normalized site distributions differed significantly ($p = 0.005$) while normalized sites did not. The outcome of these two hypothesis tests indicate that when sample sites are normalized by their median concentration, the site distributions are not different. This implies that wells with higher concentrations have proportionately higher variabilites. Considering that areas where the aquifer is not overlain by the Willamette silt are more vulnerable to nitrate contamination and are expected to have greater concentration fluctuations, it is not surprising that wells with higher concentrations would show higher variabilities in the SWV. Wells with high concentrations are more likely to occur in areas where the aquifer is exposed (due to more intensive land management practices), and because high intensity agriculture occurs in areas with higher recharge rates, high variability would be expected to occur in the areas with high concentrations. Therefore, the scaling of variability with concentration is believed to largely be a function of overlapping land use and hydrogeologic distributions. It is likely that increasing variability with increasing concentration is a trend that is applicable to other locations, as Katz and Böhlke (2000) found that higher concentration wells in Florida typically showed the highest monthly fluctuations. Areas where such scaling would not be expected include regions where the observed groundwater nitrate concentrations are disconnected from the source, as in cases where the leaching source is significantly up-gradient from the region concerned about the high groundwater nitrate concentrations.

2.5.6. Implications for Long-Term Trend Analysis

Data collected in this study suggest that substantial intra-well variability in groundwater nitrate due to seasonal effects can largely be muted at the regional-network scale, due to land use, aquifer, and vadose heterogeneities. Therefore, network sampling design may not need as much focus on sample frequency and timing, but more on having a representative spatial distribution of wells (as suggested by Figures 2.2 and 2.3). Additionally, network sampling at quarterly intervals should be able to show long term concentration change with equal validity as monthly network data.

Modeled nitrate leaching data for the SWV (presented in Chapter 3) indicate that seasonal leaching during the winter of 2004-2005 was lower than average. If the modeled data are correct, it is possible that in a wet winter, higher intra-well variabilities will be observed. It is unclear if higher amplitude intra-well variabilities would result in statistically significant network-wide monthly differences. Based on the heterogeneity controlling well response, higher precipitation would not be expected to impact the conclusion that monthly differences at the network scale are muted. However, a higher precipitation winter probably would make the lumped analysis of recharge versus non-recharge months show seasonality. Comparisons of lumped monthly nitrate concentrations would more likely show seasonality because the out-of-phase network response to monthly precipitation would have less impact on the seasonal trend analysis. This interpretation is supported by the observation that seasonality defined by recharge versus non-recharge months in this study had greater statistical significance ($p = 0.30$) than seasonality defined by monthly population differences.

2.6. Summary and Conclusions

Groundwater nitrate in the SWV shows substantial monthly intra-well variability, but does not exhibit significant network-wide seasonal trends. Probable reasons why large local fluctuations were observed without defined regional seasonality are tied to the heterogeneity (including but not limited to geologic and hydrogeologic units, soils, present and past land management, vadose transport properties, aquifer mixing, and well depth and quality) of the system being sampled. Much smaller scale (15.5 km^2 and <2.5 km^2) aquifer studies have found substantial inter-well variabilities tied exclusively to study site heterogeneities (Mitchell et al., 2005; Wilcox, 2003).

A major implication for nitrate monitoring networks is that though high intra-well seasonal variability can exist, network-wide seasonal effects at any time period should be minimal or dampened (assuming that the well-network is spatially representative). The lack of defined network-wide seasonality for monthly data implies that network sampling at quarterly intervals should be able to show long term concentration change with equal validity. This finding is positive from a network managing perspective (alleviates cost), but is less positive from a well owner's perspective (samples from individual wells are likely to have seasonal concentration fluctuations and therefore less confidence can be placed in an individual sample's ability to be representative of the well's average annual concentration). A recommended monitoring policy for quarterly baseline data collection is that a small subset of wells should be sampled monthly to determine seasonal impacts on lumped monthly data (recharge versus non-recharge months) and local intra-well variabilities.

A trend observed in this study that warrants further investigation is an apparent increase in groundwater nitrate with increasing precipitation during recharge months. Future work could examine this relationship in more detail and also examine long-term groundwater nitrate trends. Further work that could be done in the SWV includes deep vadose tracer tests, where tracers are applied at the surface and monitored in groundwater. Shallow vadose tracer tests have been done (Feaga, 2003), but no empirical data exist to show how long it will take for widespread groundwater best management practices to affect aquifer nitrate concentrations.

Significant differences were found between nitrate concentrations and population variabilites for the major hydrogeologic units in the SWV. These differences can largely be explained by historical land use patterns as well as physical and chemical properties of the units. Observed spatial differences in groundwater nitrate were much greater than the observed temporal differences.

Lastly, seasonal fluctuations were found to be of greater magnitude in wells with higher median concentrations. This is likely interrelated with the differences found between hydrogeologic units, as higher concentration (and hence higher variability) sites are typically found where the Willamette aquifer is exposed at the surface (which is also where higher intensity agricultural occurs).

3. Understanding Present and Future Regional Nitrate Leaching via SWAT for the Southern Willamette Valley, OR.

3.1. Abstract

Population growth and uncertainties in future land use/land cover (LULC) distributions create difficult challenges for assessing the future availability and quality of drinking water. To examine the effects of LULC change on nitrate leaching, the Soil and Water Assessment Tool (SWAT) was employed for the Southern Willamette Valley (SWV) of Oregon, a region with groundwater nitrate issues that is expected to nearly double in population in the next 50 years. Following model calibration, recharge and nitrate leaching values from SWAT were found to be closely related to available recharge estimates and present day groundwater nitrate concentrations. Temporal variations in nitrate leaching were also found to agree reasonably well with observed monthly fluctuations in groundwater nitrate.

Three alternative future LULC scenarios for the year 2050 were run in SWAT to determine the relative change in recharge and nitrate leaching caused by potential LULC changes. Additionally, Groundwater Best Management Practices (GW-BMPs) for nitrate were examined for present and future scenarios. Modeled data indicate that basin-wide GW-BMP implementation with present LULC more greatly impacts nitrate leaching than changes in LULC. Future shifts in LULC as well as GW-BMP implementation could potentially lessen basin-wide nitrate leaching by 32-46% of present values. These conclusions are based on modeling which includes the effects of agrarian GW-BMPs, but do not consider the effects of future changes in septic loading. This is the first study to successfully use SWAT as a tool to examine the spatial and temporal variability of nitrate leaching.

3.2. Introduction

Land use/land cover (LULC) change in many areas of the United States is a growing concern from community, public planning, and hydrologic perspectives. The growth of urban regions and their encroachment onto rural lands and small communities significantly impact rural economies and their resident's way of life (Theobold, 1988). Public planners are presently trying to assess and address this problem and understand how future land use trends are likely to impact local communities (Foley et al., 2005; Klepinger, 2005; Hulse et al., 2002). Impacts of population growth and LULC change are evident on the hydrologic cycle, as there are greater consumptive demands and changes in runoff (Noorazuan et al., 2003; Foley et al, 2005), recharge (Kim and Sultan, 2002), evapotranspiration (Pielke et al., 2002; Foley et al., 2005), and albedo (Bonan, 1997; Pielke et al., 2002; Feddema et al., 2005) associated with LULC change. Changes in LULC are believed to cause equal or greater local hydrologic effects than greenhouse-gas induced climate warming (Bonan, 1997; Pielke, 2005).

Hydrologic models are useful tools for determining the probable effects of LULC change on local hydrology and aqueous geochemistry. In regions where nonpoint source pollution is dominant, regional models are often the only feasible way to examine likely impacts of land use change on pollutant concentration.

The Southern Willamette Valley (SWV) of Oregon is a region where nonpoint source nitrate loading has substantially impacted groundwater nitrate concentrations. Nitrate concentrations in the Willamette aquifer have generated significant public concern and the Oregon Department of Environmental Quality has recently declared an 548 km^2 Groundwater Management Area (GWMA). As part of the GWMA mandate, groundwater monitoring must occur until regional groundwater nitrate concentrations are below 70% of the US-EPA's 10 mg/L maximum contaminant level. Since high nitrate concentrations observed in the vadose zone are associated with land use (Feaga et al., 2004), it is of great interest to determine how Groundwater Best Management Practices (GW-BMPs) will impact groundwater nitrate concentrations.

Several Willamette Valley studies have identified potential GW-BMPs (Feaga et al., 2004; Vick, 2004; Satell et al., 1999), but recent land use changes from row crops and peppermint (high leaching potential crops (Feaga et al., 2004)) to grass seed (lower leaching potential crops (Young et al., 2000; Feaga et al., 2004)) after a market shift could cause groundwater nitrate concentrations to significantly decline without implementing agrarian GW-BMPs.

Objectives of this study are to model nonpoint source nitrate leaching dynamics for present LULC distributions, to predict future nitrate leaching based on projected LULC scenarios, and to examine the likely impacts of basin-wide GW-BMP implementation.

3.3. Methods

3.3.1. Study Area Information

The SWV is located in west-central Oregon and is located between Albany and Eugene and bounded by the Coast Range and Cascade Mountains (see Figure 3.1). Net relief for the study area is 915 m, but the region of focus, where the GWMA is located, is on the broad and flat valley floor. Upland catchments are largely forested, with stands varying in age and composition. Land use on the valley floor is mostly agrarian, with major crops being grass seed, hay, wheat, filberts, sweet corn, and peppermint. Assorted berries, fruits, and row crops are also grown. Larger urban centers in the modeled region include Eugene and Corvallis, while smaller municipalities include Junction City, Harrisburg, Coburg, and Monroe.

Climate in the SWV is Mediterranean, characterized by cool wet winters and warm dry summers. Mean annual precipitation for the low-relief SWV is 1109 mm, with approximately 80% of the annual precipitation falling between October and March. Monthly temperatures on average range from 4.5 C^{o} in December to 19.4 C^{o} in August (OCS, 2006).

Soils in the study region are primarily silt loams and silty clay loams, with generally more well-drained soils (including some loams, fine sandy loams, and gravelly sandy loams) in the Willamette River floodplain (Knezevich, 1975; Patching 1987). Irrigated crops with higher nitrogen (N) demands (such as peppermint, vegetables, and row crops) are mostly grown in the coarser soils of the floodplain region, while grass seed (predominantly annual ryegrass, perennial ryegrass, and tall fescue) has historically been grown on the more poorly drained, silty soils. The parent material of the poorly drained soils is the Willamette silt geologic unit, which confines the Willamette aquifer where present. The Willamette aquifer is unconfined in the Willamette River floodplain region, and typically underlies the more well-drained loam soils. The Willamette aquifer is composed of alluvial sands, gravels, and silts (O'Connor et al., 2001).

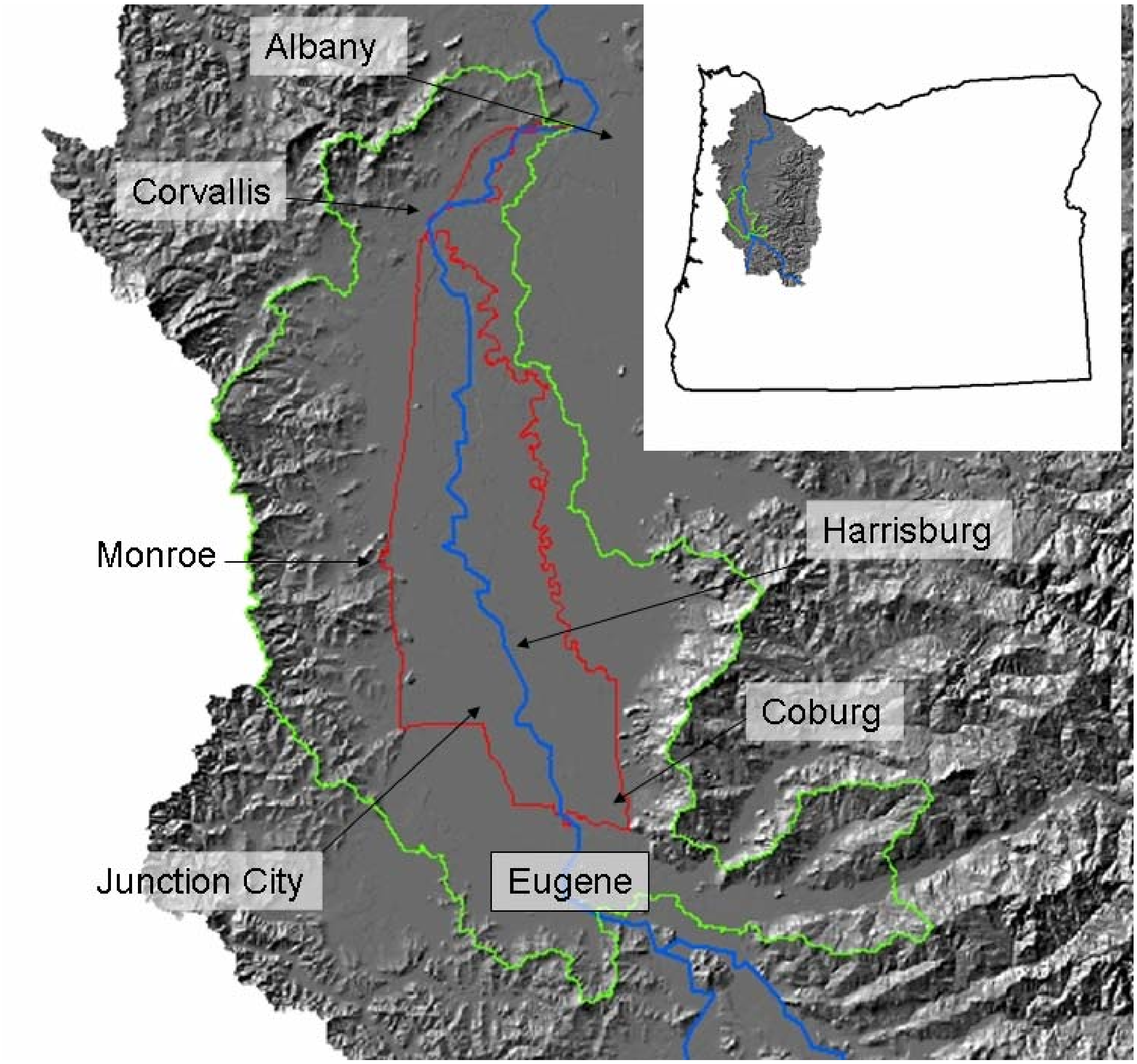

Figure 3.1. The Southern Willamette Valley of Oregon, with communities. The Groundwater Management Area is outlined in red, while the green outline is the study area modeled. The Willamette River is shown in blue.

3.3.2. SWAT Background Information

The Soil Water Assessment Tool (SWAT) is a deterministic, semi-distributed basin-scale model that runs on a daily timestep. SWAT was designed to predict the impact of land management practices on water, agricultural chemical, and sediment yields in large watersheds with varying soils, land use and management conditions for long periods of time (Neitsch et al., 2002b). Earlier models that significantly influenced the development of SWAT include CREAMS, GLEAMS, and EPIC, while SWAT is a direct development from its predecessor model SWRBB (Neitsch et al., 2002b). The models QUAL2-E and SWMM have also influenced specific modules within SWAT. The most senior model listed, CREAMS, was developed in 1980 and resulted from work of the 1970s that focused creating non-point source models after the passage of the Clean Water Act (Arnold and Fohrer, 2005).

SWAT is a physically based model, but solves for physical processes conceptually by using simplified analytical solutions and empirical equations (Abu El-Nasr et al., 2005). The code for SWAT was written with the objective of simulating all major hydrologic components as simply and realistically as possible, and to use inputs readily available over large spatial scales to enhance the likelihood that the model would become routinely used in planning and water resource decision making (Arnold and Fohrer, 2005). Applications of SWAT have generally been limited to surface water investigations, though some studies have examined recharge (Arnold and Allen, 1996; Sun and Cornish, 2005) and soil N processes (Pohlert et al., 2005).

To assess nitrate leaching in the SWV, AVSWAT2000 version 1.0, the 2000 version of SWAT that runs in ArcView GIS, was chosen over other nitrate leaching models because of its ability to simulate large catchments with diverse crop types. This made SWAT a favorable model over GLEAMS (a field scale model without a widely available GIS interface to examine N loading) (Knisel, 1993; Leonard et al., 1987), NLEAP (a field scale N leaching model with limited crop types and no widely available GIS interface) (Shaffer et al., 1991), and DAISY (a cost-prohibitive commercial 1-D carbon and N model that can be applied to spatially distributed areas in a GIS) (Hansen et al., 1991).

In SWAT, water entering the soil profile is initially determined by subtracting the calculated surface runoff abstractions from the total daily precipitation and irrigation. Once entering the soil profile, water can be removed via evapotranspirative demands, lateral flow into streams, or aquifer recharge. Vertical downward water movement in the soil occurs when the field capacity for a specific soil layer is exceeded, with the movement rate controlled by the layer's hydraulic conductivity. Soil water then successively fills up lower soil layers until it ultimately exits the soil profile and becomes shallow aquifer recharge. SWAT allows for capillary uplift of shallow groundwater to help meet evapotranspirative demands when the soil profile is dry. Nitrate movement in soils is generally governed by downward soil water movement in SWAT, but can also be uplifted into higher soil layers by capillary action (Nietsch et al., 2002). A conceptual illustration of soil water and N processes modeled by SWAT is shown in Figure 3.2. Of the N processes not modeled or inadequately modeled (as shown in Figure 3.2b), denitrification is likely the most crucial process for nitrate leaching because without it a major sink for soil N is missing. Ammonium transformations and uptake not accounted for by SWAT represent partitioning inconsistencies, but as the ammonium remains in the soil N cycle, they do not account for a net loss of N from the soil. Further discussion of the denitrification component in SWAT is discussed in the next section.

Stream base flow in SWAT is sustained mostly by recharge entering the shallow aquifer, and therefore when stream base flow percentages are well calibrated, recharge processes are also well calibrated (Sun and Cornish, 2005).

3.3.3. Modeling Approach

The modeled area for this study extends from south and east of Eugene north to Albany (see Figure 3.1). Spatial input data used include a 10 m digital elevation model (DEM) of the study area, a state soil survey soil map, and a recent LULC map (developed by Hulse et al. (2002) for Willamette Valley with Landsat images at 30 m resolution. Boundary conditions were defined for the model during data preprocessing to account for all stream flow moving through the Southern Willamette Valley. Therefore all catchments downstream of input hydrographs were modeled, making the total modeled area 2014 km^2. DEM processing in AVSWAT created subbasins, within which smaller hydrologic response units (HRUs) were created. HRUs are regions within a subbasin that have unique soil and land use properties (Neitsch, et al., 2002a). A total of 57 subbasins were created (see Figure 3.3) and 743 HRUs were developed for the present-day SWV

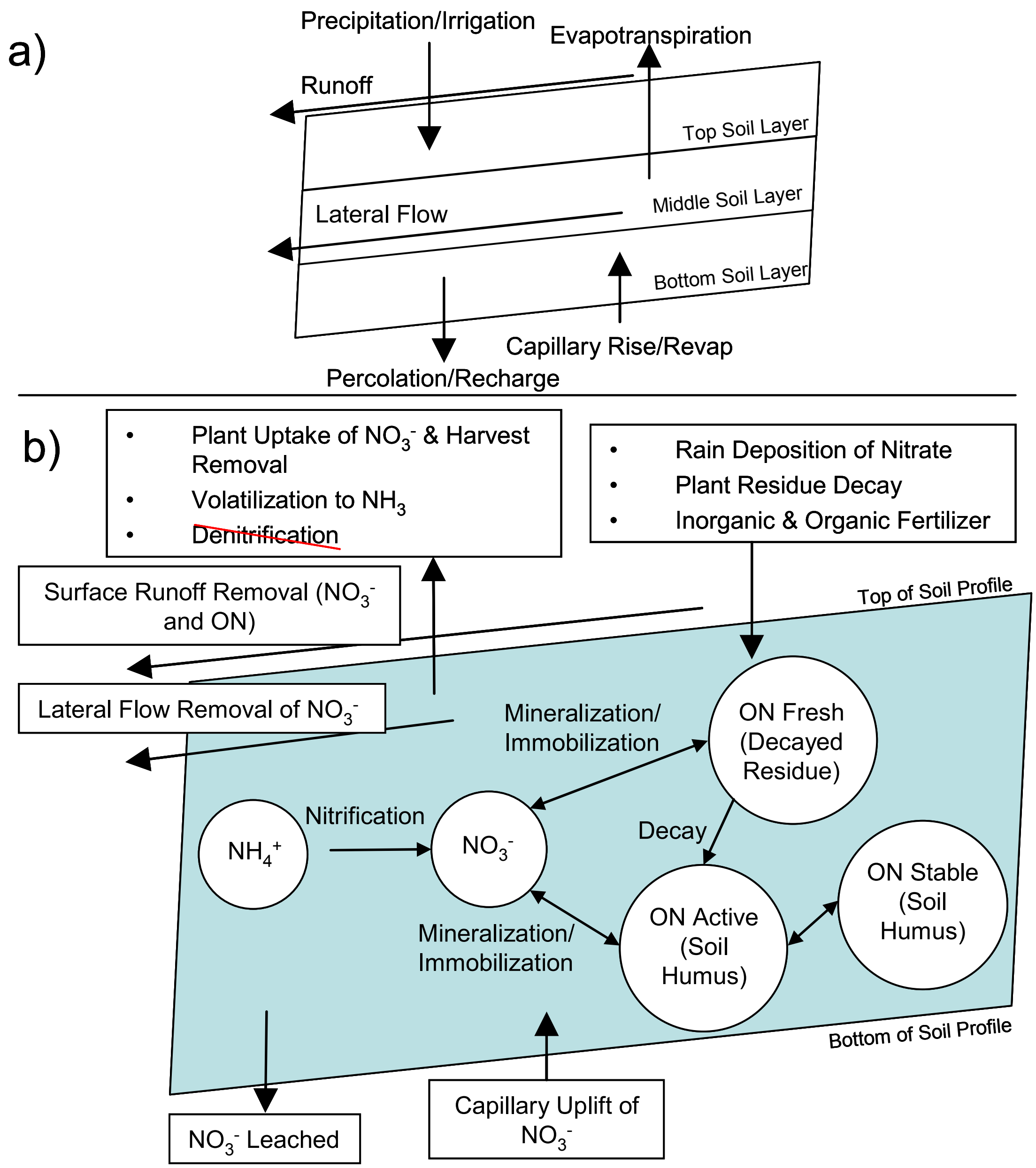

Figure 3.2. Soil processes in SWAT. a) Conceptual soil water processes modeled by SWAT. Though not well depicted in the illustration, lateral flow can drain any soil layer (10 are possible). An optional process which is not shown in the figure is crack flow. b) Soil nitrogen inputs, outputs, and cycling pathways in SWAT. Text boxes represent inputs/outputs, while circles represent different soil nitrogen pools. Cycling processes modeled are indicated next to their arrows. "ON" is an abbreviation used for organic nitrogen. Major processes not included in the SWAT2000 code include plant uptake of ammonium, ammonification, and dissimilatory nitrate reduction to ammonium, while denitrification is currently modeled inadequately.

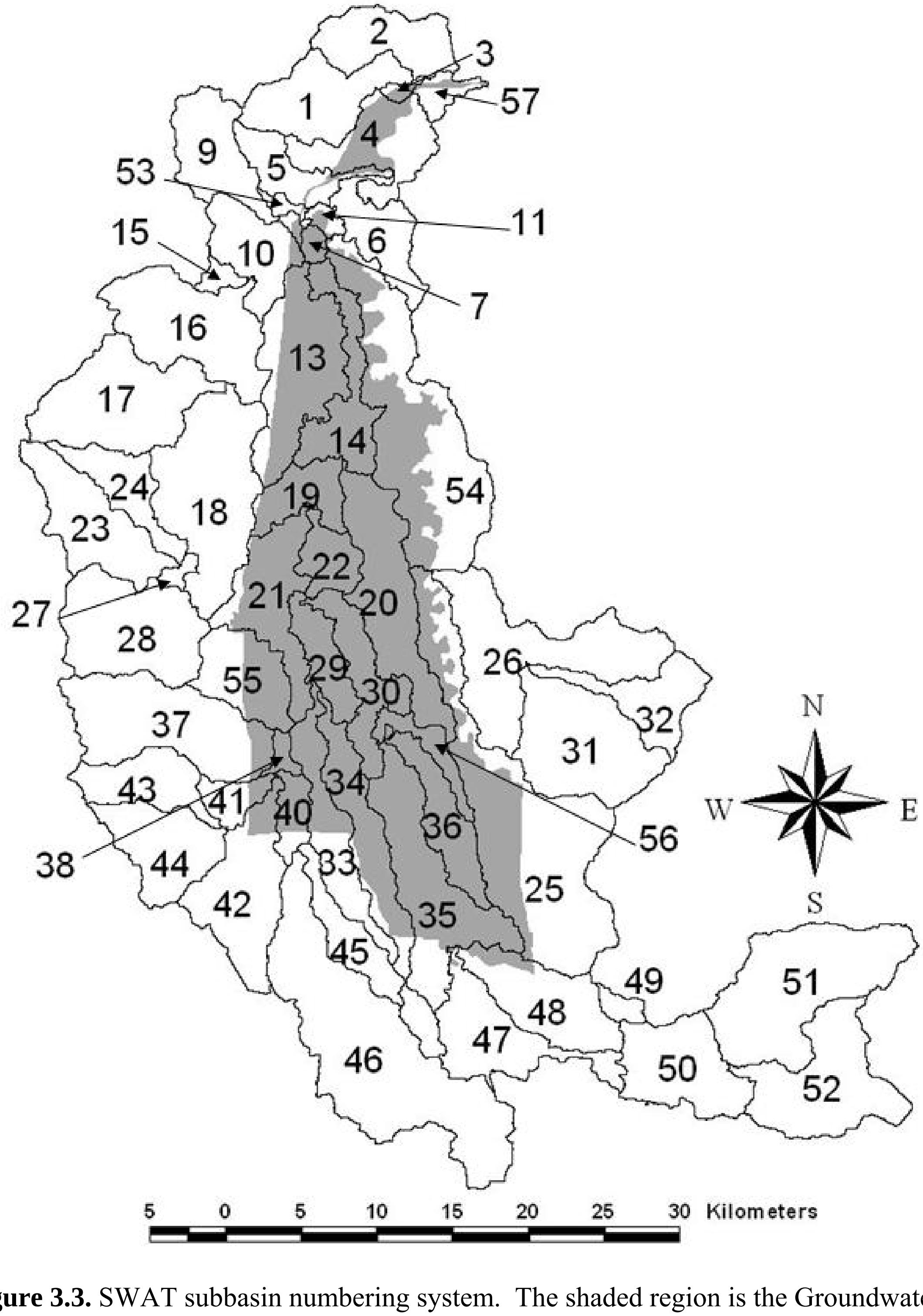

Figure 3.3. SWAT subbasin numbering system. The shaded region is the Groundwater Management Area.

simulation. The minimum threshold requirement for HRU creation was set so that 5% or more of the subbasin area needed to be covered by a land use or soil type for it to be modeled in HRUs. Subbasin outputs are calculated in SWAT by aerially weighting HRU outputs for the entire subbasin.

Soil parameters used in the model came from the state soil survey for Oregon, while soil depths were determined based on the depth to groundwater calculated in a calibrated MODFLOW model of regional groundwater flow (Craner, 2006). Soil depths from the groundwater model were used to simulate the entire depth of the vadose zone. In regions where the Willamette aquifer is semiconfined, the soil survey depth was used if the potentiometric surface elevation was above the land surface or if it made the soil shallower than recorded in the state soil survey.

Land use designation within SWAT was assigned largely from the crop and land cover database available in AVSWAT. Several important land uses found in the SWV were not included in the land cover database but were created for this model from literature values (created land covers include peppermint, sugarbeet for seed, natural grasses, natural shrubs, and residential mid/high density. New crop input values used can be found in Appendix E). Land management scenarios created for agrarian HRUs were based on recommendations found in Oregon State University Agricultural Extension publications and from communication with local Extension agents (see Appendix F for management scenarios and references).

As noted previously, only the SWV and not the entire Southern Willamette Basin (SWB) was modeled (see Figure 3.1). The entire SWB was not modeled because 1) groundwater nitrate contamination is only a problem in the SWV, 2) modeling the entire SWB would increase the modeled area from 2014 km^2 to 11572 km^2, and 3) SWAT was designed for agrarian watersheds and does not perform as well in forested headwater catchments (Eckhardt et al., 2002), like those of the Willamette River Basin. For the above reasons, input hydrographs were used so that realistic flow would be in the Willamette River. Input hydrograph locations are shown in Figure 3.4, and daily hydrograph data were obtained from the US Geological Survey (USGS) and the US Army Corps of Engineers (USACE). Hydrograph data gaps that occurred during the simulation period (from January 1993 through May 2005) were filled via linear interpolation or with values obtained via a ratio model for gaps greater than several days (see Appendix G for information on the creation of the ratio model). Approximate values for the Walterville Canal, a diversion which diverts approximately 40% of the annual flow of the McKenzie River around the USGS stream gauge in Walterville, were included in model calibration and validation.

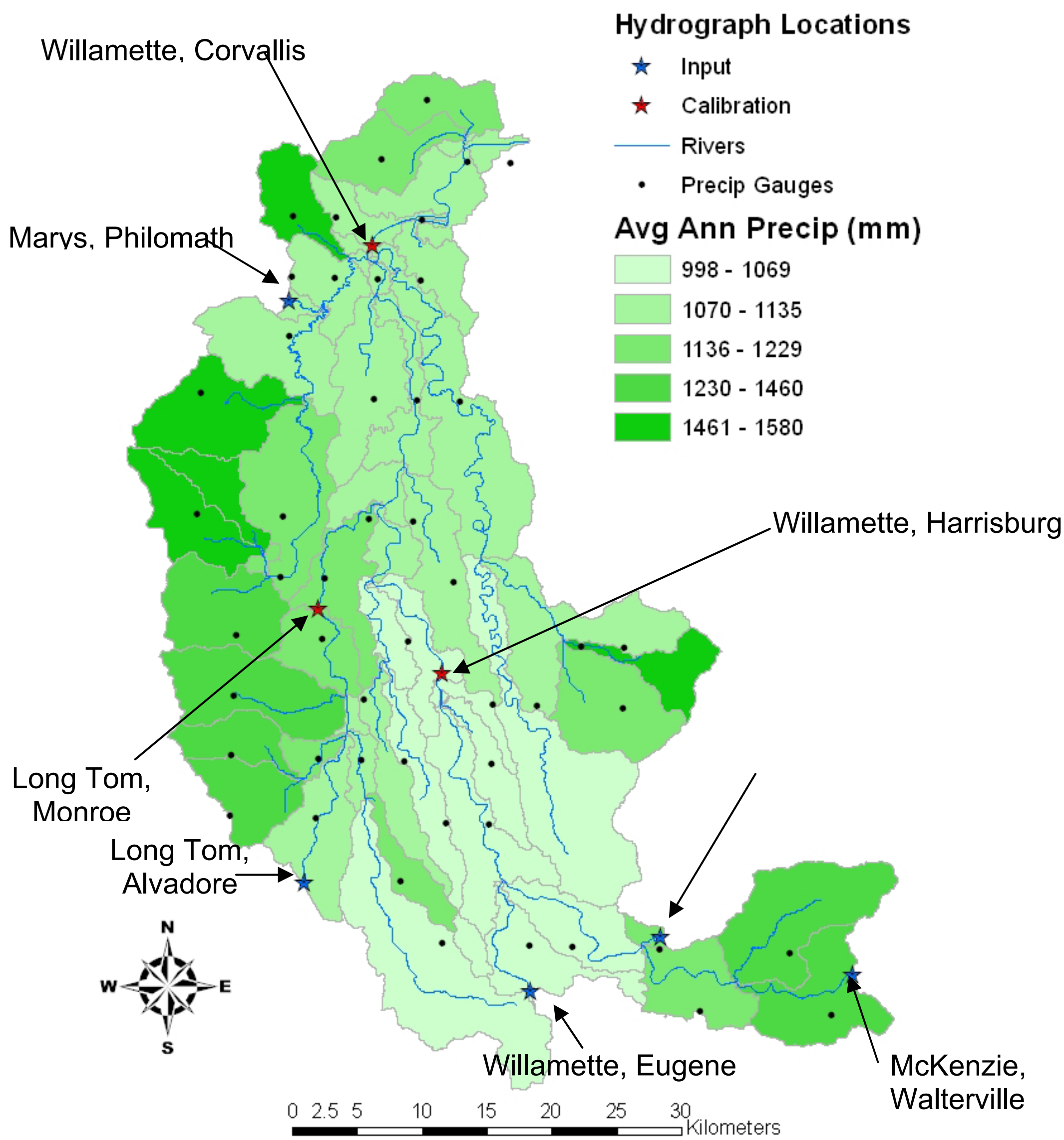

Figure 3.4. Average annual subbasin precipitation, using PRISM-derived rain gauges. Input and calibration hydrograph locations are also shown.

Climate data used in SWAT include precipitation, temperature, solar radiation, relative humidity, wind speed, and potential evapotranspiration. Solar radiation, wind speed, relative humidity, and potential evapotranspiration inputs were obtained from the Corvallis Agrimet site (US Bureau of Reclamation, 2005), while temperature input data came from the Oregon Climate Service for Eugene and Corvallis (OCS, 2006). Due to paucity of available precipitation data for the modeled area (4 rain gauge locations for a modeled area of 2014 km^2 having 915 m relief), we used spatially-distributed modeled data derived from the Precipitation Elevation Regressions on Independent Slopes Model (PRISM) of Daly and Neilson (1992). PRISM calculates monthly precipitation totals for 4 km grid cells and was chosen because of its high data quality, spatial detail, and temporal extent (Daly et al., 2002). To create daily precipitation values from PRISM data, the ratio between monthly PRISM cell values to monthly rain gauge data was determined for the cell's nearest rain gauge. Daily data were then derived for PRISM cells by multiplying daily rain gauge values by the PRISM cell's monthly ratio. PRISM derived rain gauges used by SWAT are shown in Figure 3.4, along with average subbasin precipitation. SWAT only allows one rain gauge to be assigned per subbasin, so most PRISM rain gauges created were not used.

3.3.4. Calibration

The present-day SWAT model was calibrated using USGS and USACE hydrographs from 1993 - 2001. Since much of the flow was derived outside of the modeled area, watershed derived flows were compared with SWAT watershed derived flows to obtain a more sensitive calibration. Watershed derived flow is defined as the flow at the calibration hydrograph minus the summed total of input hydrographs on a given reach. The four calibration points (see Figure 3.4) had their average annual watershed derived flow values calibrated as well as their average annual baseflow-runoff ratios. Surface calibration objectives were to have accurate watershed derived flow values and surface flow-baseflow ratios, with calibration variables including CN, ESCO, SOL_AWC, and REVAPMN. Baseflow separation was done using a program developed by Arnold et al. (1995). The objective function used for hydrographs was the Nash Sutcliffe (NS) efficiency coefficient (Nash and Sutcliffe, 1970), which is calculated:

$$NS = 1 - \frac{\sum \left(Q_{obs_i} - Q_{sim_i}\right)^2}{\sum \left(Q_{obs_i} - \overline{Q_{obs_i}}\right)^2}$$

where Q_{obs_i} = observed flow for time *i*

Q_{sim_i} = simulated flow for time *i*

$\overline{Q_{obs_i}}$ = mean observed flow from entire time period observed

High NS values are generally above 0.70, a value of 0 indicates that the model predicts equally well as the observed mean, and negative values indicate that the model gives less reliable results than the observed mean.

Following surface calibration, SWAT recharge values were examined and compared to available data. Since all available regional recharge values were model-derived (Woodward et al., 1998; Lee and Risley, 2002), SWAT values were only analyzed for spatial reasonability. Initial soil values assigned were revised after it was determined that areas expected to have lower recharge values (silty soils overlying the Willamette silt) had higher values than elsewhere. Input soil values were changed to aerially-weighted county values (from their initial state soil survey values) because state soil values were found to oversimplify the extremely heterogeneous county soils.

Nitrate calibration was performed after percolation calibration. To obtain reasonable nitrate leaching values, soil organic carbon values were set to zero to prevent denitrification from occurring. The SWAT2000 code grossly overestimates denitrification because when soil moisture exceeds 95% of field capacity, non-limited denitrification occurs as a function of soil organic carbon and temperature. As downward soil water movement only occurs at or above field capacity in SWAT, most of the mobile nitrate is stripped from soil water before movement (Pohlert et al., 2005; Neitsch et al., 2002b). Therefore, leaching values from this model cannot be considered absolute since denitrification losses are not included in the calculated values (and the model produces relative nitrate leaching magnitudes). The relative magnitudes of leaching, however, remain spatially comparable within the model and are believed to reflect expected leaching loads. All leaching loads further mentioned in this paper are relative leaching masses of NO_3-N. Minimal spatial calibration for nitrate leaching was performed, with only N uptake parameters for Italian ryegrass, tall fescue, and peppermint calibrated to reflect local values. Flow-weighted nitrate concentrations, which will be discussed, are calculated by dividing the summed mass of nitrate collected for every month in a sampling period by the summed volume collected.

3.3.5. Validation

Validation of stream flows was performed with data from 2002 – 2005. Recharge validation was not possible as no non-modeled regional recharge data set exists. Nitrate leaching data were compared with spatial groundwater nitrate data from Chapter 2, Vick (2004), Eldridge (2003), and Aitken et al. (2003) and temporal data presented in Chapter 2.

3.3.6. Projected Scenarios

After calibration and validation of the present scenario, futures scenarios were projected to examine likely effects of GW-BMP implementation and future LULC changes. Alternative futures used in this analysis include Plan Trend 2050, Development 2050, and Conservation 2050, all of which were developed by Hulse et al. (2002). The three scenarios are considered possible outcomes for the year 2050 and are based on different potential development strategies (as defined by numerous Willamette Basin stakeholders). The Plan Trend scenario is a potential outcome if present trends and policies continue, the Development scenario loosens current policies to have a more market-based development approach, while the Conservation scenario places a greater emphasis on ecosystem restoration and protection while remaining socially and economically viable for all stakeholders (Hulse et al., 2002). Additionally, all future scenarios experience the same population growth, but accommodate growth differently (with extremes being development having greater urban sprawl, while conservation has higher population densities within present urban growth boundaries).

To project future scenarios, the present model was rerun with the future LULC maps. All other inputs and variables were held constant across present and future simulations. Climate change data were not included in the future scenarios, but to an extent this weakness is mitigated by the finding that at local scales, land use can have as great or greater effects on climate (temperature, albedo, evapotranspiration, convective precipitation) than global warming (Bonan, 1997; Pielke, 2005). Finally, GW-BMP scenarios were run for present and future scenarios to determine how much of an impact relatively simple GW-BMPs are likely have on nitrate leaching in the study area.

GW-BMP scenarios used 13% less fertilizer and irrigation than the present land management scenarios. Growers of grass seed and row crops, the dominant land uses across the valley, typically apply 10-15% more fertilizer and irrigation water than recommended based on crop trials (author communication with Mellbye and McGrath, 2005). Therefore, GW-BMP scenarios for all crops use fertilization and irrigation rates recommended by Oregon State University Extension. An exception to this rule was applied to tall fescue, where instead of having grower rates 13% higher than recommended extension rates, grower rates were exact values supplied by agricultural extension agents (author communication with Mellbye and Hart, 2005) while GW-BMP rates were published extension rates. Additional GW-BMP modifications for the "row crop rotation" and "peppermint" land management scenarios include using winter cover crops between harvested crops when the harvested crops were not fall planted. Lastly, GW-BMPs were not applied to urban lawns because reliable fertilizer rates (to differentiate between GW-BMP and without GW-BMP) were unavailable. All GW-BMP and non-GW-BMP management scenarios can be found in Appendix F.

In the SWV SWAT model, subbasins where the Willamette or McKenzie Rivers cover greater than 5% of the area (subbasins 4, 14, 19, 22, 30, 35, 47, 48, and 57) have no calculated recharge occurring under the rivers (since the surface land use is water, thus having no soil profile). To make present subbasins more directly reflect anthropogenically-modified land use, both recharge and nitrate leaching map values were adjusted so that the relative area of large rivers within subbasins do not bias the recharge and leaching values observed. Future scenarios were normalized to the present modeled values that did not have river area corrections because future channel area expansion and contraction was considered an anthropogenic modification (for example, an objective of the Conservation 2050 scenario was to expand natural channels, while other alternative futures did not focus on channel expansion).

3.4. Results

NS efficiencies calculated from annual values for the Long Tom River in Monroe, the Willamette River in Harrisburg, the Corvallis drainage basin, and the entire drainage area upstream of Corvallis are shown in Table 3.1. The Monroe reach yielded the best NS values, while the Harrisburg reach had the lowest efficiency and grossly under-predicted flow, with hydrographs for both reaches found in Figure 3.5. Table 3.2 summarizes changes in LULC between different scenarios examined, while Table 3.3 shows average annual basin values of percolation, nitrate leaching, plant N uptake, and fertilizer applied for all scenarios examined.

Maps of present recharge and future recharge (expressed as a percentage of present recharge values) are shown in Figure 3.6. SWAT recharge values fall between estimated recharge values of 254 – 381 mm by Lee and Risley (2002) and 254-762 mm by Woodward et al. (1998). The land use change which had the greatest influence on recharge was urbanization, with declines generally occurring in areas with increased impervious surface area. GW-BMP scenarios for recharge were found to differ minimally from their corresponding non-GW-BMP scenario, and are therefore not presented as a figure.

Table 3.1. Calibration and validation Nash Sutcliffe values for surface flow at calibration hydrographs. Observed and simulated baseflow ratios agree closely. Differences between observed and simulated values at Harrisburg are likely related to large gains occurring in the reach.

	LT Monroe	Will Harris	Will Corv	Entire Basin
Calibration	0.66	-3.01	0.36	0.40
Validation	0.22	-0.15	-5.47	-1.87
Observed Baseflow	0.66	0.72	0.69	0.69
Simulated Baseflow	0.67	0.58	0.7	0.67

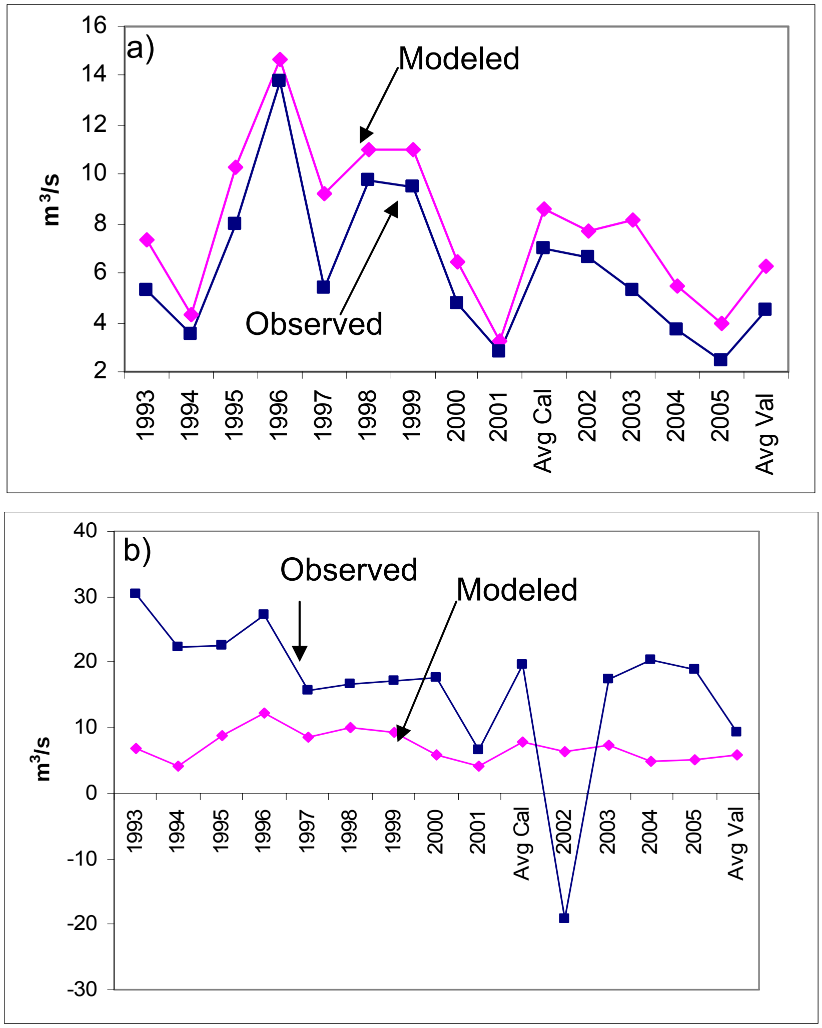

Figure 3.5. Example calibration and validation hydrographs of watershed derived flow. Years used in calibration were 1993-2001, while validation years were 2002-2005. a). Hydrograph for Long Tom River in Monroe, OR. This site had the highest Nash-Sutcliffe efficiencies in both calibration and validation periods. b). Willamette River in Harrisburg, OR. This site generally had the poorest fit hydrographs, with the model significantly under predicting flow all years except 2002. The under prediction of flow is believed to be associated with gains for the Harrisburg reach associated with deep aquifer flow paths, while the low observed flow value in 2002 is thought to be associated with input errors used in calculating the observed watershed derived flow.

Table 3.2. Area allocations for different land use types for the modeled area for present and future scenarios. a). Areas of land use defined for different scenarios. Land use types are composed of their corresponding SWAT land use classes. b). Total areas of land use grouped by land use types from Table 3.2a.

3.2a)

		Percent Area of Land Use Type for Different Scenarios			
Grouped Land Use Type	Composed of SWAT Land Use Classes:	Present	Plan Trend 2050	Development 2050	Conservation 2050
Urban	Urban Land Use + Roads	10.3	11.7	13.55	11.16
Roads	Roads	3.71	3.68	4.52	3.81
Forest	Evergreen+Deciduous+Mixed Forests	31.05	30.77	30.04	30.66
Irrigated Ag	Peppermint + Row Crop Rotation+ Orchards	4.69	1.23	1.11	0.46
RYEG	Italian (Annual) Ryegrass	24.68	31.43	27.66	23.2
FESC	Tall Fescue	13.81	16.52	16.53	16.39
WWHT	Winter wheat and Ryegrass Rotation	6.62	0.32	0.13	0.14
NATS	Natural Vegitation (shrubs)	7.09	6.33	9.4	7.97
NATG	Natural Vegitation (grass) + Oak Savanna	0	0	0	7.72
WATR	River Channel	1.72	1.69	1.57	2.21

3.2b)

Totals of:	Composed of Land Use Types:	Present	Plan Trend 2050	Development 2050	Conservation 2050
Agricultural Land	Irrigated Ag+RYEG+FESC+WWHT	49.8	49.5	45.43	40.19
Natural Land	Forest+NATS+NATG+WATR	39.86	38.79	41.01	48.56
Total Non-Fertilized Land	Roads+Forest+NATS+NATG+WATR	43.57	42.47	45.53	52.37

Table 3.3. Average basin-wide values of recharge, potential nitrate leaching, plant nitrogen uptake, and fertilizer application. Ratios of projected futures and GW-BMPs to parent* and present values are calculated in the 3 rightmost columns.

	Average Annual Basin Values for Different LULC				% Change from Parent* Scenario, Nitrate Leaching	% Change from Parent* Scenario, Fertilizer Applied	% Change from Present Scenario, Nitrate Leaching
	Recharge (mm)	Nitrate Leached (kg/ha)	Plant N Uptake (kg/ha)	Fertilizer Applied (kg/ha)			
Present	460.85	51.27	66.23	106.02			
Present GW-BMP	459.53	37.53	65.97	90.85	-26.8	-14.3	-26.8
Plan Trend 2050	457.50	47.82	69.35	105.47			-6.7
Plan Trend 2050 GW-BMP	457.27	34.63	68.00	90.23	-27.6	-14.5	-32.5
Development 2050	454.22	44.32	67.77	99.94			-13.6
Development 2050 GW-BMP	453.97	31.92	66.40	85.55	-28.0	-14.4	-37.7
Conservation 2050	459.47	38.96	58.38	86.94			-24.0
Conservation 2050 GW-BMP	459.33	27.87	56.90	73.88	-28.5	-15.0	-45.6

* Parent scenario is defined as the original scenario to which the GW-BMP is applied. The parent scenario of Plan Trend 2050 GW-BMP is Plan Trend 2050.

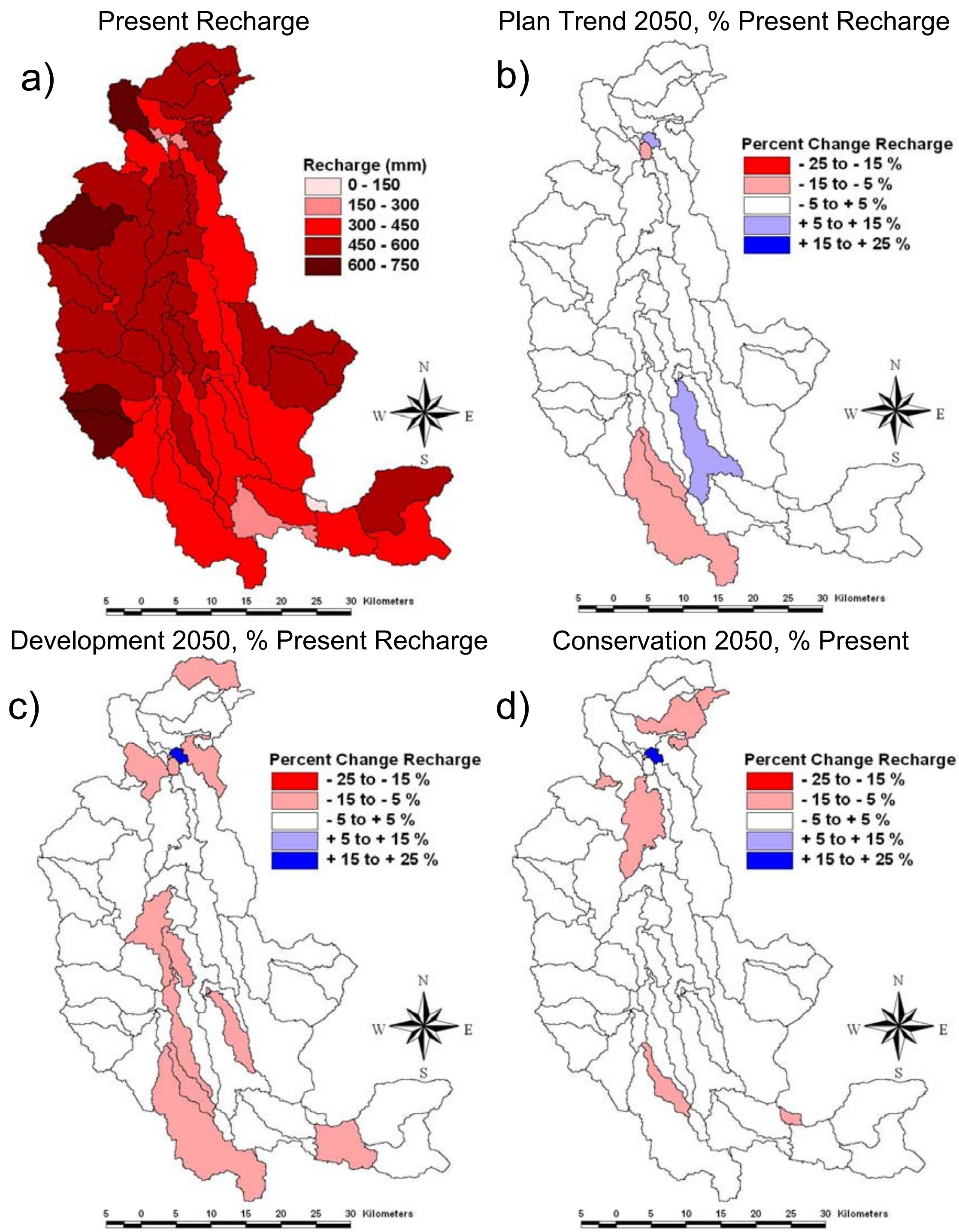

Figure 3.6. Simulated average annual recharge values as well as the percent difference between average annual present and future subbasin recharge values. a) present recharge b) Plan Trend 2050 recharge c) Development 2050 recharge, d) Conservation 2050.

Figure 3.7a shows modeled spatial nitrate leaching with validation groundwater nitrate data. Modeled data agree reasonably with observed data, except in basins 34 and 35. Figure 3.7b shows the leaching potential if valley-wide GW-BMPs were implemented, while Figure 3.7c displays present GW-BMP leaching values as a percent change from present leaching values. Figure 3.8 compares the percent difference (relative to the present) of nitrate leaching between future and GW-BMP scenarios. Figure 3.8 considers leaching effects of agricultural practices, but does not consider septic leaching (which is thought to be a major nitrate source in localities the SWV). In general, future scenarios show the greatest changes in nitrate leaching where irrigated crops (mostly near the Willamette River) are replaced with grass seed or natural lands or when urban areas expand into previous forest or natural shrub areas. GW-BMP scenarios showed significantly greater declines in leaching relative to their non-GW-BMP counterpart, with average leaching declines of ~ 28% (see Table 3.3).

Figures 3.9 and 3.10 were used for validation of the N cycling and leaching component of the model. Figure 3.9 compares median monthly groundwater nitrate concentrations from August 2004 through May 2005 (field data originally presented Chapter 2) with relative nitrate leaching masses derived for the modeled basin by SWAT. Beginning in November 2004, when recharge commenced, SWAT leaching dynamics exhibit similar trends to the groundwater nitrate data. Relative monthly nitrate concentrations (flow-weighted and averaged for the entire basin) for leachate are compared with observed flow-weighted average nitrate concentrations from peppermint and row crop fields in the Willamette Valley in Figure 3.10. General leachate concentration trends are similar between observed and simulated values.

The relationship between precipitation and nitrate mass leached is investigated in Figure 3.11. Notably, the linear trend exhibited is similar to that observed in groundwater nitrate concentrations presented in Chapter 2.

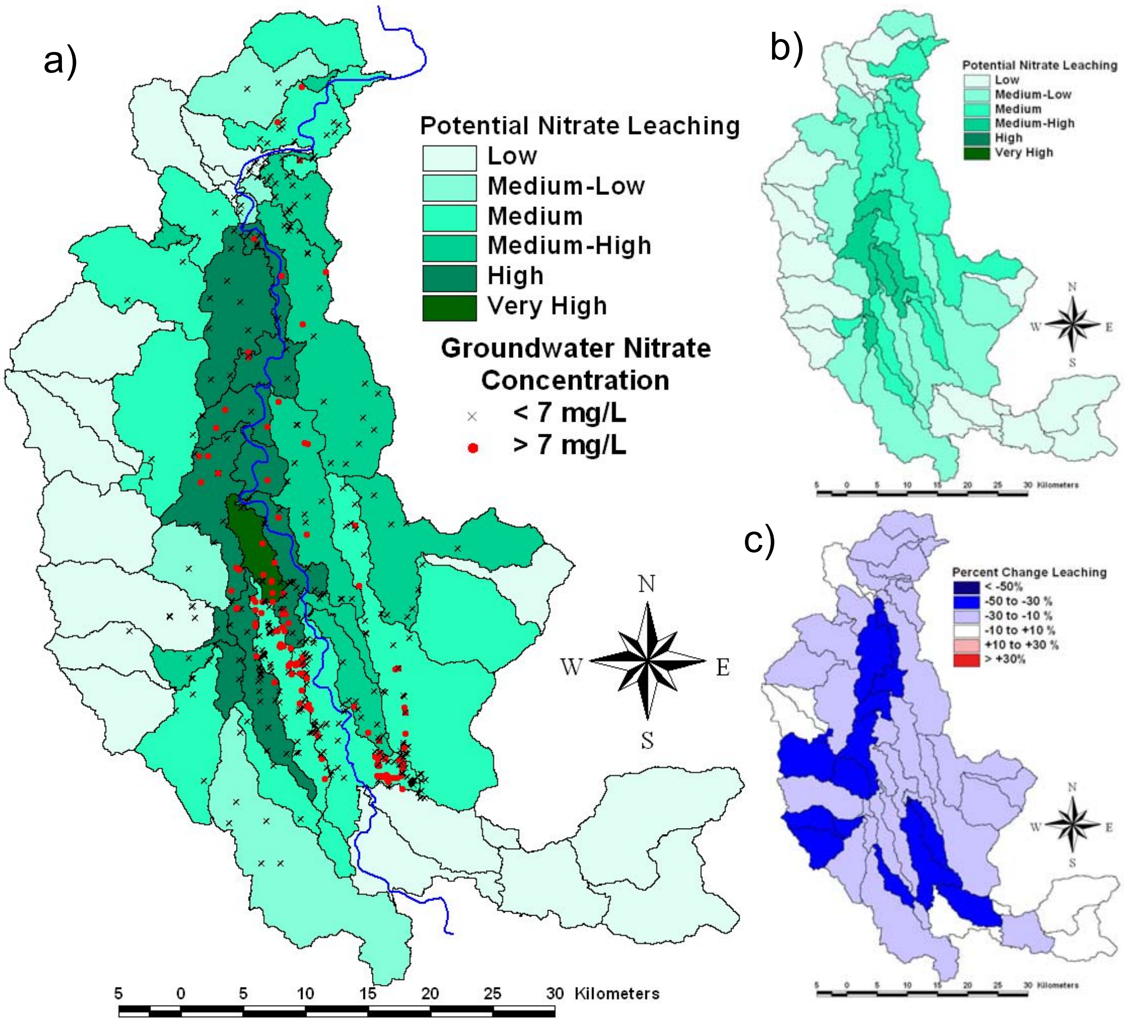

Figure 3.7. Average annual potential nitrate leaching potential based on present LULC. a) Present potential nitrate leaching with groundwater nitrate concentrations. 7 mg/L NO_3-N is used as a cutoff value for low concentrations because the Groundwater Management Area was declared based on a regional distribution of wells at or above 7 mg/L. b) Present potential leaching if GW-BMPs are implemented. c) Present potential leaching if GW-BMPs are implemented, expressed in percent change from present leaching without GW-BMPs.

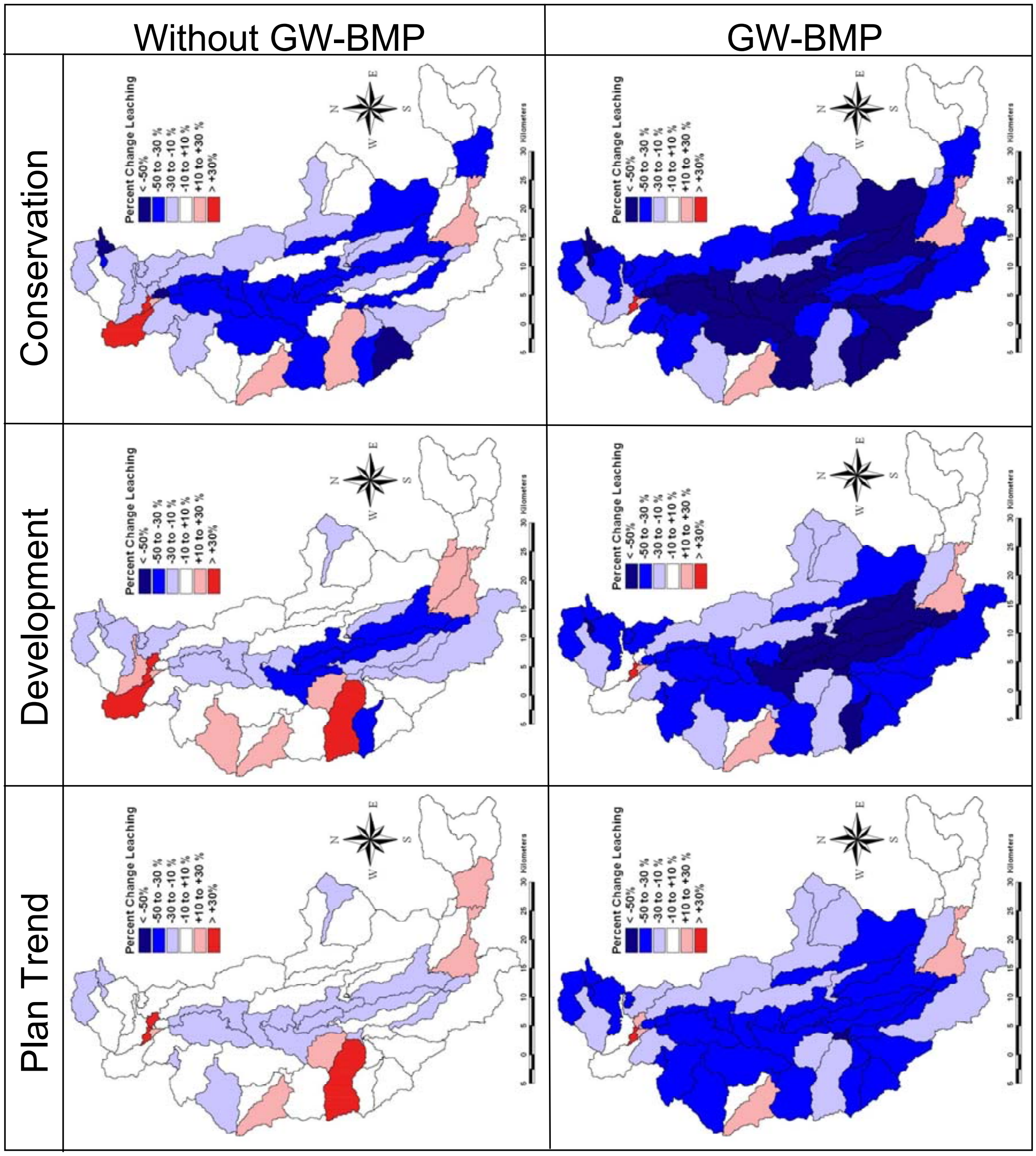

Figure 3.8. Potential nitrate leaching for future scenarios, with and without GW-BMPs. Without GW-BMP and GW-BMP values are expressed as the percent change from present subbasin leaching values. These results consider the adoption of agricultural GW-BMPs, but do not consider changes in septic loading or the impacts of denitrification. Color classes are as follows: Dark Blue = < -50%, Blue = -50 to -30%, Light Blue = -30 to -10%, White = -10 to +10%, Pink = +10 to +30%, Red = > +30%.

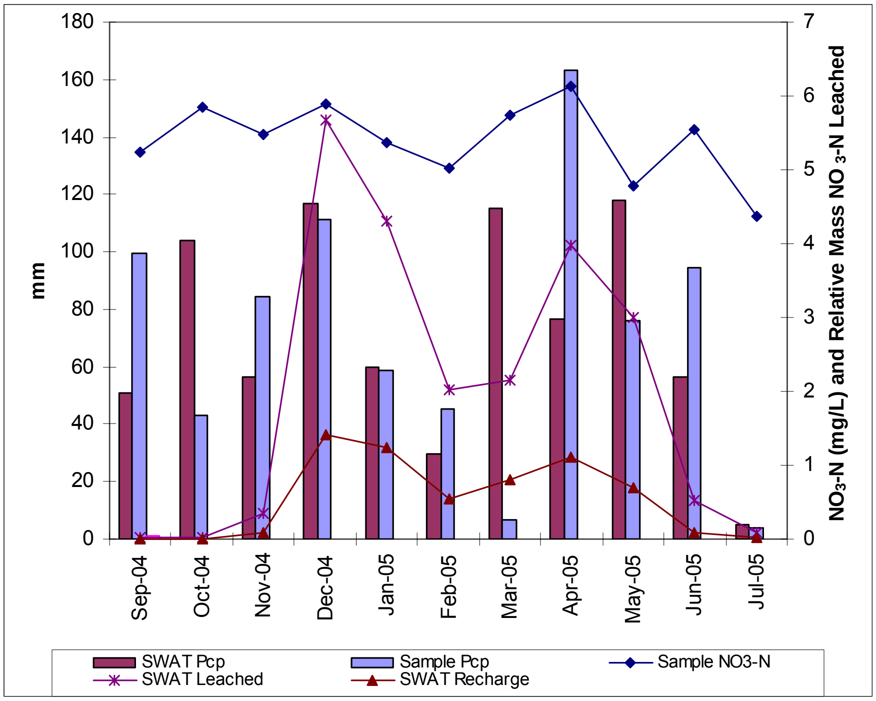

Figure 3.9. Monthly median groundwater nitrate concentrations compared to simulated SWAT nitrate leaching masses. SWAT leaching masses are presented unitless to prevent misinterpretation, as the effects of denitrification are not reflected in SWAT leaching masses. Precipitation values between monthly sampling events (for observed data, which was collected mid-month) and for calendar months (SWAT data), as well as the basin-average monthly recharge values from SWAT are presented. Beginning in November, recharge occurs and groundwater nitrate values are expected to be influenced by monthly leachate masses. The final month where significant recharge was confirmed with field data was April, and therefore groundwater nitrate concentrations are not expected to reflect leachate masses as closely thereafter.

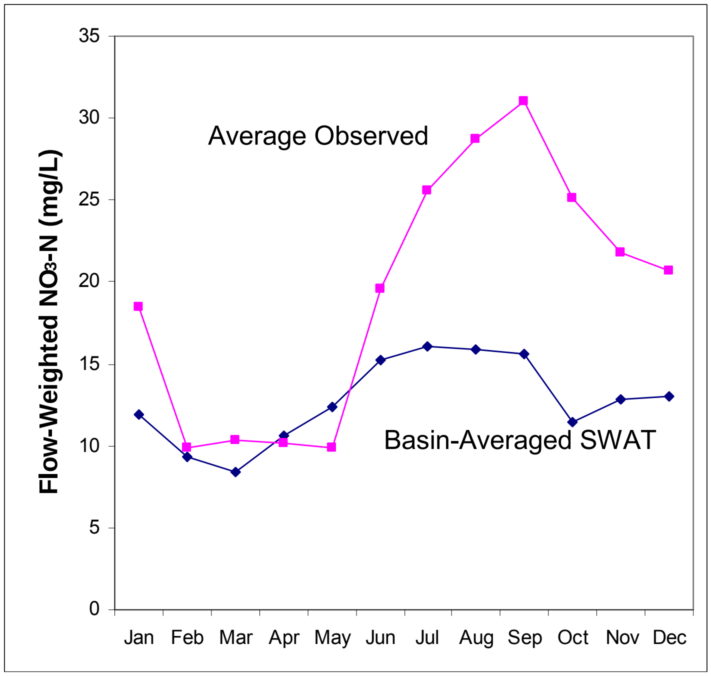

Figure 3.10. Average monthly nitrate leachate (flow-weighted from 1993-2005) values from SWAT compared to observed flow-weighted leachate concentrations. Observed concentrations are average monthly values for peppermint and row crops in the Willamette Valley, based on the data of Feaga et al. (2004). Though SWAT concentrations are expressed in mg/L, they are not absolute values because denitrification was not accounted for in the model.

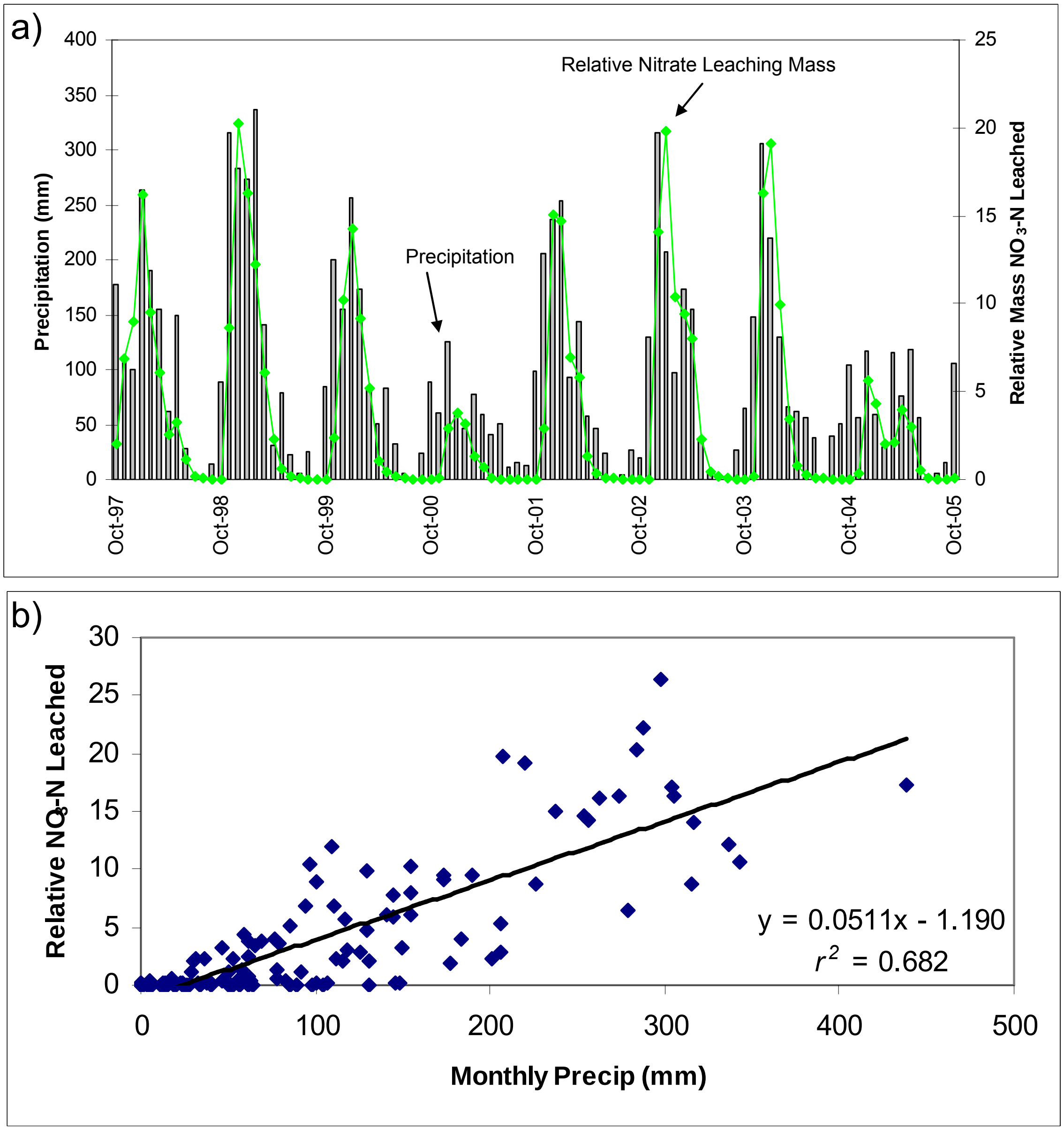

Figure 3.2. Relationships between modeled precipitation and nitrate leaching. a) Annual nitrate leaching trends b) comparison between leaching and precipitation values.

3.5. Discussion

3.5.1. Surface Calibration

Surface water calibration was done for the modeled basin at 3 locations, constituting 4 different calibration areas (Long Tom drainage, Harrisburg drainage, Corvallis drainage downstream of Harrisburg and Monroe, and the total modeled basin in Corvallis). All 4 calibration areas had reasonable surface flow-baseflow ratios, as shown in Table 3.1. Of the 4 calibration sites, only the Long Tom in Monroe had a relatively high calibration NS coefficient (0.66), while the Corvallis basins had similar efficiencies (0.36 Corvallis drainage and 0.40 entire modeled basin). The Harrisburg drainage had by far the worst efficiency (-3.01), and the poor modeled fit is believed to be related to gains and possibly to uncertainty in input hydrographs. The segment of the Willamette River immediately upstream of the Harrisburg gauging station has been documented with field data (from 1992, 1993, and 1996) to have significant gains (Laenen and Risley, 1997; Lee and Risley, 2002), with the volume of the gain varying by season and year. Laenen and Risley (1997) note that for the Harrisburg reach of the Willamette River, hyporheic flow can account for flow differences of up to 1000 cfs or 15% of the total river flow. Since SWAT does not simulate groundwater flow between subbasins, gains or losses associated with groundwater flow paths greater in scale than the subbasin will be unaccounted for. A regional groundwater model for the SWV developed by Craner (2006) indicates that flow paths associated with gains along Willamette River near Harrisburg are greater in scale than the SWAT subbasins modeled.

Uncertainty in input hydrographs could further impact the Harrisburg reach calibration. The modeled Walterville Canal data has error associated with it because the 40% flow approximation used does not simulate daily and monthly flow well (compared to available flow data for the canal from 2003-2006, with data provided through communication with Vandonkelaar, (2006)). Additionally, data from the USACE in Eugene did not undergo an internal quality check and could have significant error associated with it. The aforementioned reasons are why we believe that the Harrisburg

NS values for calibration and validation are poor. Uncertainties associated with inputs as well as gains and losses in the Willamette River are likely to be the cause of the relatively low NS coefficients for the Corvallis basins (Lee and Risley (2002) found the Willamette River between Eugene and Corvallis to gain approximately 470 cfs at certain periods in the year).

Surface water validation NS efficiencies declined significantly, relative to calibration NS values, for all hydrographs (except Harrisburg, which moved up to -0.15 from -3.01). Notably, in 2002 the watershed derived flow for the Harrisburg basin was -19.1 m^3/s (see Figure 3.5b) and the entire basin (in Corvallis) was -0.2 m^3/s. Such large magnitude differences in observed flow (no other years had negative watershed derived flow, and 2002 was not a drought year) could be caused by extreme hyporheic flow or more likely erroneous input/validation data.

Though surface water calibration and validation was unsatisfactory for the SWV using SWAT, this issue is largely irrelevant when examining nitrate leaching. Since surface flow-baseflow ratios are well calibrated for the model (see Table 3.1), SWAT should be abstracting the correct amount of runoff and hence have accurate quantities of precipitation entering the soil profile. Furthermore, SWAT is designed to predict the impact of management on water and agricultural chemical yields for ungauged basins (Arnold and Allen, 1996). As the soil physics in SWAT are very similar to those in EPIC, the modeling approach of this study essentially uses SWAT as a tool to run and spatially compile ~700 EPIC models (one for each HRU). Additionally, though this approach does not take surface water N into account, a sensitivity analysis varying NPERCO (the SWAT variable used to calibrate surface water N, which ratios the runoff nitrate concentration to the soil nitrate concentration) found basin-wide nitrate leaching varied by less than 5% throughout NPERCO's range. Since nitrate leaching values are relative concentrations, the uncertainty caused by not including surface N calibration is minimal and should not significantly affect nitrate leaching outcomes and conclusions.

3.5.2. Spatial Distribution of Recharge: Present and Future

Figure 3.6a shows the average annual recharge for subbasins using the recent LULC map. The basin average recharge value is 460.9 mm, with an average of approximately 398 mm of recharge on the Willamette silts and 459 mm of recharge on the unconfined Willamette aquifer. SWAT simulated higher recharge for the geologic units in the Coast Range and the Coburg Hills, where higher precipitation occurs (however, a significant amount of this recharge could be lost to lateral flow after exiting shallow soils). The lowland recharge values fall between basin-scale recharge estimates derived by Lee and Risley (2002) (254 – 381 mm) and those of Woodward et al. (1998) (254 – 508 mm on Willamette silts, 508 - 762 mm on the unconfined Willamette aquifer). Examining Figure 3.6a, most subbasin scale recharge values change minimally with LULC change, with changes generally associated with differences in impervious surface areas. The Development scenario shows substantially more basin-wide decline than other future scenarios, largely because of greater road construction and urbanization. Subbasin 11 is the only subbasin which shows significant increases in recharge in future scenarios, which is related to declines in road area and decreases in river channel area. Channel area affects recharge because SWAT does not calculate recharge when water is the land cover.

The effects of GW-BMPs on present recharge at the subbasin scale are relatively small, with no subbasins changing by greater than 5%. Future GW-BMPs scenarios showed little relative change in recharge because irrigated agriculture is a minor land use at the subbasin scale for both present and future scenarios (see Table 3.2), indicating that land use change has greater effects than GW-BMPs on regional recharge.

3.5.3. Spatial Distribution of Nitrate Leaching: Present and Future

Nitrate leaching distribution, as shown in Figure 3.7a for the present, is similar to the observed distribution of groundwater samples. Higher leaching is both observed and modeled in areas near the Willamette river (where the Willamette aquifer is unconfined), corresponding to the region where high N-demanding irrigated crops are often grown. Regions where modeled values show lower leaching than expected appear to be influenced by model scale and parameterization. Notably, subbasins 34 and 35 were expected to have higher leaching values than SWAT determined. The primary cause for this discrepancy is that septic leachate is not modeled by SWAT, and subbasins 34 and 35 are in rural areas on septics that have higher population densities than elsewhere in the modeled area (see Appendix D). Notably, subbasin 35 includes Coburg, the largest community in Oregon (969 residents (US Census Bureau, 2006)) that does not have a wastewater treatment plant. Isotopic analyses of groundwater nitrate from both subbasins support that a sizeable septic influence exists (Vick, 2004).

Subbasin size is also believed to cause poor simulated-observed nitrate leaching relationships in both subbbasins 34 and 35. Subbasin 34's land use is 35% urban, while subbasin 35 is 19% urban, and both are associated with higher density road networks and residential areas of the Eugene area. Modeled nitrate leaching values for urban areas are generally lower than agricultural land areas, and hence when a subbasin has a high percentage of urban areas, lower modeled leaching values occur. Additionally, the Coburg area (an identified problem area with high density septics) is a relatively small region in subbasin 35, so even if septic loadings were modeled, the loading from Coburg could be counterbalanced by larger areas with lesser loading in the subbasin.

Though SWAT does not directly simulate septic loading from rural land uses, modeling septics by using higher soil organic N values for areas influenced by septics was considered. This modeling approach was used by Santhi et al., (2001) to simulate high manure loading for dairies in Texas. Reasons why this approach was not used for this study was because it would be difficult to accurately assign appropriate loadings for future scenarios having different rural development patterns, and because areas currently operating on septics near urban areas may switch to centralized sewage treatment in the future.

Future scenarios of nitrate leaching (Figure 3.8) show significant declines along the Willamette River corridor and are largely due to crop type changes from peppermint and row crop rotations to grass seed. Declines in other regions generally reflect a switch to crops with lower N demands or the conversion of cropland to natural areas (natural grassland, shrubs, or forest). Greater leaching declines are observed for the Development and Conservation scenarios than the Plan Trend scenario because the Plan Trend has 9 and 23% more farmland than the Development and Conservation scenarios and has 18 and 25% less natural area. The Conservation scenario has the greatest leaching declines because it has less agricultural land and more natural land than any other scenario (see Table 3.2). Subbasins with increases in nitrate leaching are generally caused by the spread of urban areas onto lands that were previously forested or natural. Only subbasin 37 has increased leaching associated with increased agricultural land area. Additionally, though several subbasins in the Coast Range and near Corvallis and Eugene have increases in leaching, their leaching loads remain relatively low.

The effects of GW-BMPs on subbasin nitrate leaching are readily apparent in Figures 3.7c and 3.8. Though GW-BMP N inputs were ~15% less than non-GW-BMP inputs, subbasins showed declines in leaching of ~28% (Table 3.3). This larger leaching decline is likely associated with greater cover crop use in GW-BMP scenarios, lower irrigation rates, and more efficient N uptake.

Figure 3.7c shows that for the present scenario, subbasins with the greatest decline after GW-BMP implementation would be those near the Willamette River and several in the Coast Range. These larger declines are associated with dominant crop types in the subbasin. Leaching declines near the Willamette River are related to irrigated-agriculture GW-BMPs (which reduce both irrigation and fertilization by 13% and make use of more cover crops). Leaching declines in the Coast Range subbasins, which are mostly composed of forest and some lowland tall fescue fields, reflect the proportionately greater fertilizer declines modeled for tall fescue fields (see Appendix F).

Comparing the present GW-BMP implementation scenario with future non-GW-BMP scenarios, it appears that a change in LULC alone would not be as effective at mitigating groundwater nitrate concerns as implementing GW-BMPs with the current cropping system (Table 3.3). Changes in LULC can, however, cause greater local leaching declines than basin-wide GW-BMP implementation for current crops (largely due to the decline in irrigated agriculture and increases in natural areas for specific subbasins). If projected shifts in LULC to near monoculture grass seed production occur along with GW-BMP adoption, agrarian leaching loads are projected to decline by between 32 and 46% of present values (see Table 3.3). As uncertainty exists in future markets and LULC distribution, present GW-BMP implementation is the safest course of action for lessening groundwater nitrate concentrations. Additionally, the Plan Trend 2050 scenario in Table 3.3 indicates that without extreme changes in development (as would be the case in the Development 2050 or Conservation 2050 scenarios), basin-wide leaching may decline by only 7% with only crop change and no GW-BMPs.

Though uncertainty is associated with markets and public policy, which will impact future LULC, the resurgence of large-scale high intensity agriculture to levels of the early 1990s in the SWV is unlikely. As the Haber-Bosch process, which is the process used in N fertilizer manufacturing, is highly dependent on natural gas (natural gas is ~90% of the ammonia production cost (Petroleum News, 2006), with 4 lbs of N fertilizer having the energy equivalent of ~1 gallon diesel fuel (Helsel, 1992)), fluctuations in gas and oil prices acutely impact growers of crops with high N demand. Given that future natural gas prices will increase due to the long-term decline of US reserves (EIA, 2006), profitable farming of crops with high N requirements will likely remain difficult. Additionally, the decline of infrastructure supporting row crop and peppermint harvest (no local canneries remain in business, large peppermint producers and their distilleries have been bought out by grass seed farms) would also make it less likely that row crops and peppermint will expand in acreage. Lastly, increased growing efficiencies and education are likely to lessen nitrate-leaching impacts of high intensity agriculture if resurgence occurs.

3.5.4. Examination of Temporal Nitrate Leaching Dynamics

Temporal nitrate leaching variability in the SWV has been documented by Feaga et al. (2004) and observed in groundwater nitrate concentrations (see Chapter 2). An observed association between periods of recharge and higher groundwater nitrate concentrations indicates that temporal groundwater nitrate concentrations are related to periods of high leaching in the SWV (see Chapter 2). Figure 3.9 compares median monthly groundwater nitrate concentrations from the well network in Chapter 2 to the basin-averaged temporal nitrate leaching mass of SWAT. After November, when both data sets indicate seasonal recharge begins, leaching values from SWAT reasonably mimic those of groundwater nitrate. Observed groundwater nitrate variability is substantially less than vadose leachate variability, which is likely explained by the dilution of leachate when entering the aquifer. During recharge periods, inconsistencies between the two data sets could be due to differences in precipitation between sampling periods. Groundwater nitrate values were influenced by precipitation between sampling periods, while SWAT leaching values were calculated for calendar months. Notably, SWAT shows lower leaching in April than December, while sample data shows concentrations being slightly higher in April. It is believed that SWAT's underprediction of April nitrate leaching is due to it having significantly less precipitation occur than was observed between sampling events. Interestingly, SWAT shows higher leaching during April than March or May, both of which are months that had higher precipitation than April. This can be explained by examining SWAT's monthly recharge values, which indicate that much of the precipitation falling in March goes into soil storage, effectively filling the soil profile and allowing for the April leachate peak. Additionally, as SWAT simulated April as being relatively dry, proportionately more of May's precipitation was needed to fill the soil profile, causing there to be less leaching than in April.

Other data used to examine the validity of SWAT's temporal percolation and leaching values include average observed monthly flow-weighted leaching values, which were compared to SWAT-derived averages. Trends in SWAT's flow-weighted leaching are similar to data presented by Feaga et al. (2004) for the Willamette Valley, as shown in Figure 3.10. Notably, both show relative high leachate concentrations during the late spring and summer. Both data sets also exhibit seasonal flushing, where concentrations decline substantially due to the dilution and movement of soil water nitrate with the onset of fall and winter rains. The amplitude of seasonal concentration fluctuations for SWAT data is considerably less than the average observed values. This is likely because SWAT values are for the entire modeled area (including urban and forested areas), while the observed data are from the highest leaching crops within the SWV.

Annual leaching variability was found to largely be dependent on annual precipitation values, with months having higher precipitation generally having higher leaching (especially in mid-winter, when the soil profile is commonly close to field capacity), as shown in Figure 3.11a. This increased leaching response to precipitation was observed in wells during recharge months in the winter of 2004-2005 (see Chapter 2). These modeled data lend support to the hypothesized linear relationship between precipitation and nitrate leaching during recharge months (see Figure 3.11b). Additionally, modeled values in Figure 3.11a suggest that the data presented in Chapter 2 for the winter of 2004-2005 may have a lower seasonal response signal than other years.

3.6. Summary and Conclusions

SWAT was used in this study to examine spatial and temporal leaching of nitrate from the soil profile. Though this study applied SWAT in an atypical way, spatial and temporal leaching outputs were qualitatively validated with existing groundwater and soil water nitrate data, and thus the model is believed to represent regional nitrate leaching processes well. Drawbacks of the nitrate leaching model are that it does not include denitrification and that N loadings from septic tanks could not modeled, therefore enabling only agricultural GW-BMPs to be evaluated. Additionally, as a regional scale non-point source nitrate model, localized leaching issues could be overlooked because of their relatively small scale. To overcome scaling issues, the current model could be refined to use smaller subbasins so that localized leaching hotspots could more readily be examined.

After validation of the initial SWAT model using recent LULC inputs, changes in nitrate leaching were examined for alternative futures exhibiting different LULC distributions. Additionally, GW-BMPs for nitrate and their relative effect on leaching were examined for future and present scenarios (however both GW-BMP scenarios and non-GW-BMP scenarios do not include septic leaching impacts). Assessment of GW-BMP and non-GW-BMP alternative futures relative to present land use practices found that a basin-wide GW-BMP implementation with present cropping is expected to result in larger nitrate leaching declines than future LULC change alone. Future shifts in LULC and the use of GW-BMPs were found to reduce basin-wide nitrate leaching by approximately 32 – 46% of present values.

Improvements that could be made to SWAT to enhance its future use as a non-point source nitrate leaching model include correcting soil denitrification kinetics, including a rural residential land use class with septic loading, and calculating subbasin nitrate leaching in the subbasin output file. Considerable time and effort is required to calculate nitrate leaching from the HRU output file, where it is presently printed (for this study, a filtering program was written to quickly sort and sum nitrate leaching data).

Temporal and Spatial Variability of Groundwater Nitrate in the Southern Willamette Valley of Oregon

4. Study Conclusions

Data collected in this study indicate that substantial temporal and spatial variability exists in groundwater nitrate for the SWV. Generally, wells were found to increase in groundwater nitrate concentrations during wet winter months. A hypothesis put forth to explain this trend is that since vadose-zone nitrate concentrations are generally higher than those observed in groundwater for the SWV, the mobilization of high concentration soil water during the winter rains causes an increase in shallow groundwater nitrate concentrations.

High intra-well seasonal variabilities were observed in the SWV, especially in wells with higher nitrate concentrations. However, intra-well variabilities were not found to significantly impact network-wide seasonality. Heterogeneities present in the SWV, including land use, soils, vadose transport properties, aquifer mixing, and well depth, are believed to be largely accountable for the muted network-wide seasonal response. Notably, wells were found to respond to seasonal precipitation at different time scales, causing their responses to be out of phase with one another. Implications of these findings on long-term trend analyses for groundwater nitrate monitoring networks is that even in locales with high intra-well variability, network-wide seasonal effects can be expected to be minimal (compared to individual wells). Additionally, quarterly sampling should be sufficient to determine seasonality (as opposed to monthly data) if the sampling network is spatially representative. A recommended sampling approach is to collect quarterly baseline data with a small subset of wells sampled monthly (so that local intra-well variabilities and the seasonal impacts of lumped monthly data can be examined).

Modeling of spatial nitrate leaching using SWAT found reasonable agreement between modeled spatial leaching indices and observed groundwater nitrate concentrations. Alternative future scenarios with land use/land cover (LULC) change for 2050 found that basin-wide nitrate leaching is expected to decline by between 7 to 24%

of present leaching values, with local changes being substantially greater. Local change was found to be greatest near the Willamette river (where the Willamette aquifer is unconfined), largely due to the projected decline in importance of high intensity irrigated agriculture crops. In the most probable alternative future, Plan Trend 2050 (the Development and Conservation 2050 futures were intended to bracket plausible futures), basin-wide leaching is projected to decline by only 7%.

Analyses of GW-BMPs for present and future scenarios indicates that GW-BMP implementation for the SWV would be expected to cause substantially greater basin-wide declines in nitrate leaching than LULC change alone (GW-BMPs cause leaching to decline by ~28% of present simulated values). Future scenarios with GW-BMPs applied indicate that basin-wide leaching could decline by between 32 and 46% of present leaching values. All future and GW-BMP scenarios consider agrarian N loading, but do not consider the effects of septic loading or denitrification, and thus should be viewed with caution. The most conservative course of action to alleviate groundwater nitrate contamination is to encourage GW-BMP implementation, as future development trends are uncertain and modeled values indicate greater leaching declines are likely to be associated with GW-BMPs.

Temporal nitrate leaching data from SWAT was found to adequately simulate temporal data from this study and other temporal studies. Coupling modeled and field data together, further interpretations can be drawn and a greater understanding of SWV leaching and recharge processes is possible. Notably, modeled data further supports the hypothesis suggested in Chapter 2 (based on field data) that a linear relationship could exist between precipitation and groundwater nitrate concentrations, as SWAT shows greater nitrate export to occur in wetter months. Furthermore, SWAT data implies that recharge was likely occurring in months identified as "non-recharge months" in Chapter 2, but possibly not in substantial enough quantities to cause groundwater levels to rise (net aquifer recharge was less than net aquifer discharge). This would be expected to impact intra-well variability and further impact phase relations between wells. Lastly, modeled data indicates that the intra-well seasonal fluctuations observed in the field data collected could be lower than during average or high precipitation years, as the total nitrate mass leached for winter 2004-2005 was lower than most years.

Future studies that could improve or capitalize on this work include the following: 1) Wells sampled in this study should continue to be monitored because a valuable baseline data is available for these wells and to determine if this study's results would differ were data collected in a wetter winter. Additionally, more field data could determine the validity of the linear increase observed between median monthly groundwater nitrate values and precipitation during recharge months. 2) The SWAT model created could be improved with smaller a subbasin size so that localized leaching occurrences are more likely to be observed and identified. The addition of septic loading would also enhance the model's spatial validity. Additionally, altering the SWAT source code to reflect how other models calculate denitrification would greatly enhance the current SWV SWAT model. A good model for consideration would be EPIC, since much of the nitrate processing in SWAT is based on the EPIC code. 3) The nitrate loading output from SWAT could be paired with a groundwater flow model for the SWV to determine aquifer fate and transport of groundwater nitrate. Currently, a calibrated steady-state MODFLOW model exists for the SWV (Craner, 2006), and the linkage of the two models would create a powerful tool for assessing likely aquifer remedial times.

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APPENDICES

Appendix A

Well Information and
Site Specific Temporal Groundwater Chemistry Parameters

The following pages include summaries of well information and monthly field parameter values for all wells sampled in this study. Sampling criterion applied to all wells are as follows: well log exists, well depth 50 ft or shallower, screening interval 15 ft or less, drilling date within the last 30 years, and the well must pass coliform bacteria test on the initial sampling date.

General notes for Appendix A:

- Several well owners did not consent to having their personal information divulged, and thus their name and addresses are not included in this appendix.
- Sample sites not meeting all of the criteria outlined above (9 and 11) have explanations given on their summary sheets.
- Two geologic units are given for each well (from O'Connor et al. (2001) and Gannett and Caldwell (1998)) because of different classification systems used for each study. The O'Connor et al. (2001) study generally has greater detail and subdivides the SWV into several different geologic units, while Gannett and Caldwell (1998) lump the Quaternary geologic units in the SWV. The Gannett and Caldwell map also has a greater regional area for the Willamette silt because they map it in areas where thin deposits exist, while O'Connor et al. (2001) required the silt to obscure the original geomorphic features underlying it and to have soils corresponding to those typical of flood deposits.
- Soil data is presented from two different data sets, one being at the county level (which has greater spatial resolution and more accurate unit identification) and the other at the state level (which generalizes units). Boundaries and values from the state soil survey were used in the SWAT model.
- Spatial coordinates are given in Northing and Easting values for UTM zone 10 N.
- Nitrate data for some wells include the month of July 2004. The sampling network used in this study was set up in July and August of 2004, so data from July 2004 is not discussed or presented elsewhere in the study because it is not representative of the entire network.

- In early sampling months, available sampling equipment for field parameters varied, and therefore early field parameter data is not presented because of lower data quality. Early field parameters collected but not presented include specific conductivity in July 2004, temperature and total dissolved solids for August 2004, dissolved oxygen in September 2004, and specific conductivity in October 2004. These measurements are considered valid for stabilization criteria (because the values observed for stabilization can be interpreted relative to one another), but not for data analysis.
- Error bars on the nitrate data are for 95% confidence intervals. Groundwater nitrate seasonality was assessed from these plots, with seasonal wells being defined as those which have monthly concentrations where error bars do not overlap and which seem to be influenced by precipitation (either having peaks in high precipitation months or generally showing higher nitrate concentrations during wetter periods).

Site ID:	1	**OWRD Well Log:**	Lane 2251
Owner:	Howard, Kyle		
Address:	91876 North Coburg Rd., Eugene, OR 97408		

Drilling Date:	1991	**Geology (Gannett and Caldwell, 1998):**	Qs
Depth (ft):	39	**Geology (O'Connor et al., 2001):**	Qg2
Screening Interval (ft):	none	**Easting:**	494061
State Soil Survey:	Dayton-Amity-Aloha	**Northing:**	4889642
County Soil Survey:	Salem Gravelly Silt Loam		

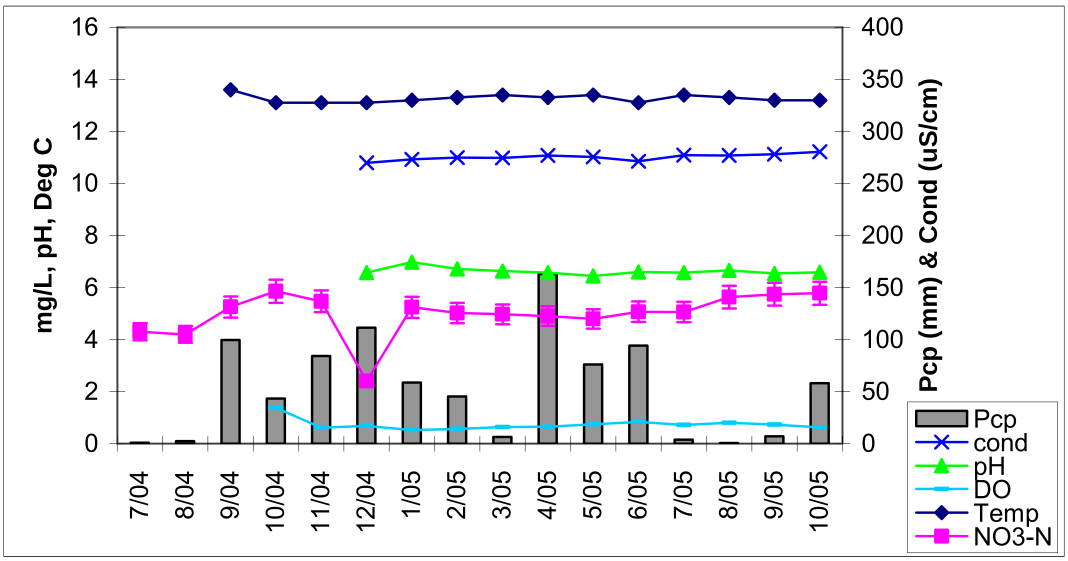


Figure A1: Well 1 trends and interpretations.

Nitrate concentrations in well 1 displayed little variability after September 2004 (excluding December, which may be an outlier). The lack of observable seasonality is believed to be largely due to denitrification, which is suggested by the low dissolved oxygen (DO) concentrations. Trends in groundwater nitrate and DO are generally similar as both show minimal seasonal variability.

Nitrate concentrations in well 1 are higher than other wells (wells 12 and 13) with mean DO concentrations < 1 mg/L. This is likely due to its close proximity (~200m) to a large confined animal feeding operation (CAFO) with ~800 dairy cows. Additionally, a dense rural housing division (on septic) is ~300m up gradient of well 1. Therefore, denitrification is likely occurring and lessening local nitrate levels, but could be rate-limited and unable to lower nitrate concentrations further.

Site ID:	2	**OWRD Well Log:**	Lane 729
Owner:	Halverson, Lloyd		
Address:	31591 Coburg Bottom Loop Rd Coburg, OR 97408		

Drilling Date:	1978	**Geology (Gannett and Caldwell, 1998):**	Qal
Depth (ft):	50	**Geology (O'Connor et al., 2001):**	Qalc
Screening Interval (ft):	40-48	**Easting:**	492263
State Soil Survey:	Newberg-Chehalis-Cloquato	**Northing:**	4887466
County Soil Survey:	Camas Gravelly Sandy Loam		

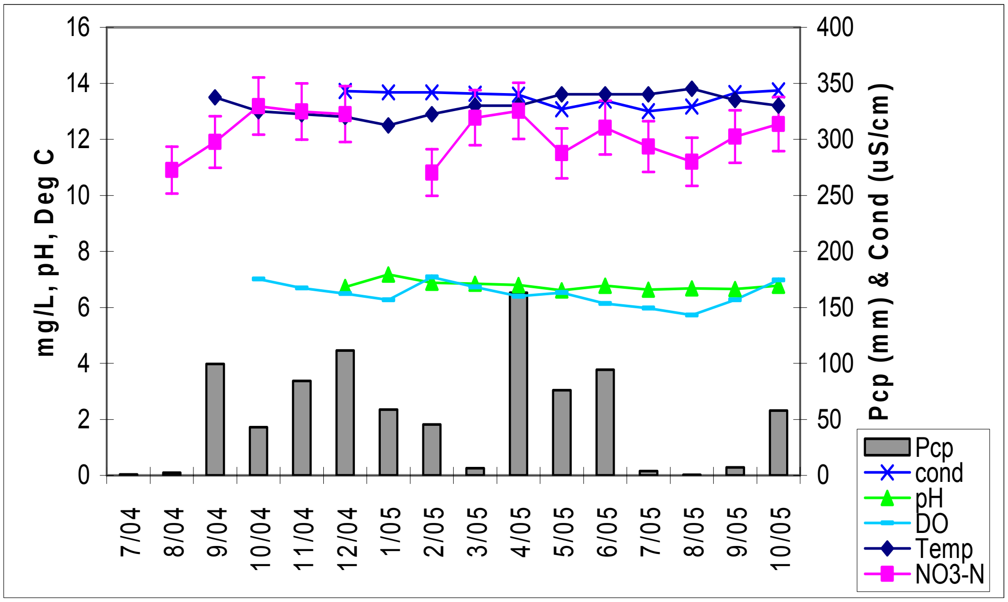


Figure A2: Well 2 trends and interpretations.
Well 2 shows a seasonal relationship between nitrate and precipitation during wetter months. High nitrate concentrations are believed to be associated with the high septic densities of the Coburg area and intensive farming practices.

Site ID:	3	**OWRD Well Log:**	Lane 62886
Owner:	Oregon Department of Enviromental Quality		
Address:	Green Island traffic median, N end of Coburg Loop Rd.		

Drilling Date:	2003	**Geology (Gannett and Caldwell, 1998):**	Qal
Depth (ft):	24	**Geology (O'Connor et al., 2001):**	Qalc
Screening Interval (ft):	20-23	**Easting:**	492263
State Soil Survey:	Malabon-Coburg-Salem	**Northing:**	4888507
County Soil Survey:	Newburg Loam		

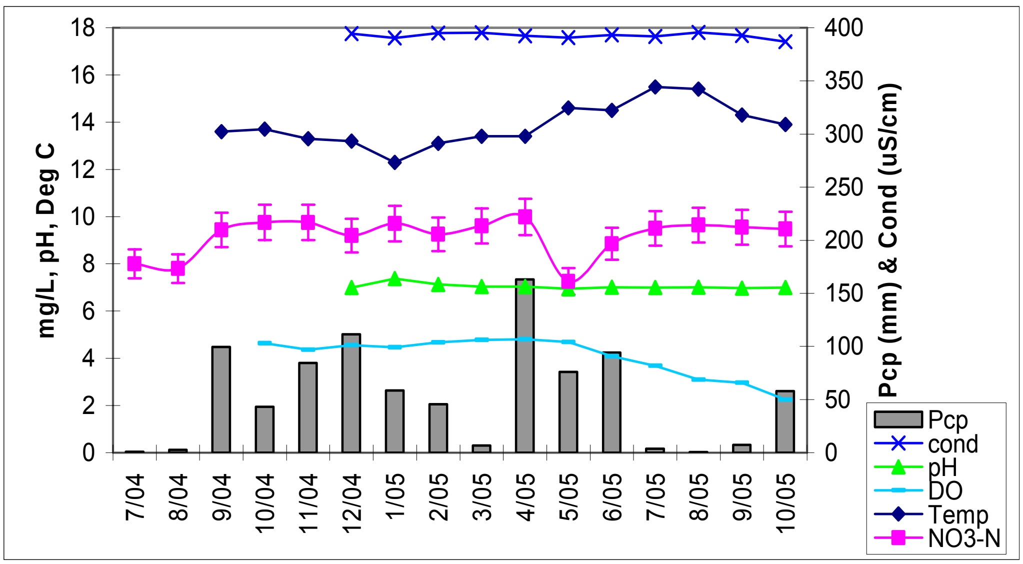


Figure A3: Well 3 trends and interpretations.

Well 3 has relatively high nitrate concentrations, which are believed to be associated with the high septic density and intensive farming of the Coburg area. Seasonality in groundwater nitrate was observed in well 3, with the highest observed nitrate concentration in the wettest month of the study. Relatively large temperature fluctuations are believed to be associated with its shallow depth. Other monitoring wells (6, 12, and 16) show similar trends in temperature, with higher values during summer months.

Site ID:	4	**OWRD Well Log:**	Lane 7904
Owner:	Younger, Robert and Robin		
Address:	30918 Crossroads Ln		

Drilling Date:	1977	**Geology (Gannett and Caldwell, 1998):**	Qal
Depth (ft):	23	**Geology (O'Connor et al., 2001):**	Qalc
Screening Interval (ft):	none	**Easting:**	489235
State Soil Survey:	Newberg-Chehalis-Cloquato	**Northing:**	4892386
County Soil Survey:	Camas Gravelly Sandy Loam		

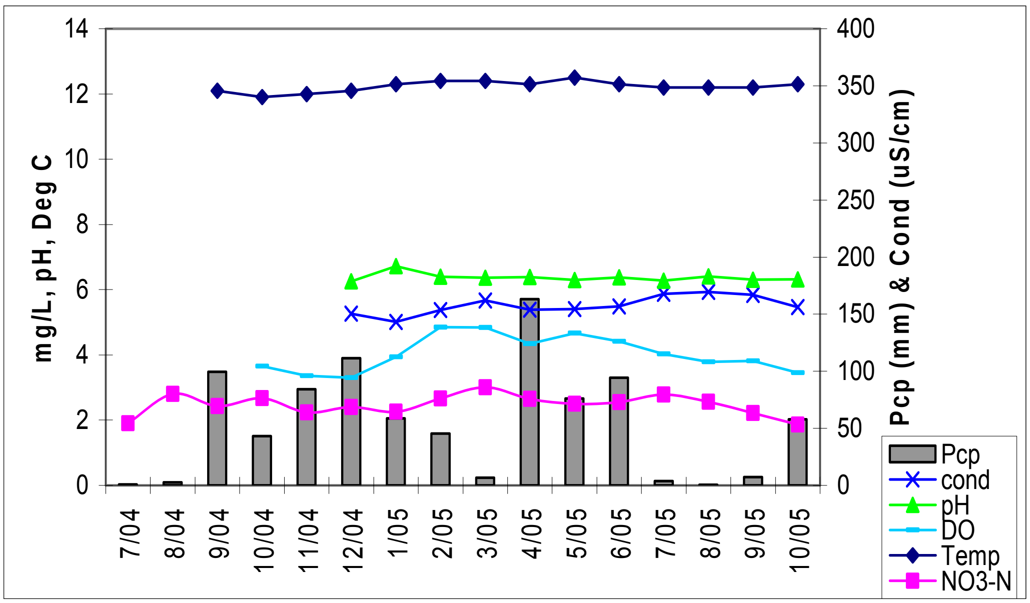


Figure A4: Well 4 trends and interpretations.
Well 4 has low groundwater nitrate concentrations, with no discernable seasonality observed. The low concentrations are believed to be associated with lower intensity cropping practices (grass seed, orchards, and Christmas trees) and low housing density. Additionally, hyporheic interactions with the Willamette River (500 m away) may occur during the winter, which could cause nitrate seasonality to be non-existent. Hyporheic interactions were observed at similar distances from the Willamette River by Hinkle et al. (2001). Low conductivity values for the area could also suggest younger groundwater which could be influenced by the Willamette River.

Site ID:	5	**OWRD Well Log:**	Lane 60805
Owner:	Bedacht, Manfred		
Address:	30204 Heather Oak Dr., Junction City, OR 97448		

Drilling Date:	2002	**Geology (Gannett and Caldwell, 1998):**	Qs
Depth (ft):	38	**Geology (O'Connor et al., 2001):**	Qg1
Screening Interval (ft):	none	**Easting:**	487058
State Soil Survey:	Malabon-Coburg-Salem	**Northing:**	4890115
County Soil Survey:	Chapman Loam		

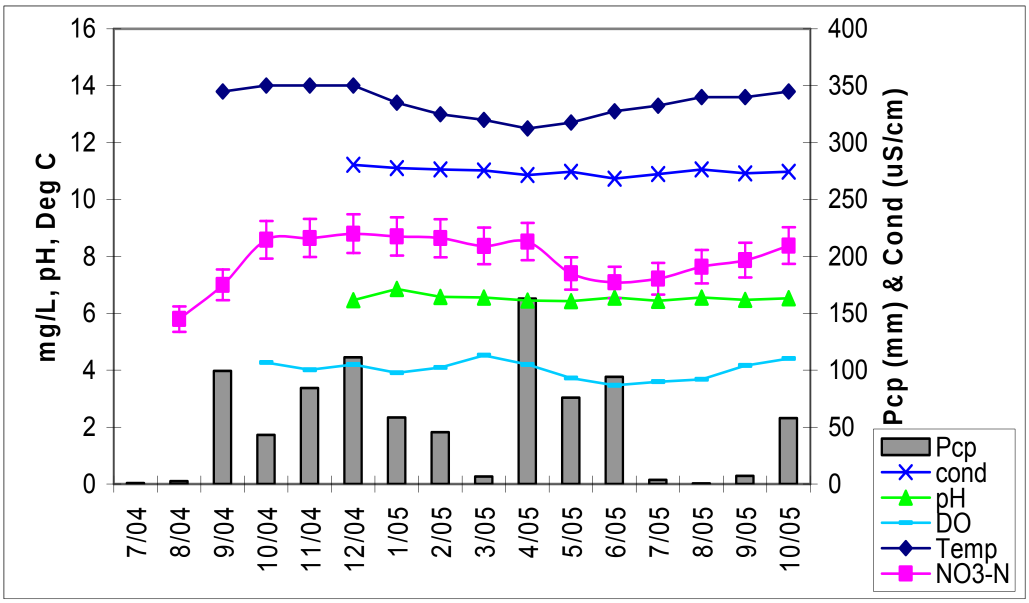


Figure A5: Well 5 trends and interpretations.
Well 5 displays seasonal nitrate trends which appear to reflect wetter and drier periods more than monthly precipitation patterns. Higher nitrate values are believed to be associated with higher septic densities and agriculture.

Site ID:	6	**OWRD Well Log:**	Lane 62885
Owner:	Oregon Department of Enviromental Quality		
Address:	Washburne Wayside		

Drilling Date:	2003	**Geology (Gannett and Caldwell, 1998):**	Qal
Depth (ft):	24	**Geology (O'Connor et al., 2001):**	Qg1
Screening Interval (ft):	20-23	**Easting:**	480624
State Soil Survey:	Malabon-Coburg-Salem	**Northing:**	4903079
County Soil Survey:	Salem Gravelly Silt Loam		

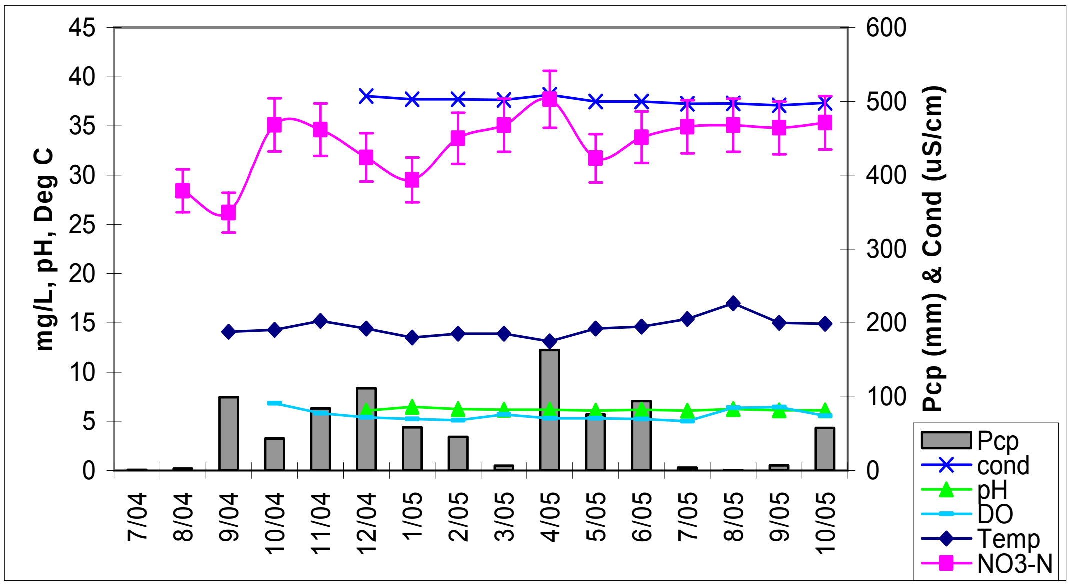


Figure A6: Well 6 trends and interpretations.
Well 6 has extremely high nitrate and conductivity values, which are believed to be associated with point source pollution (~1100 cows are in a CAFO 500m up gradient from the well), while all other wells are thought to be predominantly nonpoint source influenced. Large seasonal fluctuations were observed in well 6, with the highest observed concentration occurring in April.

Site ID:	7	**OWRD Well Log:**	Lane 1984
Owner:	Fisher, Don		
Address:	93735 Strome Ln Junction City, OR 97448		

Drilling Date:	1991	**Geology (Gannett and Caldwell, 1998):**	Qs
Depth (ft):	49	**Geology (O'Connor et al., 2001):**	Qg1
Screening Interval (ft):	32-38	**Easting:**	484882
State Soil Survey:	Malabon-Coburg-Salem	**Northing:**	4894847
County Soil Survey:	Malabon Silty Clay Loam		

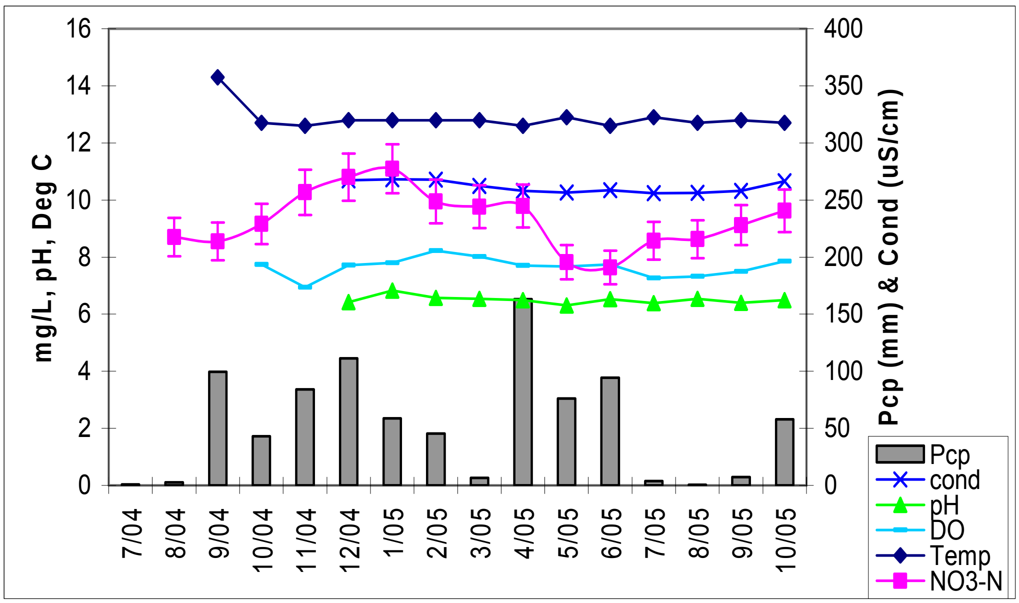


Figure A7: Well 7 trends and interpretations.
Well 7 has higher nitrate concentrations during recharge months, specifically November through January. Nitrogen sources are likely to be a mix of leachate from high intensity agricultural fields and septics up gradient of the well.

Site ID:	8	**OWRD Well Log:**	Lane 6370
Owner:	Parker, Jean (initially, new resident moved in at end of study)		
Address:	1700 Deal St, Junction City, OR 97448		

Drilling Date:	1988	**Geology (Gannett and Caldwell, 1998):**	Qs
Depth (ft):	27	**Geology (O'Connor et al., 2001):**	Qg1
Screening Interval (ft):	22-27	**Easting:**	484125
State Soil Survey:	Malabon-Coburg-Salem	**Northing:**	4897212
County Soil Survey:	Malabon Silty Clay Loam		

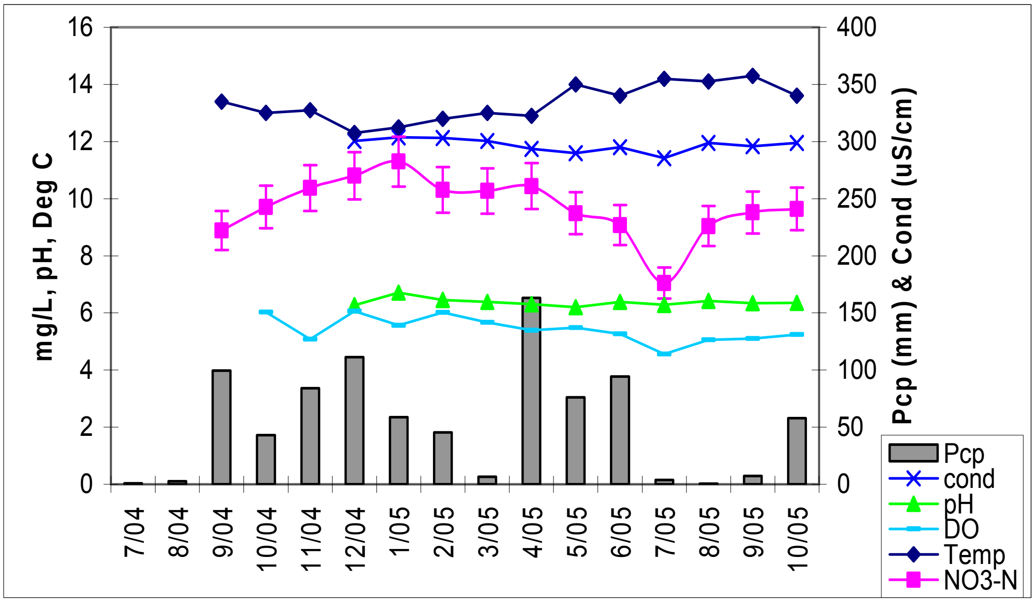


Figure A8: Well 8 trends and interpretations.
Well 8 shows increases in nitrate concentrations during the November through January recharge period, with declines occurring after April. High nitrate concentrations are likely associated with high intensity agriculture and septics upgradient of the well.

Site ID:	9	**OWRD Well Log:**	Lane 670
Owner:	Stewart, Tom		
Address:	28179 High Pass Rd., Junction City, OR 97448		

Drilling Date:	1988	**Geology (Gannett and Caldwell, 1998):**	Qs
Depth (ft):	45	**Geology (O'Connor et al., 2001):**	Qg2
Screening Interval (ft):	25-44*	**Easting:**	481097
State Soil Survey:	Dayton-Amity-Aloha	**Northing:**	4895698
County Soil Survey:	Malabon Silty Clay Loam		

*Well 9 has a total screening interval of 19 ft, which exceeds the maximum screening interval (15 ft) criteria for sampling wells. An attempt was made to find a different well in the same region that met all sampling criteria. Permission was not granted for any other well that did, but it was determined that data from the region was important enough to allow for continued sampling from well 9.

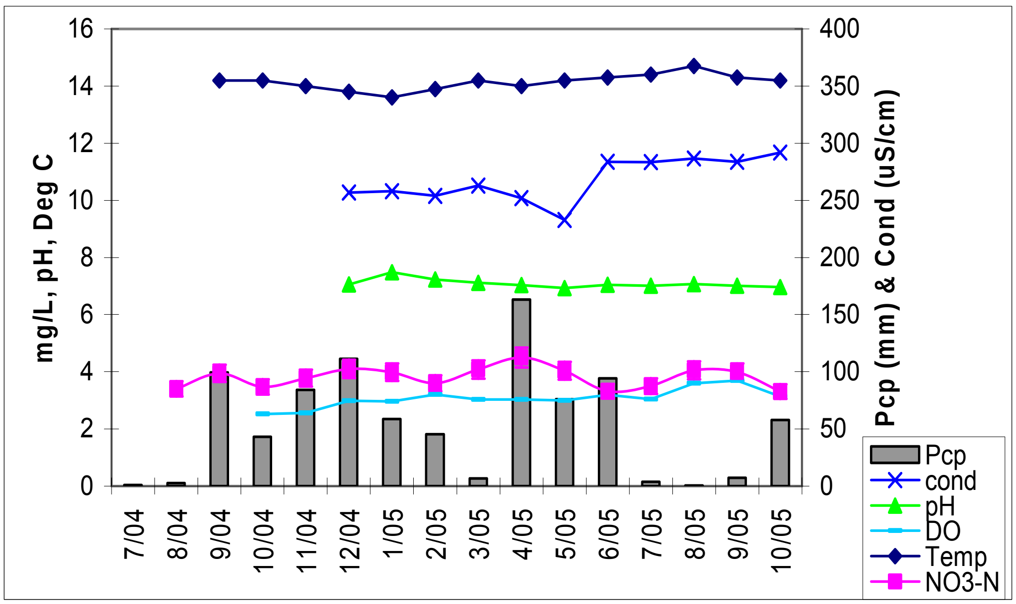


Figure A9: Well 9 trends and interpretations.
Well 9 has low-amplitude seasonal nitrate fluctuations, with peaks generally associated with high rainfall months. The lower nitrate concentrations observed are likely associated with lower local housing densities and grass-seed dominated agriculture. The large jump in conductivity in June is believed to be caused by the initiation of irrigation, which could possibly have lead to the capture of groundwater with higher conductivity values (Well 9 is a domestic well that is used only in the summer for lawn irrigation).

Site ID:	10	**OWRD Well Log:**	Linn 50250
Owner:	x		
Address:	x		

Drilling Date:	1996	**Geology (Gannett and Caldwell, 1998):**	Qs
Depth (ft):	47	**Geology (O'Connor et al., 2001):**	Qff2
Screening Interval (ft):	34-43	**Easting:**	485923
State Soil Survey:	Dayton-Amity-Aloha	**Northing:**	4907621
County Soil Survey:	Amity Silt Loam		

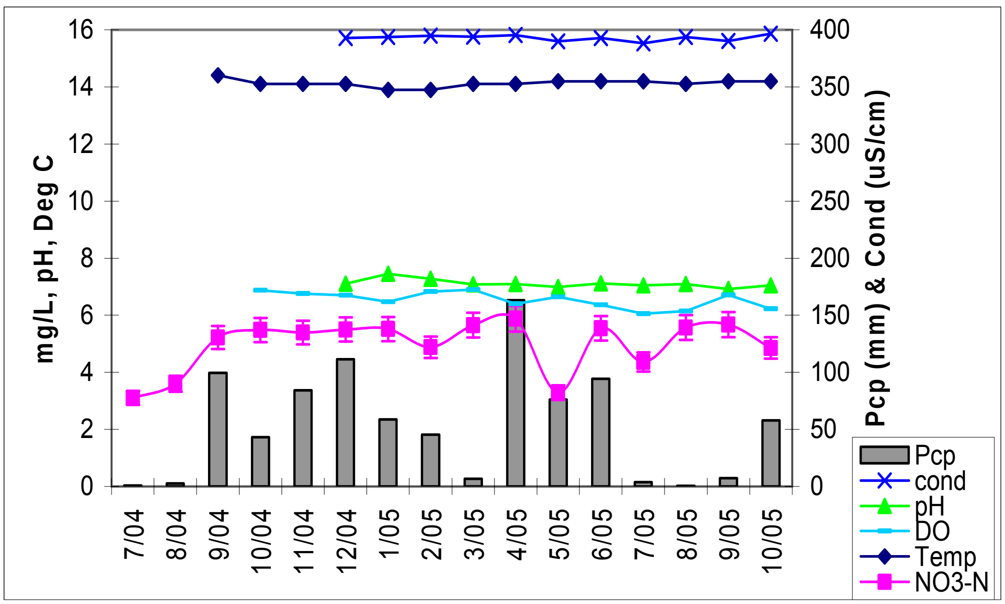


Figure A10: Well 10 trends and interpretations.
Well 10 displays seasonal nitrate variability, with increases between summer and fall/winter months and a peak during the highest precipitation month. Most wells sampled had unexplainable non-recharge month groundwater nitrate variability, but wells 10, 15, and 18 all show similar non-recharge month fluctuations (with a high April, low May, high June, and low July). One possible explanation for this could be that these wells experienced recharge in April and June (causing high concentrations of groundwater nitrate) and no recharge in May and July (when dilution of high-concentration shallow groundwater could occur).
The mid to low-level nitrate concentrations observed are likely to be impacted by grass-seed agriculture. Relatively high conductivity values observed in wells 10, 11, and 15 are believed to be associated with the aquifer being confined by the Willamette silt of O'Connor et al. (2001).

Site ID:	11	**OWRD Well Log:**	*
Owner:	Branson		
Address:	29702 Nicewood		

Drilling Date:	*	**Geology (Gannett and Caldwell, 1998):**	Qs
Depth (ft):	*	**Geology (O'Connor et al., 2001):**	Qff2
Screening Interval (ft):	*	**Easting:**	484976
State Soil Survey:	Dayton-Amity-Aloha	**Northing:**	4914813
County Soil Survey:	Woodburn Silt Loam		

*Well 11 was initially IDed as a well with an existing log. Further investigation 8 months into the study indicated that the well was improperly IDed and no well log exists.

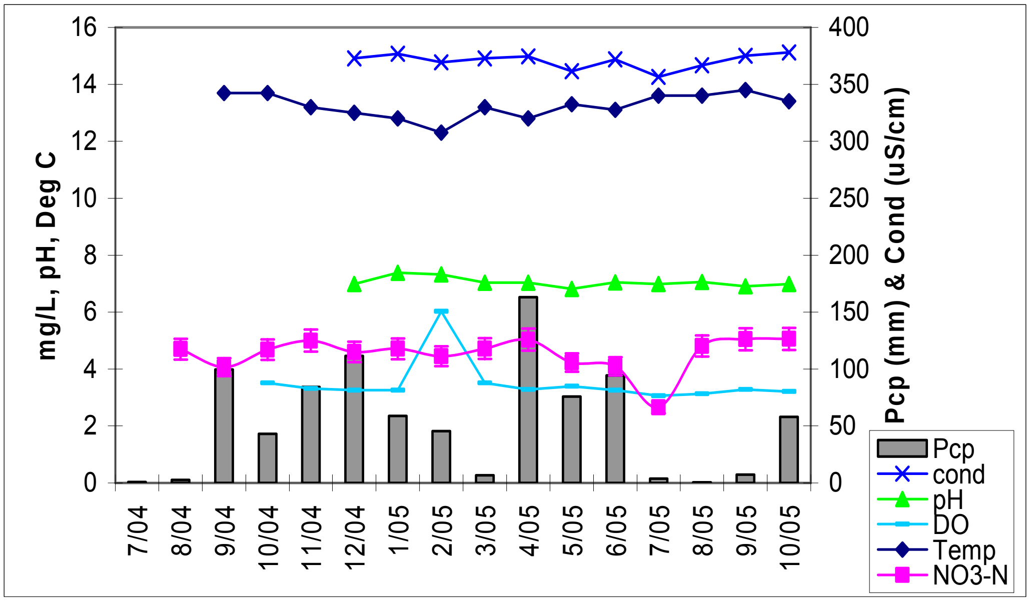


Figure A11: Well 11 trends and interpretations.
Well 11 displayed low-amplitude seasonal nitrate variability, with peaks occurring during recharge months. Relatively low nitrate values are likely tied to grass-seed dominated agriculture.
Relatively high conductivity values observed in wells 10, 11, and 15 are believed to be associated with the aquifer being confined by the Willamette silt of O'Connor et al. (2001).

Site ID:	12	**OWRD Well Log:**	Benton 52471
Owner:	Oregon Department of Enviromental Quality		
Address:	28894 Hulbert Rd., Seed Research		

Drilling Date:	2003	**Geology (Gannett and Caldwell, 1998):**	Qal
Depth (ft):	28	**Geology (O'Connor et al., 2001):**	Qalc
Screening Interval (ft):	24.4-27.4	**Easting:**	480813
State Soil Survey:	Newberg-Chehalis-Cloquato	**Northing:**	4920112
County Soil Survey:	Chehalis Silty Clay Loam		

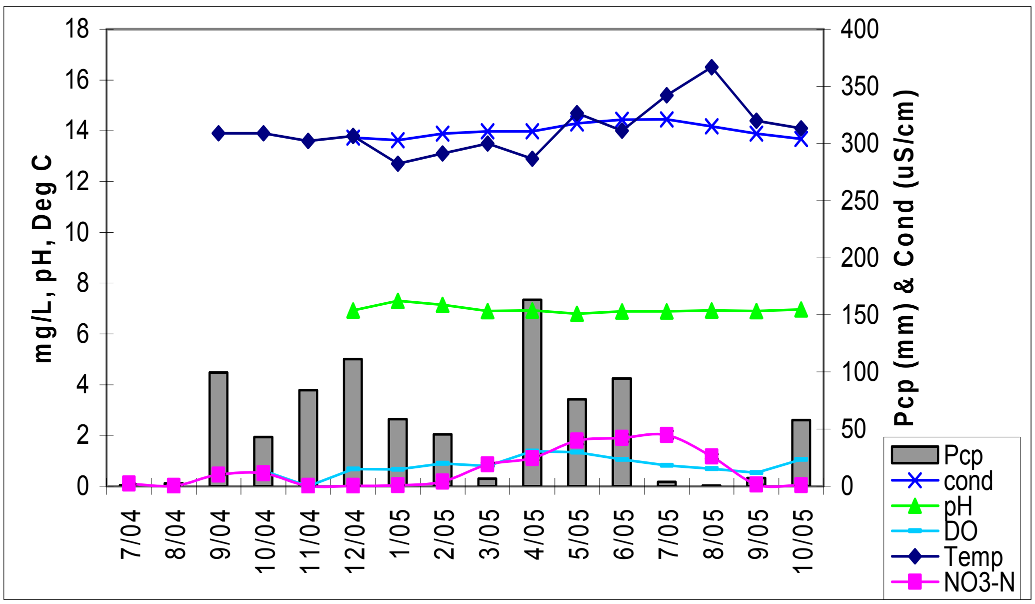
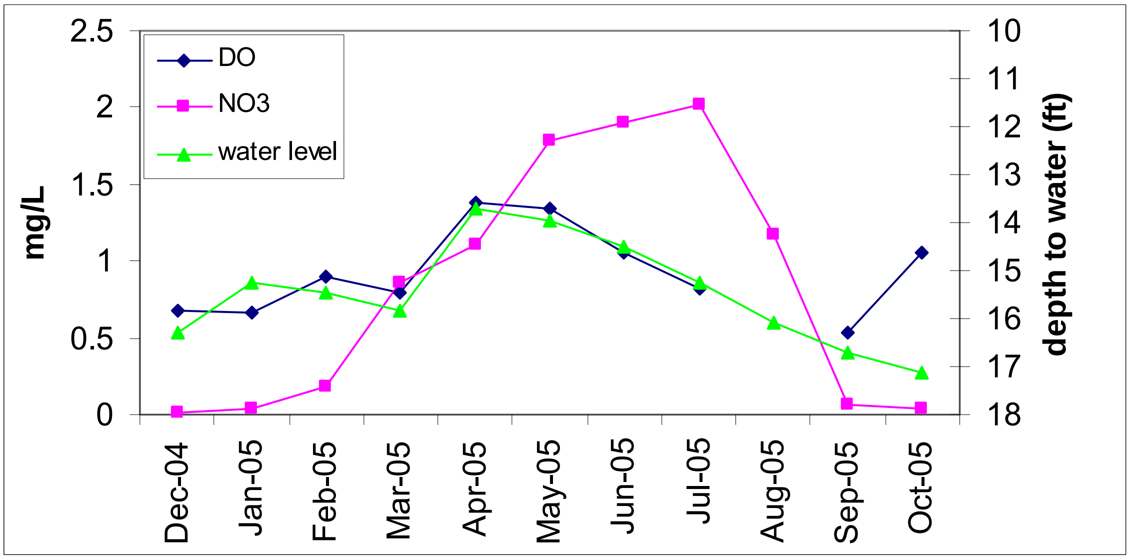


Figure A12: Well 12 trends and interpretations.

Well 12 has low nitrate concentrations and low dissolved oxygen values, implying that denitrification is locally occurring in the groundwater. The well log for well 12 indicates that a redox boundary is passed, with soil color changing from a dark brown red to a blue-gray color 18 ft below the surface. Since iron requires greater reducing conditions than nitrate for it to be used as an electron acceptor, it is expected that minimal quantities of nitrate would be found in the well.

A relationship between water level, dissolved oxygen (DO), and nitrate concentrations was observed in well 12 (see figure below). Notably, when groundwater levels rose, DO concentrations increased in the shallow aquifer, followed by increases in groundwater nitrate concentrations. Similar trends between DO and nitrate concentrations were observed in wells 1, 4, 16, and 17. However, well 16, the only well with a DO-NO_3^- trend and data for groundwater levels, does not have corresponding increases in DO during high water-level months, indicating that the correlation between water level and DO exists for well 12 and possibly no others.

A correlation between DO and nitrate concentrations was found network wide (excluding site 6) and is presented in Appendix C. It is not clear why the relationship between nitrate and DO does not break down at higher DO concentrations. Additionally, wells 4, 16, and 17 have aerobic DO conditions, but appear to have groundwater nitrate trends correlated to temporal DO trends.



Site ID:	13	**OWRD Well Log:**	Bent 51993
Owner:	x		
Address:	x		

Drilling Date:	2002	**Geology (Gannett and Caldwell, 1998):**	Qs
Depth (ft):	36	**Geology (O'Connor et al., 2001):**	Qg1
Screening Interval (ft):	none	**Easting:**	477312
State Soil Survey:	Malabon-Coburg-Salem	**Northing:**	4915286
County Soil Survey:	Coburg Silty Clay Loam		

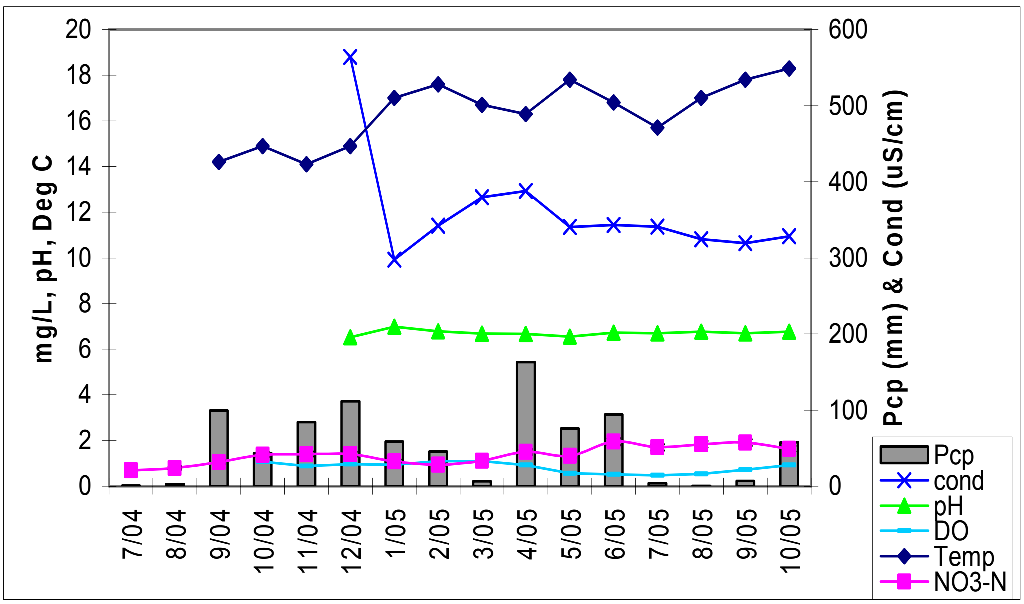


Figure A13: Well 13 trends and interpretations.

Well 13 had no discernable seasonality in groundwater nitrate. Low DO concentrations were observed, as well as iron rich (orange-brown) early-purge water. It is likely that iron reduction is occurring in the shallow groundwater near well 13, and therefore denitrification also is occurring.

Groundwater temperature values were found to be extremely high in well 13, and conductivity displayed considerable variability. A possible explanation for these observations would be a leaky petroleum storage tank (well 13 is at a service station), which is inferred by the iron dissolution and the black sediments noted on the well log. A sufficiently large bacterial colony could possibly cause high groundwater temperatures.

Site ID:	14	**OWRD Well Log:**	Bent 5306
Owner:	Steuwe, Vern and Gladys		
Address:	2296 SE Kiger Island		

Drilling Date:	1986	**Geology (Gannett and Caldwell, 1998):**	Qal
Depth (ft):	33	**Geology (O'Connor et al., 2001):**	Qalc
Screening Interval (ft):	23-31	**Easting:**	481191
State Soil Survey:	Newberg-Chehalis-Cloquato	**Northing:**	4929290
County Soil Survey:	Cloquato Silt Loam		

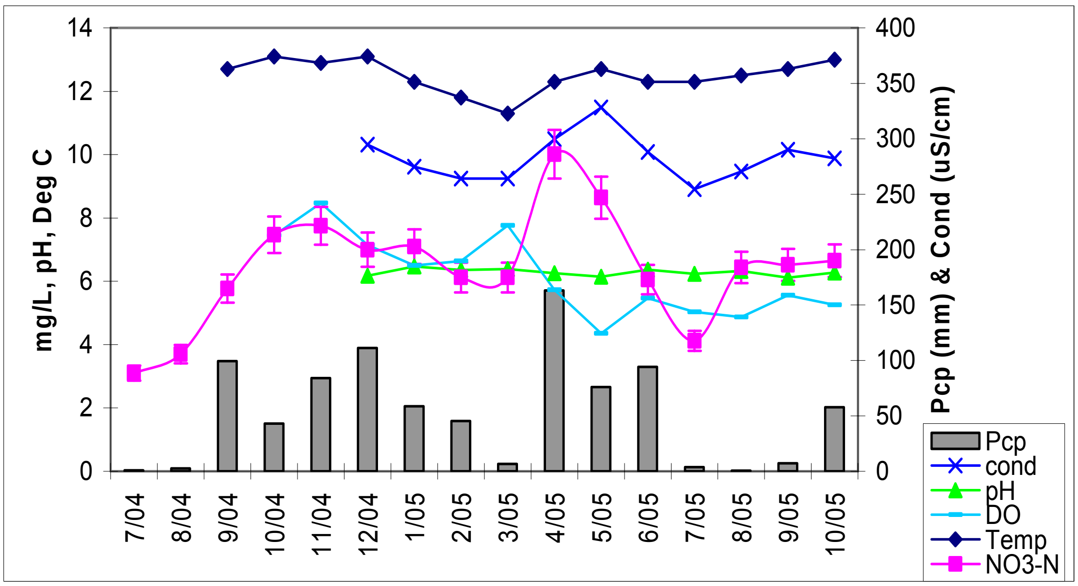


Figure A14: Well 14 trends and interpretations.
Well 14 has extremely high seasonal nitrate variability in addition to high variability in conductivity and DO. High nitrate concentrations are thought to be associated with high local septic density, row crops, and possibly an abandoned CAFO (this CAFO has at least 1 uncovered manure pile, but is thought to be ~200 m down gradient of the well). Hyporheic groundwater flow reversals from the nearby Willamette River (0.6 km from well 14) may however cause groundwater with high nitrate values from the CAFO to enter the well's capture zone.
Reasons for high parameter variability are not well understood, but it could be impacted by groundwater flow reversals or a pond on the property. The pond has near-constant inflow (roof drainage in winter and well pumping in summer) but no outlet. Constant local recharge via the pond may allow for rapid geochemical changes in the local aquifer. Additionally, the initial coliform bacteria test performed in July 2004 for Well 14 was positive (indicating surface bacteria were present), while a retest in August 2004 was negative (retested due to a sampling error where a spigot attachment was not removed). However, if the initial July 2004 sample was not impacted by the suspected attachment, it indicates that well 14 had (possibly has) surficial contaminant sources.

Site ID:	15	**OWRD Well Log:**	Linn 10769
Owner:	Carpenter, John and Mary		
Address:	29980 Church Dr., Shedd OR 97377		

Drilling Date:	1977	**Geology (Gannett and Caldwell, 1998):**	Qs
Depth (ft):	40	**Geology (O'Connor et al., 2001):**	Qff2
Screening Interval (ft):	none	**Easting:**	485260
State Soil Survey:	Woodburn-Amity-Willamette	**Northing:**	4928628
County Soil Survey:	Woodburn Silt Loam		

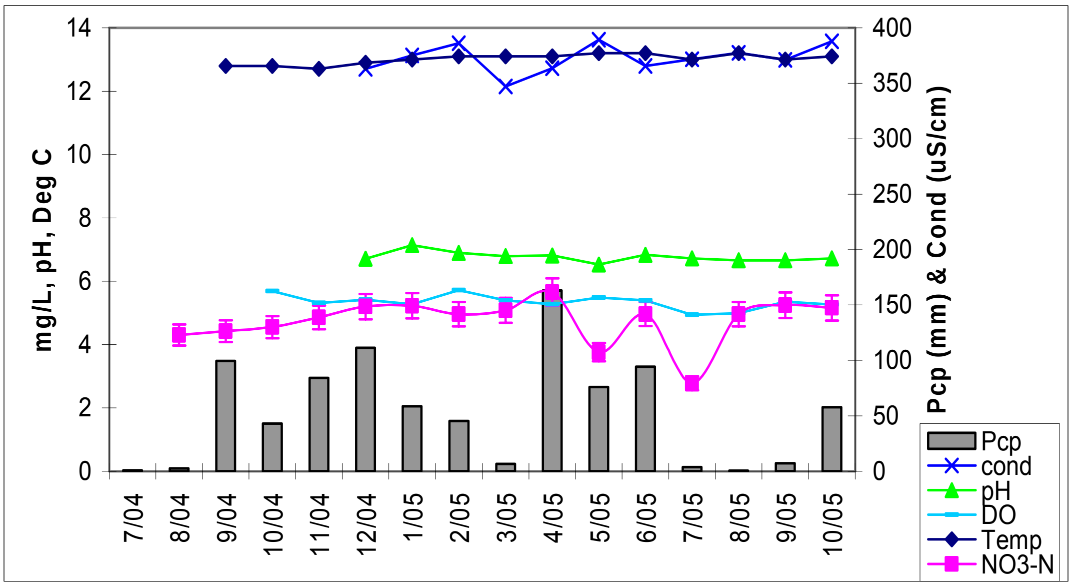


Figure A15: Well 15 trends and interpretations.
Seasonal groundwater nitrate fluctuations were observed in Well 15, with higher concentrations generally occurring in recharge months. Most wells sampled from had unexplainable non-recharge month groundwater nitrate variability, but wells 10, 15, and 18 all show similar non-recharge month fluctuations (with a high April, low May, high June, and low July). One possible explanation for this could be that these wells experienced recharge in April and June (causing high concentrations of groundwater nitrate) and no recharge in May and July (when dilution of high-concentration shallow groundwater could occur).

Relatively high conductivity values observed in wells 10, 11, and 15 are believed to be associated with the aquifer being confined by the Willamette silt (of O’Connor et al. (2001)) in these regions. Well 15 shows relatively high variability in conductivity, which may be associated with the installation and use of a lawn irrigation system which began in March (this also meant that samples were collected at a spigot ~25 ft from the wellhead instead of at the wellhead for the remainder of the study).

Site ID:	16	**OWRD Well Log:**	Linn 55752
Owner:	Oregon Department of Enviromental Quality		
Address:	Harvest Rd		

Drilling Date:	2003	**Geology (Gannett and Caldwell, 1998):**	Qal
Depth (ft):	20	**Geology (O'Connor et al., 2001):**	Qalc
Screening Interval (ft):	17-20	**Easting:**	484598
State Soil Survey:	Malabon-Coburg-Salem	**Northing:**	4930426
County Soil Survey:	Malabon Silty Clay Loam		

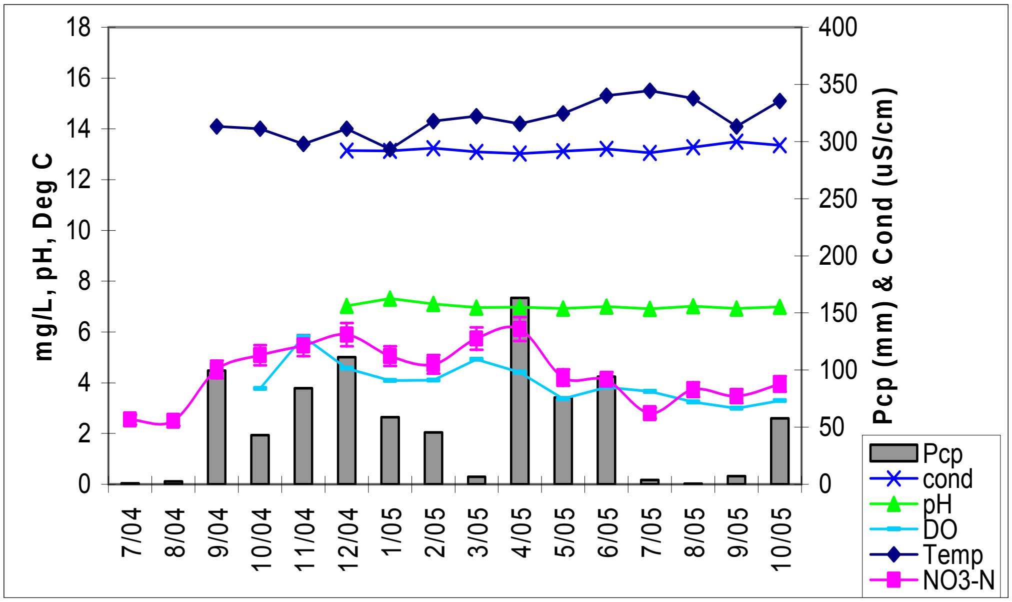


Figure A16: Well 16 trends and interpretations.
Well 16 shows pronounced groundwater nitrate seasonal variability, with well-defined peaks occurring in the highest precipitation months. Nitrate and DO trends are found to be remarkably similar, which is difficult to explain because denitrification generally occurs when DO concentrations are below 1 mg/L.

Site ID:	17	**OWRD Well Log:**	Lane 62312
Owner:	Phillips, Don and Jean		
Address:	29516 McMullin Ln., Junction City, OR 97448		

Drilling Date:	2003	**Geology (Gannett and Caldwell, 1998):**	Qal
Depth (ft):	39	**Geology (O'Connor et al., 2001):**	Qalc
Screening Interval (ft):	28-38	**Easting:**	484693
State Soil Survey:	Newberg-Chehalis-Cloquato	**Northing:**	4902038
County Soil Survey:	Chehalis Silty Clay Loam		

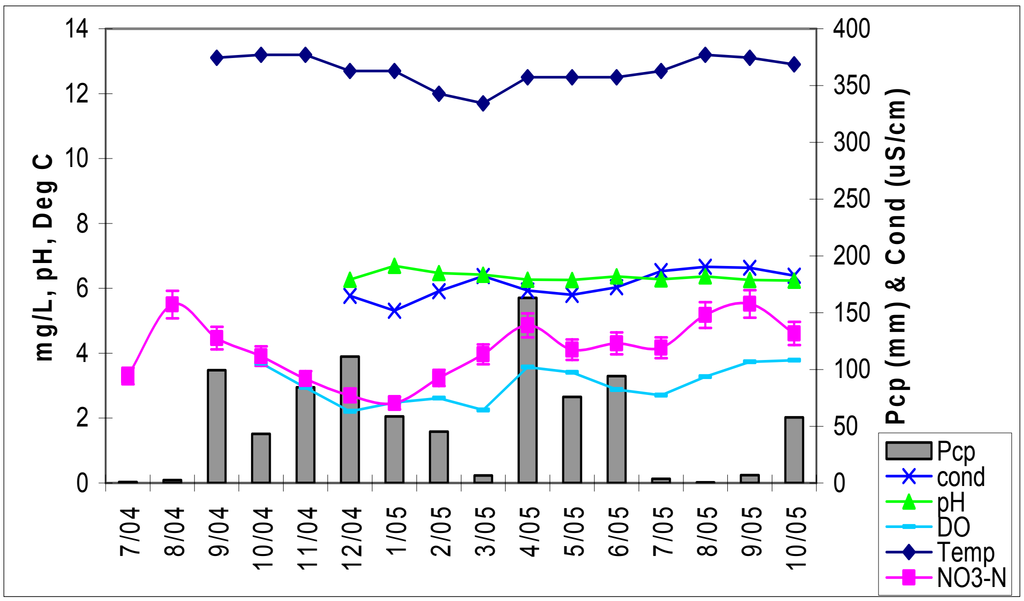


Figure A17: Well 17 trends and interpretations.
Seasonal fluctuations in groundwater nitrate are present in well 17, but the trend generally differed from other wells. Notably, a sizeable decline in nitrate concentrations occurred during the late-fall and early-winter recharge event. However, high concentrations occurred in April (similar to most wells), and the highest concentrations occurred in the late summer months. Explanations for this could be that it is next to a large peppermint field (which could have over-irrigation in the summer, leading to concentration spikes) or that well 17 is less than 100m from a meander slough (many regions near the Willamette River have sloughs where part of the Willamette River's channel once existed). As a surficial expression of the water table, the slough could allow for surface conditions and contaminants to more quickly enter the shallow aquifer. Another possible explanation for the abnormal nitrate seasonality is fluctuations in DO, as groundwater nitrate concentrations follow a similar trend to DO concentrations. If DO is a controlling factor, it would imply that aerobic denitrification can occur locally in the Willamette aquifer. Wells 4 and 16 also imply that aerobic denitrification may be occurring, as well as data presented in Appendix C.

Site ID:	18	**OWRD Well Log:**	Bent 574
Owner:	x		
Address:	x		

Drilling Date:	1987	**Geology (Gannett and Caldwell, 1998):**	Qal
Depth (ft):	42	**Geology (O'Connor et al., 2001):**	Qalc
Screening Interval (ft):	28-36	**Easting:**	478353
State Soil Survey:	Newberg-Chehalis-Cloquato	**Northing:**	4914434
County Soil Survey:	Camas Gravelly Sandy Loam		

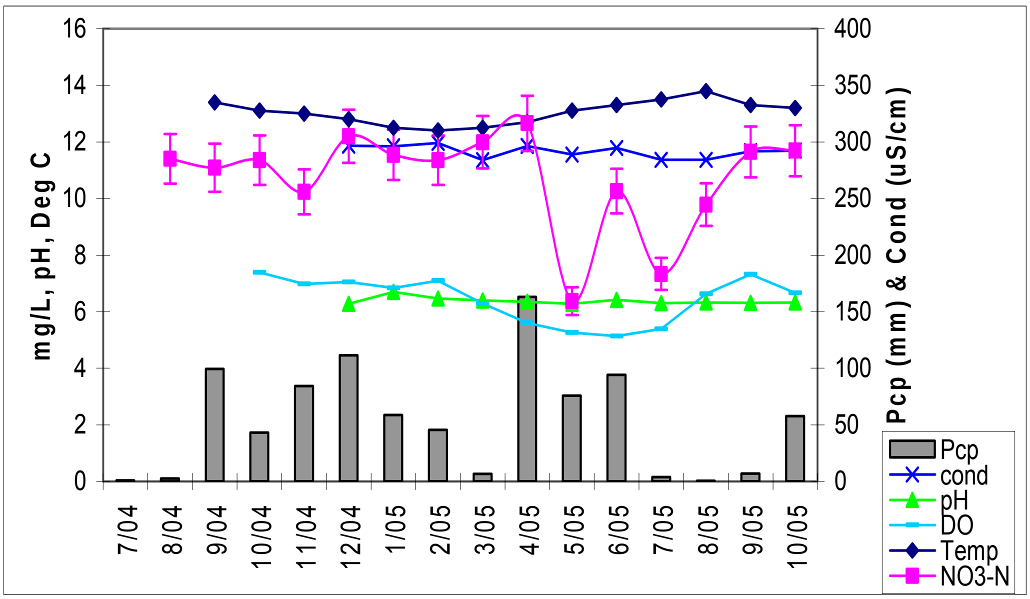


Figure A18: Well 18 trends and interpretations.
Seasonal variability in groundwater nitrate was observed in well 18. Notably, peaks occurred in December and April, the two highest precipitation months. Local nitrate loading sources are thought to be from septics and high intensity agriculture. Most wells sampled from had unexplainable non-recharge month groundwater nitrate variability, but wells 10, 15, and 18 all show similar non-recharge month fluctuations (with a high April, low May, high June, and low July). One possible explanation for this could be that these wells experienced recharge in April and June (causing high concentrations of groundwater nitrate) and no recharge in May and July (when dilution of high-concentration shallow groundwater could occur).

Site ID:	19	**OWRD Well Log:**	Bent 7101
Owner:	Briles, Cynthia and James		
Address:	24771 to 24773 HWY 99W, Monroe OR, PO Box 497		

Drilling Date:	1986	**Geology (Gannett and Caldwell, 1998):**	Qal
Depth (ft):	30	**Geology (O'Connor et al., 2001):**	Qg1
Screening Interval (ft):	20-29	**Easting:**	476933
State Soil Survey:	Dayton-Amity-Aloha	**Northing:**	4906486
County Soil Survey:	Malabon Silty Clay Loam		

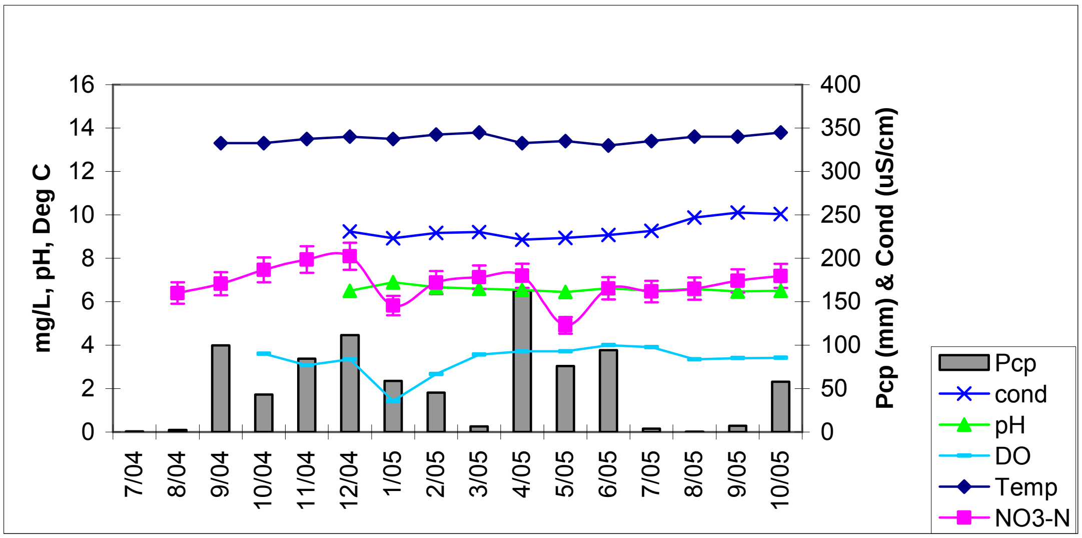


Figure A19: Well 19 trends and interpretations.
Well 19 had pronounced variability in groundwater nitrate, with peaks occurring December and April, the months with the highest precipitation. Nitrate sources for the well are believed to largely be agricultural or residual from a previous dairy on the property.

Appendix B

Duplicate Analysis of Nitrate Data

Sample spikes, blanks, and duplicates composed 10% of all sample analyses run. Results of spike and blank analyses are shown in Figure B1. Spike and blank data show high precision and a slight bias, with observed concentrations being slightly higher than spike values. As strict adherence to protocol 4500-NO_3^-B (from Eaton et al. (1995)) was taken in spike preparation, it is unclear whether spikes prepared in this study or those used by Central Analytical Laboratory for instrument calibration are erroneous. If spikes used in the instrument calibration are inaccurate, all nitrate values reported in this study are approximately 9% higher in concentration than their actual value.

Duplicates collected are presented in Figure B2. Of the 25 duplicates collected, 6 had absolute differences > 10% between the two samples. It is unclear why these duplicates showed poor agreement. Possibly because samples were not acidified in the field (samples were stored on ice until the end of the sample day, at which point they were frozen until analysis), microbial degradation may have occurred in some samples before freezing. For all 25 duplicates, the mean difference between samples was 6.8%, with a standard deviation of 18.9%. The mean value indicates that duplicate samples on average had slightly higher concentrations than initial samples (with a 0.0% duplicate difference indicating that the values of the duplicate pair agree exactly). The 95% confidence interval calculated based on this standard deviation was +/- 7.4%. The mean and standard deviation values for the duplicate population excluding the 6 samples with absolute differences > 10% were 0.0% and 3.7%, indicating that the samples with large errors caused the population mean and standard deviation to substantially inflate.

Table B1 presents duplicate data collected. All 4 duplicates collected in July and August 2005 had high (>10%) disagreement between duplicate values, but no temporal explanation exists to explain why both months had large differences.

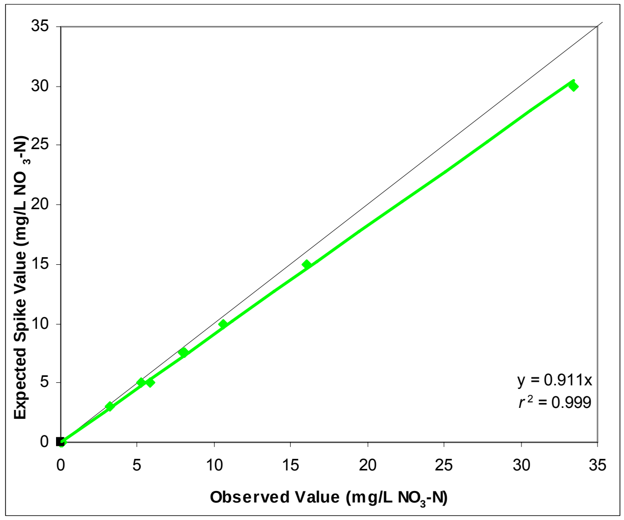


Figure B1. Spike data submitted to Central Analytical Laboratory (CAL), with a one-to-one line shown in black. Expected spike values are calculated based on a known mass of nitrate present in the spike, while the observed values are those reported by CAL.

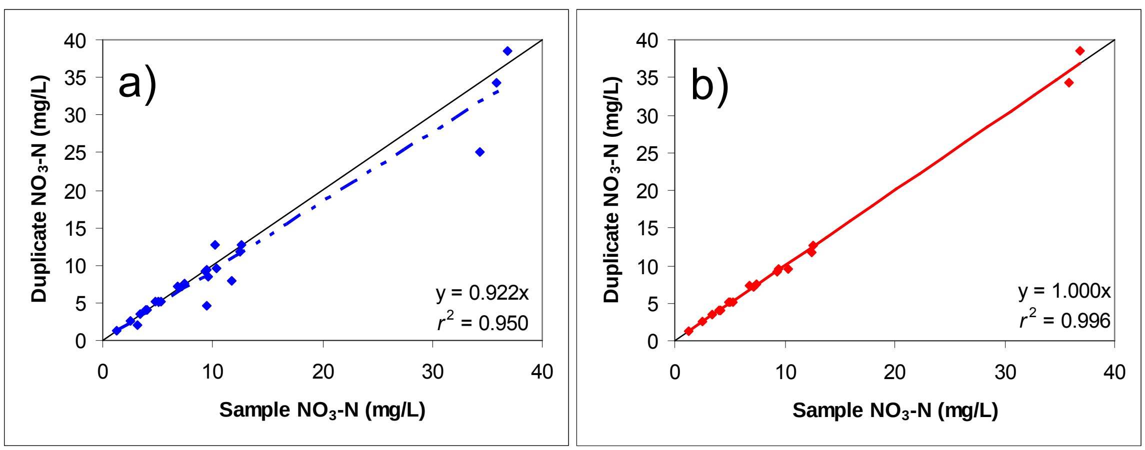


Figure B2. a) Duplicate data plot, with one-to-one line in solid black. 6 of the 25 duplicates had large differences between samples. It is unclear why such differences exist. b). One-to-one plot with duplicate data, excluding the 6 outliers.

Table B1. Groundwater nitrate duplicate data. The percent difference is calculated as (Sample 1- Duplicate)/((Sample 1 + Duplicate)/2).

Month	Site #	Sample 1	Duplicate	% Difference
Sep-04	5	6.8	7.3	-6.3
Sep-04	1	5.3	5.2	1.3
Sep-04	10	5.2	5.2	-0.6
Oct-04	14	7.4	7.5	-1.2
Oct-04	6	35.8	34.3	4.2
Dec-04	9	4.1	4.0	2.5
Jan-05	6	34.3	25.1	30.8
Feb-05	7	10.4	9.6	7.9
Feb-05	3	9.4	9.2	1.8
Mar-05	19	7.1	7.1	-0.1
Mar-05	17	4.0	4.0	0.0
Apr-05	18	12.6	12.7	-0.4
Apr-05	6	36.8	38.6	-4.7
May-05	2	10.3	12.7	-21.6
May-05	13	1.3	1.3	-0.8
Jun-05	4	2.6	2.6	0.4
Jun-05	15	4.9	5.1	-4.6
Jul-05	8	9.5	4.7	68.1
Jul-05	11	3.2	2.1	42.3
Aug-05	8	9.6	8.5	11.3
Aug-05	18	11.7	7.9	39.6
Sep-05	2	12.5	11.7	6.4
Sep-05	16	3.4	3.6	-4.9
Oct-05	11	5.0	5.1	-0.8
Oct-05	3	9.5	9.5	0.3

Appendix C

Field Parameters Collected
(Dissolved Oxygen, Temperature, Conductivity, pH, Depth to Water)

Field parameters collected to determine sufficient purging for nitrate samples are presented in this appendix. Field parameters (pH, temperature, specific conductivity, and dissolved oxygen (DO)) were collected throughout the well purge, and values recorded herein are the last values recorded before nitrate sampling occurred. Field equipment used included a YSI model 52 for measuring DO and temperature, while a YSI model 63 was used to measure temperature, pH, and specific conductivity. Wells were purged for a minimum of 15 minutes and samples were collected after all field parameters were stable over 3 consecutive 3 minute recording intervals. Stabilization protocols were based on those of Koterba et al. (1995). Data from August through October 2004 had stabilization parameters recorded, but are not presented because of lower data confidence. Additionally, August through October 2004 had slightly different purge protocols, with purging occurring for 5 minutes and samples being collected several minutes later after field parameters were considered stable. Duplicate samples collected using both purge protocols were compared and found to have negligible differences.

For Figures C5-C16, linear regression lines between groundwater nitrate and field parameters are shown only if their r^2 is greater than 0.40. Depth to groundwater data are presented in Figure C17.

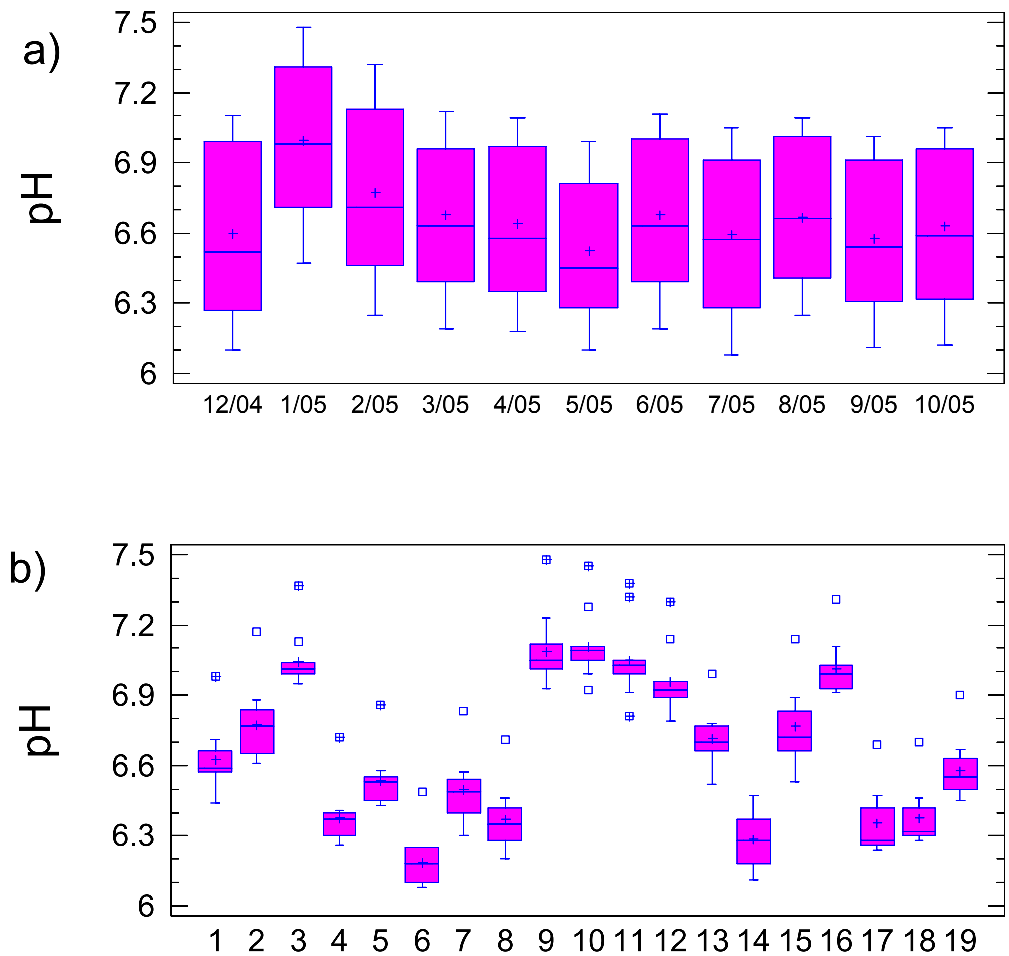


Figure C1. pH figures for monthly variability (a) and intra-well variability (b). The high network-wide values observed in January are thought to be associated with poor calibration of the meter and probably do not reflect a regional difference. Following sampling in March and all months thereafter, a post-sampling pH check was conducted to ensure data quality.

Table C1. Observed monthly pH values.

	12/2004	1/2005	2/2005	3/2005	4/2005	5/2005	6/2005	7/2005	8/2005	9/2005	10/2005	Average	Std Dev
1	6.57	6.98	6.71	6.63	6.58	6.44	6.60	6.57	6.66	6.54	6.59	6.62	0.14
2	6.73	7.17	6.88	6.84	6.80	6.61	6.77	6.63	6.68	6.65	6.77	6.78	0.16
3	6.99	7.37	7.13	7.03	7.04	6.95	7.01	7.00	7.01	6.97	7.00	7.05	0.12
4	6.26	6.72	6.40	6.37	6.39	6.30	6.38	6.28	6.41	6.31	6.32	6.38	0.13
5	6.46	6.86	6.58	6.55	6.45	6.43	6.55	6.44	6.55	6.48	6.53	6.53	0.12
6	6.10	6.49	6.25	6.19	6.18	6.10	6.19	6.08	6.25	6.11	6.12	6.19	0.12
7	6.42	6.83	6.57	6.54	6.49	6.30	6.52	6.39	6.54	6.40	6.49	6.50	0.14
8	6.27	6.71	6.46	6.39	6.30	6.20	6.39	6.28	6.42	6.34	6.35	6.37	0.13
9	7.06	7.48	7.23	7.12	7.03	6.93	7.05	7.01	7.07	7.01	6.96	7.09	0.15
10	7.10	7.45	7.28	7.08	7.09	6.99	7.11	7.05	7.09	6.92	7.05	7.11	0.14
11	6.99	7.38	7.32	7.03	7.03	6.81	7.04	6.99	7.05	6.91	6.99	7.05	0.16
12	6.92	7.30	7.14	6.90	6.93	6.79	6.89	6.89	6.92	6.90	6.96	6.96	0.14
13	6.52	6.99	6.78	6.68	6.66	6.55	6.73	6.70	6.77	6.70	6.77	6.71	0.12
14	6.18	6.47	6.36	6.39	6.26	6.14	6.37	6.24	6.33	6.11	6.28	6.28	0.11
15	6.71	7.14	6.89	6.79	6.81	6.53	6.83	6.72	6.66	6.66	6.72	6.77	0.16
16	7.03	7.31	7.11	6.96	6.97	6.92	7.00	6.91	7.02	6.93	6.99	7.01	0.11
17	6.27	6.69	6.47	6.42	6.27	6.26	6.37	6.28	6.37	6.26	6.24	6.35	0.13
18	6.28	6.70	6.46	6.39	6.35	6.28	6.42	6.30	6.32	6.31	6.32	6.38	0.12
19	6.50	6.90	6.67	6.60	6.55	6.45	6.63	6.51	6.59	6.48	6.50	6.58	0.13
Average	6.60	7.00	6.77	6.68	6.64	6.53	6.68	6.59	6.67	6.58	6.63	6.67	
Std Dev	0.33	0.32	0.33	0.28	0.30	0.29	0.28	0.30	0.28	0.29	0.30	0.30	

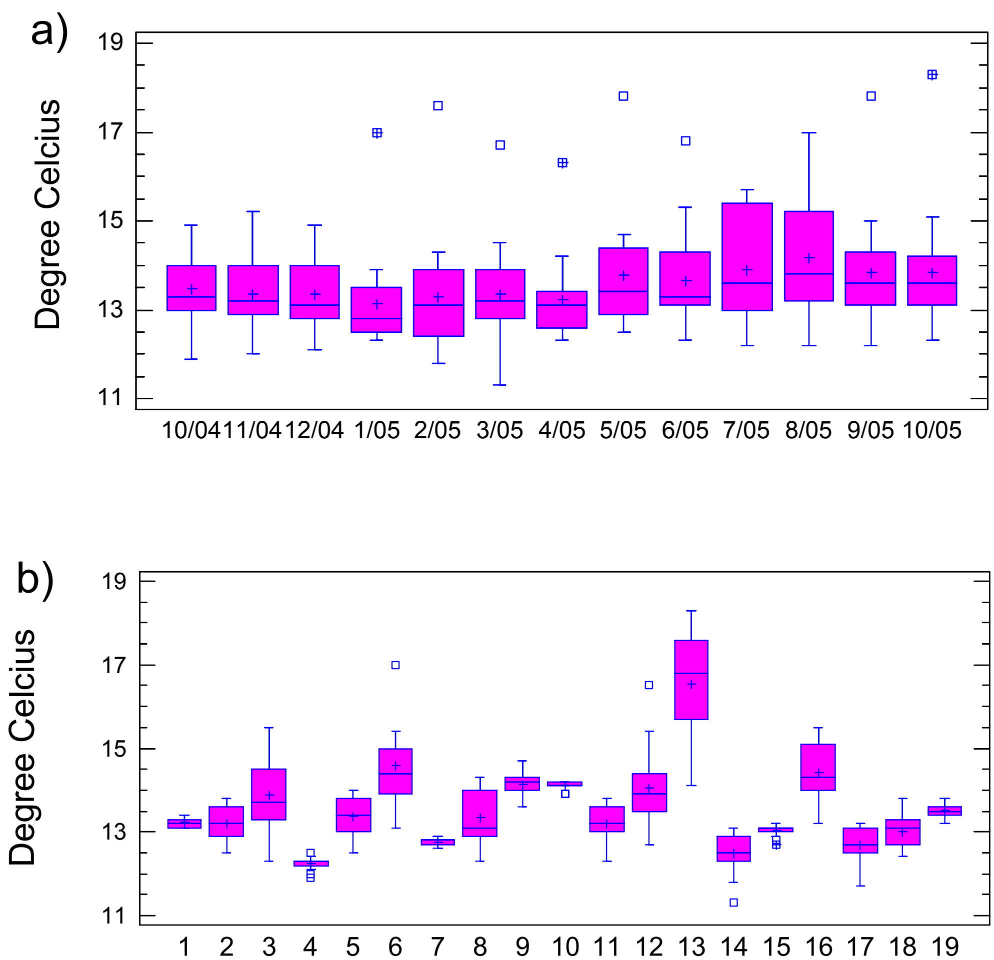


Figure C2. Groundwater temperature figures for monthly variability (a) and intra-well variability (b).

Table C2. Observed monthly groundwater temperature values.

	10/2004	9/2004	11/2004	12/2004	1/2005	2/2005	3/2005	4/2005	5/2005	6/2005	7/2005	8/2005	9/2005	10/2005	Average	Std Dev
1	13.6	13.1	13.1	13.1	13.2	13.3	13.4	13.3	13.4	13.1	13.4	13.3	13.2	13.2	13.3	0.1
2	13.5	13.0	12.9	12.8	12.5	12.9	13.2	13.2	13.6	13.6	13.6	13.8	13.4	13.2	13.2	0.4
3	13.6	13.7	13.3	13.2	12.3	13.1	13.4	13.4	14.6	14.5	15.5	15.4	14.3	13.9	13.9	0.9
4	12.1	11.9	12.0	12.1	12.3	12.4	12.4	12.3	12.5	12.3	12.2	12.2	12.2	12.3	12.2	0.2
5	13.8	14.0	14.0	14.0	13.4	13.0	12.8	12.5	12.7	13.1	13.3	13.6	13.6	13.8	13.4	0.5
6	14.1	14.3	15.2	14.4	13.5	13.9	13.9	13.1	14.4	14.6	15.4	17.0	15.0	14.9	14.6	1.0
7	14.3	12.7	12.6	12.8	12.8	12.8	12.8	12.6	12.9	12.6	12.9	12.7	12.8	12.7	12.9	0.4
8	13.4	13.0	13.1	12.3	12.5	12.8	13.0	12.9	14.0	13.6	14.2	14.1	14.3	13.6	13.3	0.6
9	14.2	14.2	14.0	13.8	13.6	13.9	14.2	14.0	14.2	14.3	14.4	14.7	14.3	14.2	14.1	0.3
10	14.4	14.1	14.1	14.1	13.9	13.9	14.1	14.1	14.2	14.2	14.2	14.1	14.2	14.2	14.1	0.1
11	13.7	13.7	13.2	13.0	12.8	12.3	13.2	12.8	13.3	13.1	13.6	13.6	13.8	13.4	13.3	0.4
12	13.9	13.9	13.6	13.8	12.7	13.1	13.5	12.9	14.7	14.0	15.4	16.5	14.4	14.1	14.0	1.0
13	14.2	14.9	14.1	14.9	17.0	17.6	16.7	16.3	17.8	16.8	15.7	17.0	17.8	18.3	16.4	1.4
14	12.7	13.1	12.9	13.1	12.3	11.8	11.3	12.3	12.7	12.3	12.3	12.5	12.7	13.0	12.5	0.5
15	12.8	12.8	12.7	12.9	13.0	13.1	13.1	13.1	13.2	13.2	13.0	13.2	13.0	13.1	13.0	0.2
16	14.1	14.0	13.4	14.0	13.2	14.3	14.5	14.2	14.6	15.3	15.5	15.2	14.1	15.1	14.4	0.7
17	13.1	13.2	13.2	12.7	12.7	12.0	11.7	12.5	12.5	12.5	12.7	13.2	13.1	12.9	12.7	0.5
18	13.4	13.1	13.0	12.8	12.5	12.4	12.5	12.7	13.1	13.3	13.5	13.8	13.3	13.2	13.0	0.4
19	13.3	13.3	13.5	13.6	13.5	13.7	13.8	13.3	13.4	13.2	13.4	13.6	13.6	13.8	13.5	0.2
Average	13.6	13.5	13.4	13.3	13.1	13.3	13.3	13.2	13.8	13.7	13.9	14.2	13.8	13.8	13.6	
Std Dev	0.6	0.7	0.7	0.7	1.1	1.3	1.1	0.9	1.2	1.1	1.1	1.4	1.2	1.3	0.9	

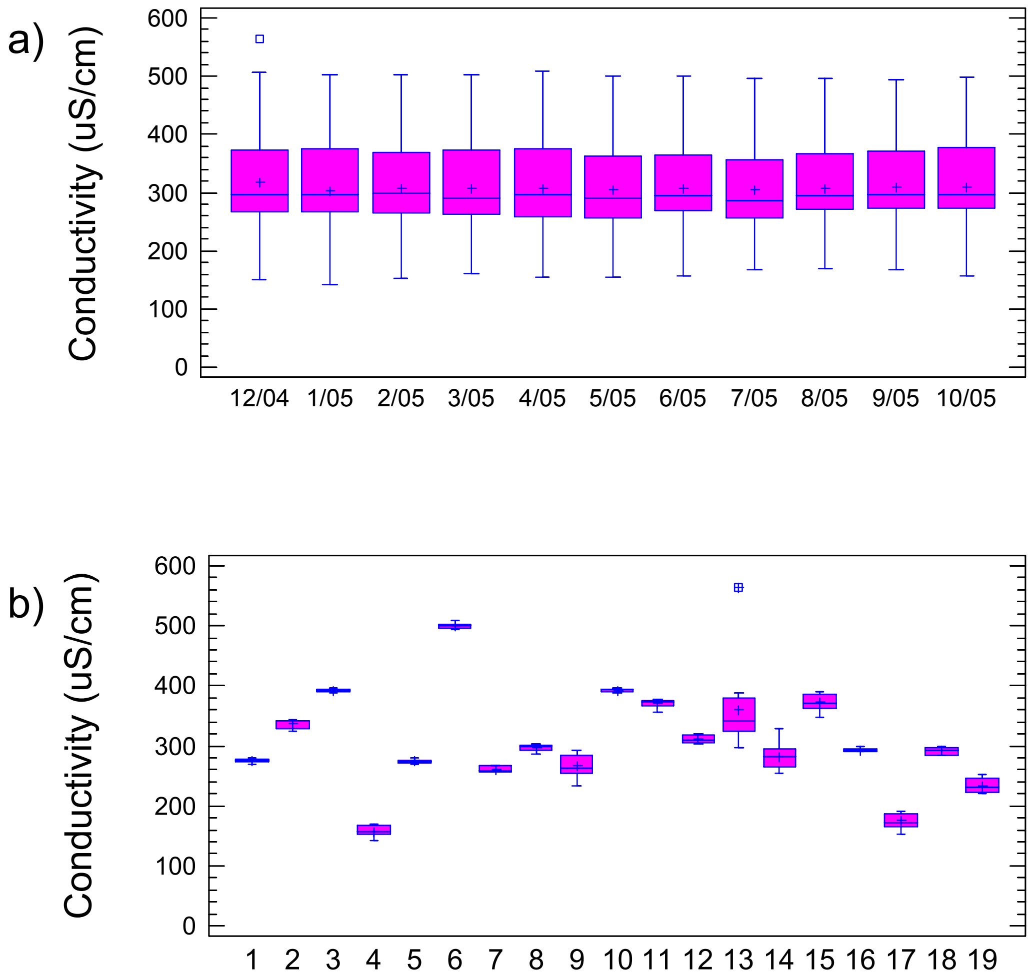
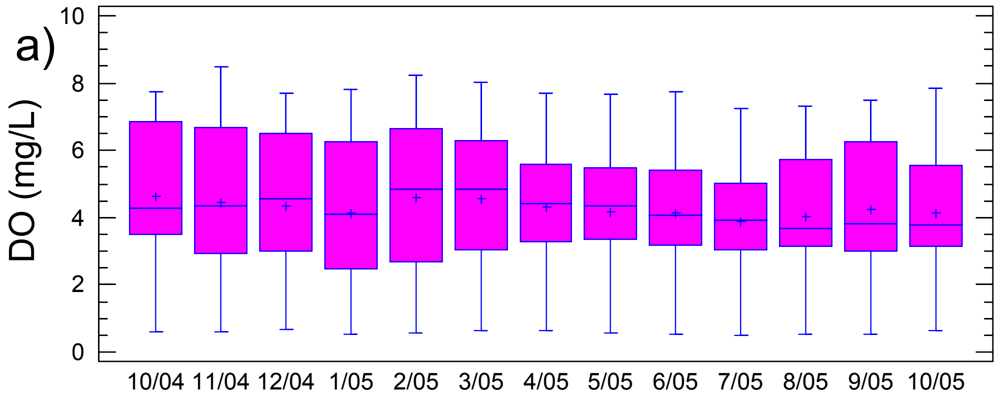


Figure C3. Specific conductivity values for network-wide monthly variability (a) and intra-well variability (b).

Table C3. Observed monthly specific conductivity values.

	12/2004	1/2005	2/2005	3/2005	4/2005	5/2005	6/2005	7/2005	8/2005	9/2005	10/2005	Average	Std Dev
1	269.9	273.1	274.9	274.5	276.8	275.3	271.3	277.1	277.0	278.2	280.4	275.3	3.1
2	343.0	341.9	341.8	340.6	339.9	326.8	334.4	325.0	329.3	341.2	343.8	337.1	6.9
3	394.4	390.5	395.0	395.4	392.4	390.8	393.4	392.0	395.6	392.7	387.0	392.7	2.6
4	150.4	143.1	153.6	161.9	153.9	154.4	156.9	167.7	169.6	166.8	156.1	157.7	8.1
5	280.4	277.7	276.3	275.5	271.6	274.3	268.2	272.4	276.4	273.0	274.5	274.6	3.3
6	507.0	503.0	503.0	502.0	509.0	500.0	500.0	496.8	497.1	494.4	498.1	500.9	4.4
7	267.2	268.0	267.9	262.2	258.0	256.5	258.6	256.0	256.3	258.0	266.3	261.4	5.0
8	300.5	303.8	303.2	300.6	293.6	289.7	295.1	285.4	298.7	295.9	298.8	296.8	5.7
9	256.9	258.1	254.0	262.8	251.8	232.7	283.6	283.5	286.5	283.6	291.8	267.8	18.9
10	392.7	393.5	394.8	393.8	395.3	389.9	392.9	388.2	393.5	390.1	396.6	392.8	2.5
11	372.9	376.7	369.4	372.9	374.6	361.5	371.9	356.6	366.8	375.0	377.9	370.6	6.6
12	305.1	302.7	308.6	310.6	310.6	317.5	320.8	320.9	314.9	308.5	304.0	311.3	6.4
13	564.0	297.8	342.1	379.8	387.7	340.7	343.2	341.1	324.4	319.2	328.2	360.7	72.1
14	294.8	274.9	264.2	264.0	299.5	328.4	288.1	254.6	270.4	290.0	282.3	282.8	20.7
15	363.0	375.0	386.2	347.0	363.5	389.4	365.5	371.6	377.4	371.1	387.9	372.5	12.7
16	292.1	291.8	294.0	291.0	289.5	291.5	293.7	290.1	294.9	299.9	296.7	293.2	3.1
17	164.7	151.8	169.0	182.4	169.4	165.6	172.3	186.7	190.4	189.4	182.7	174.9	12.3
18	296.7	296.0	299.0	283.8	296.4	288.7	294.7	284.2	284.1	291.9	292.2	291.6	5.6
19	231.0	223.0	229.2	230.4	221.3	223.5	226.9	231.7	246.8	252.7	251.1	233.4	11.4
Average	318.2	302.2	306.6	306.9	308.1	305.1	306.9	304.3	307.9	309.0	310.3	307.8	
Std Dev	101.3	84.5	82.8	80.6	84.7	83.1	80.1	77.0	76.0	75.1	78.2	80.4	



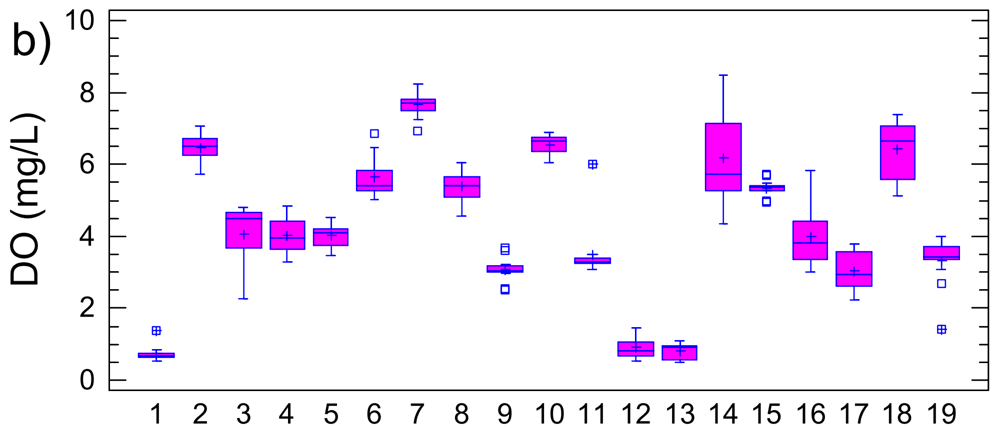


Figure C4. Dissolved oxygen (DO) values for network-wide monthly variability (a) and intra-well variability (b).

Table C4. Observed monthly dissolved oxygen values.

	10/2004	11/2004	12/2004	1/2005	2/2005	3/2005	4/2005	5/2005	6/2005	7/2005	8/2005	9/2005	10/2005	Average	Std Dev
1	1.38	0.61	0.68	0.52	0.57	0.64	0.65	0.74	0.84	0.72	0.80	0.73	0.63	0.73	0.21
2	7.01	6.69	6.49	6.26	7.08	6.73	6.40	6.53	6.13	5.97	5.72	6.26	6.97	6.48	0.41
3	4.64	4.36	4.55	4.47	4.67	4.78	4.80	4.68	4.08	3.68	3.09	2.96	2.25	4.08	0.83
4	3.65	3.36	3.30	3.94	4.85	4.84	4.34	4.67	4.41	4.03	3.79	3.82	3.45	4.03	0.54
5	4.27	4.01	4.20	3.91	4.10	4.52	4.20	3.73	3.47	3.60	3.68	4.17	4.41	4.02	0.32
6	6.84	5.83	5.40	5.25	5.10	5.70	5.29	5.29	5.24	5.02	6.39	6.45	5.56	5.64	0.58
7	7.74	6.94	7.72	7.80	8.22	8.02	7.71	7.67	7.74	7.26	7.32	7.50	7.85	7.65	0.33
8	6.03	5.08	6.06	5.56	6.02	5.67	5.39	5.48	5.26	4.56	5.05	5.10	5.24	5.42	0.45
9	2.52	2.56	2.99	2.96	3.21	3.03	3.03	3.00	3.19	3.04	3.60	3.69	3.14	3.07	0.33
10	6.87	6.76	6.70	6.47	6.82	6.89	6.40	6.64	6.36	6.05	6.14	6.71	6.22	6.54	0.29
11	3.51	3.31	3.25	3.26	6.02	3.51	3.28	3.39	3.25	3.06	3.13	3.28	3.21	3.50	0.77
12	0.61	1.45	0.68	0.67	0.90	0.79	1.38	1.34	1.05	0.82	0.69	0.54	1.05	0.92	0.31
13	1.07	0.88	0.97	0.94	1.10	1.10	0.93	0.56	0.52	0.48	0.53	0.73	0.92	0.83	0.23
14	7.45	8.47	7.15	6.50	6.64	7.76	5.73	4.35	5.47	5.03	4.87	5.56	5.25	6.17	1.26
15	5.69	5.31	5.42	5.27	5.72	5.40	5.27	5.49	5.40	4.94	4.99	5.36	5.26	5.35	0.22
16	3.77	5.82	4.57	4.09	4.10	4.92	4.41	3.37	3.83	3.66	3.24	3.00	3.29	4.01	0.78
17	3.69	2.94	2.21	2.48	2.61	2.25	3.57	3.41	2.88	2.71	3.28	3.73	3.78	3.04	0.57
18	7.39	6.98	7.05	6.84	7.10	6.29	5.60	5.26	5.13	5.39	6.63	7.32	6.66	6.43	0.81
19	3.60	3.07	3.35	1.43	2.67	3.57	3.71	3.71	4.00	3.91	3.35	3.40	3.42	3.32	0.67
Average	4.62	4.44	4.35	4.14	4.61	4.55	4.32	4.17	4.12	3.89	4.02	4.23	4.13	4.28	
Std Dev	2.27	2.25	2.24	2.22	2.27	2.24	1.89	1.93	1.92	1.84	1.98	2.14	2.08	2.03	

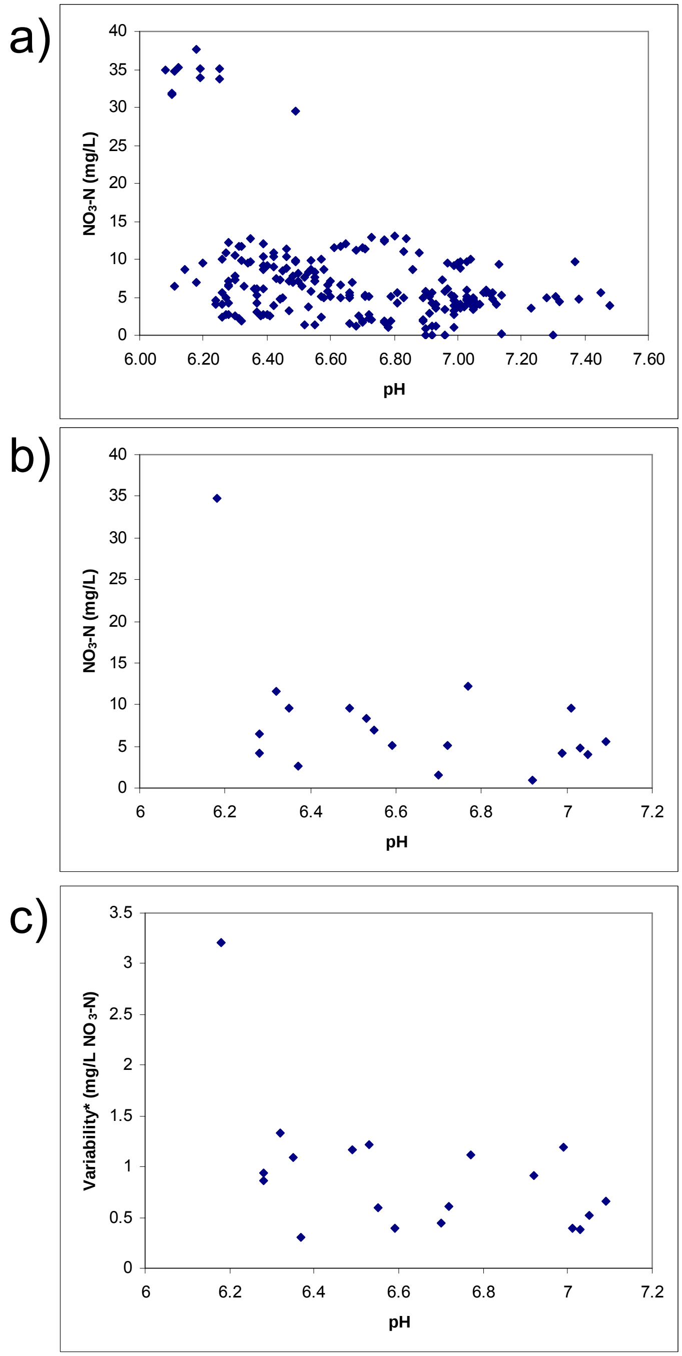


Figure C5. a) Plot of groundwater nitrate concentrations versus pH for all data points where both parameters exist. No correlation between pH and groundwater nitrate was found. b) Median groundwater nitrate concentration versus median pH. c) Groundwater nitrate variability versus pH. Nitrate variability was calculated as the nonparametric equivalent of a standard deviation, which is the spread between the 30.85th and 69.15th percentile of a well's sample population.

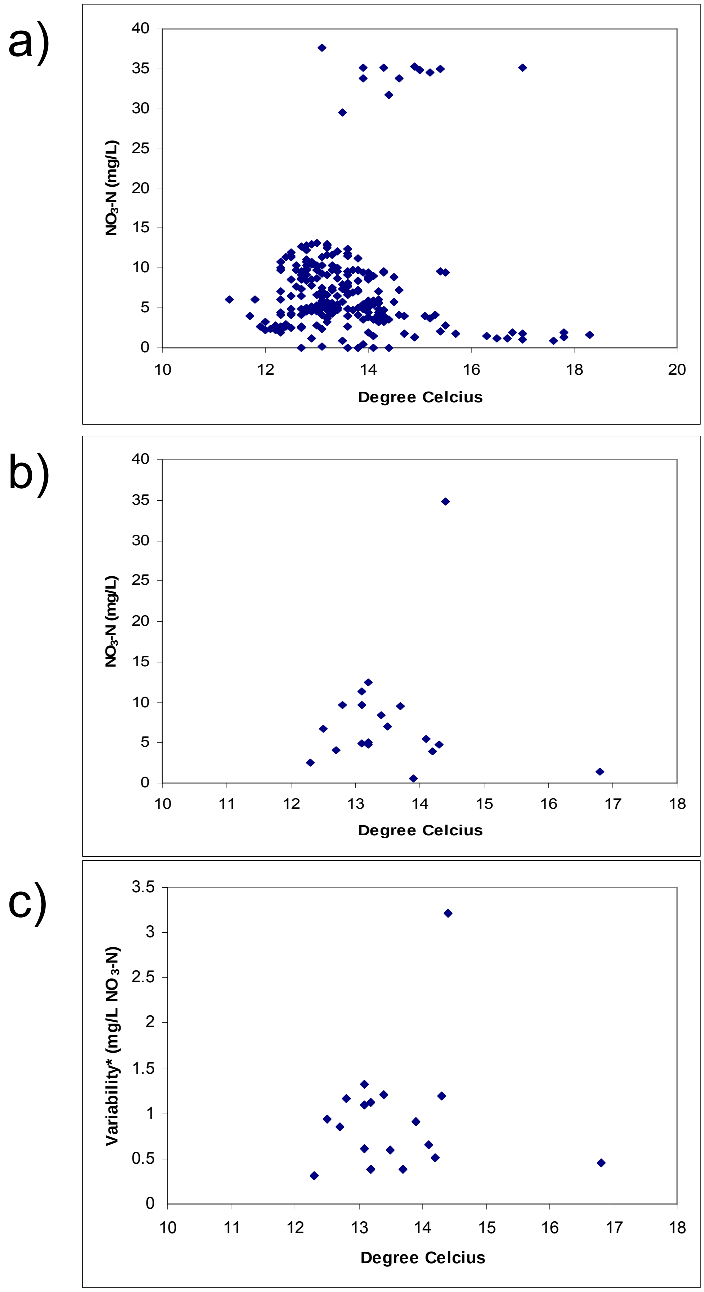


Figure C6. a) Plot of groundwater nitrate concentrations versus groundwater temperature for all data points where both parameters exist. No correlation between temperature and groundwater nitrate was found. b) Median groundwater nitrate concentration versus median groundwater temperature. c) Groundwater nitrate variability versus groundwater temperature. Nitrate variability was calculated as the nonparametric equivalent of a standard deviation, which is the spread between the 30.85th and 69.15th percentile of a well's sample population.

a)

b)

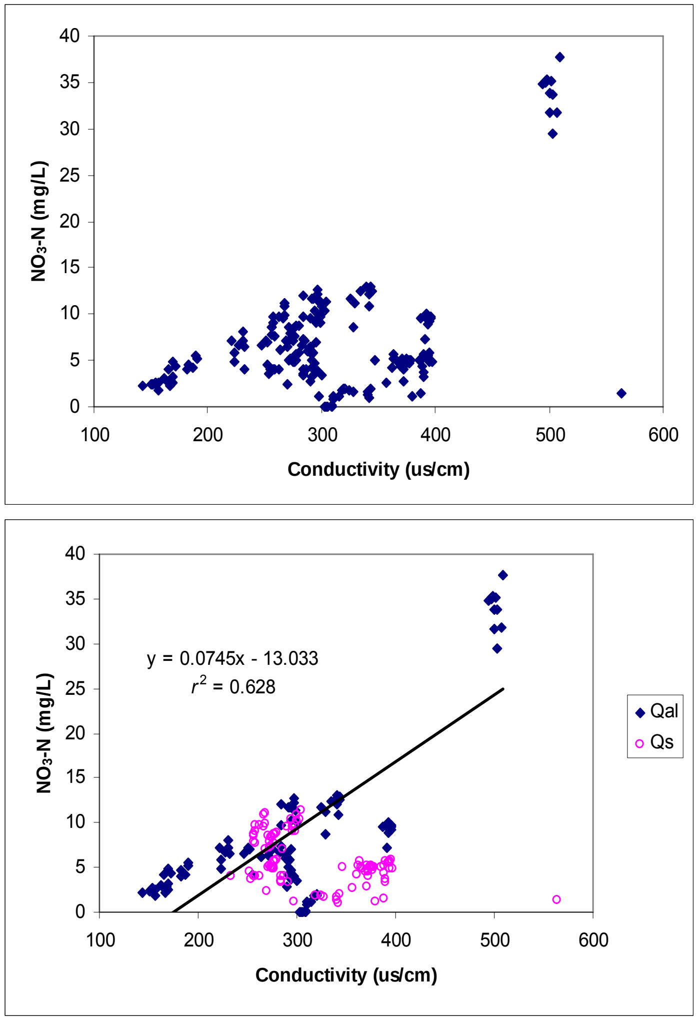
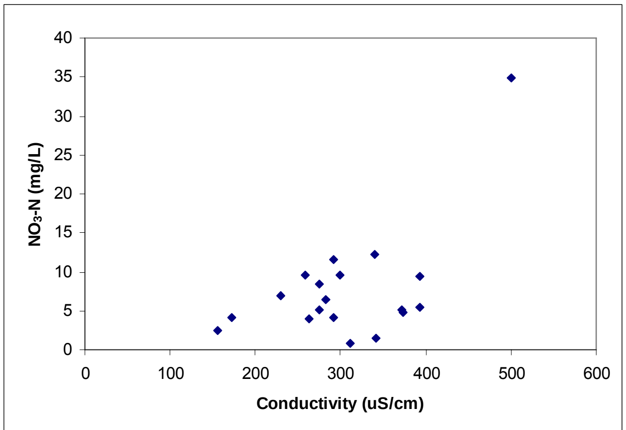


Figure C7. Plot of groundwater nitrate concentrations versus specific conductivity (a) for all data points where both parameters exist. b) Nitrate concentrations versus specific conductivity for all sites sorted by the geologic units of Gannett and Caldwell (1998). For the Willamette aquifer unit (Qal), a correlation exists between nitrate and specific conductivity. Specific conductivity and groundwater nitrate concentrations do not correlate for the entire population because wells penetrating the Willamette silt (as defined by O'Connor et al. (2001)) were found to have high specific conductivity values, but low nitrate values.

a)



b)

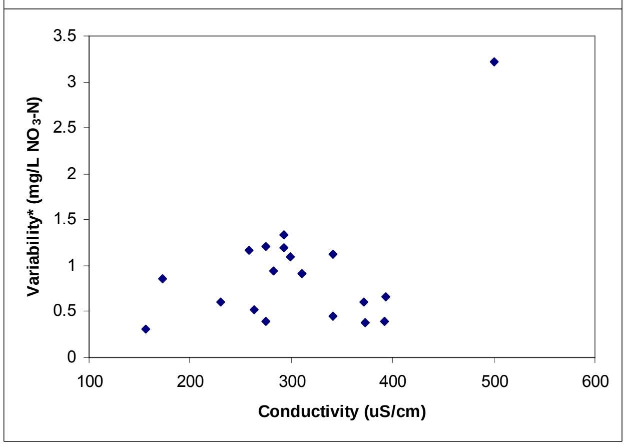


Figure C8. a) Median groundwater nitrate concentration versus median specific conductivity. b) Groundwater nitrate variability versus specific conductivity. Nitrate variability was calculated as the nonparametric equivalent of a standard deviation, which is the spread between the 30.85th and 69.15th percentile of a well's sample population.

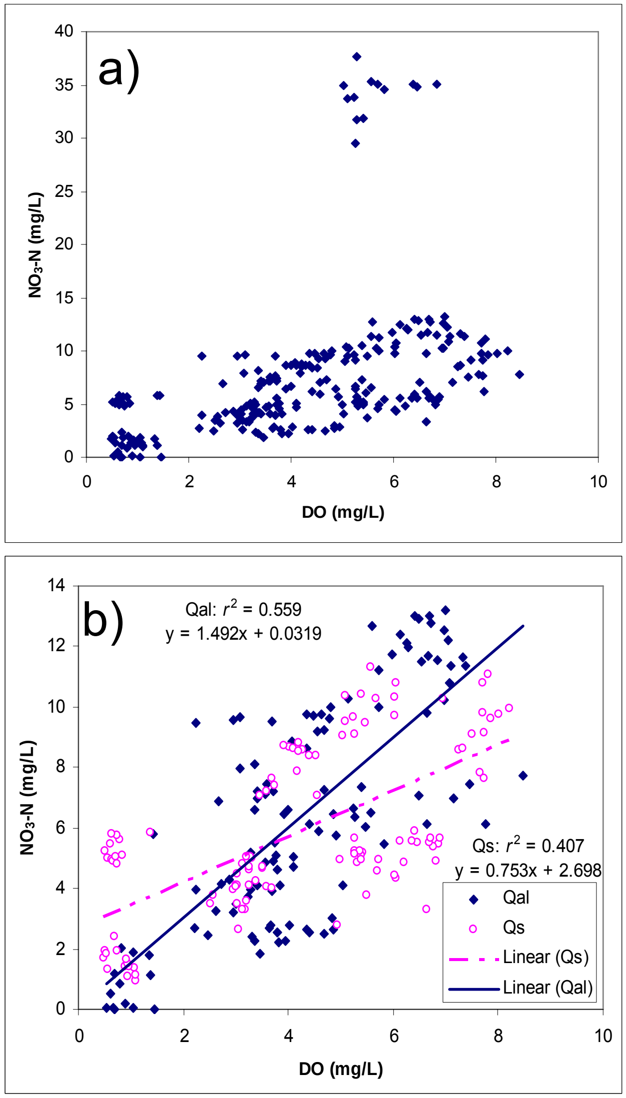


Figure C9. a) Plot of groundwater nitrate concentrations versus dissolved oxygen (DO) for all data points where both parameters exist. b) Nitrate concentrations versus DO for all sites (excluding well 6, which reflects a CAFO point source), sorted by the geologic units of Gannett and Caldwell (1998). Both Willamette silt (Qs) and Willamette aquifer (Qal) units show a correlation between nitrate and DO. The r^2 value when both geologic units are combined is 0.453 (again excluding well 6).

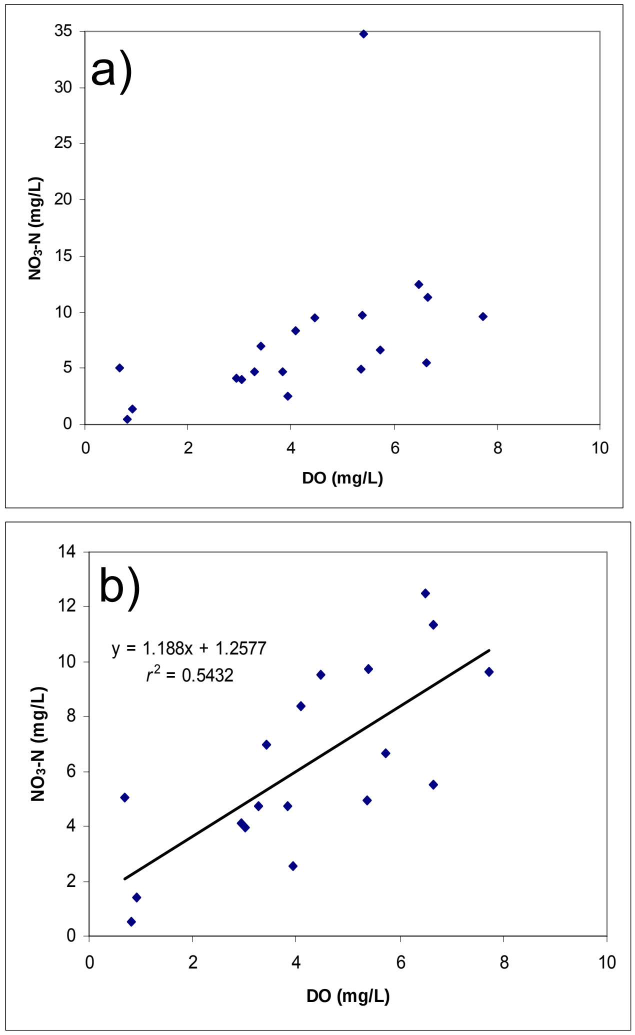


Figure C10. a) Median groundwater nitrate concentration versus median DO. b) When well 6 is excluded, a correlation is observed between groundwater nitrate and DO.

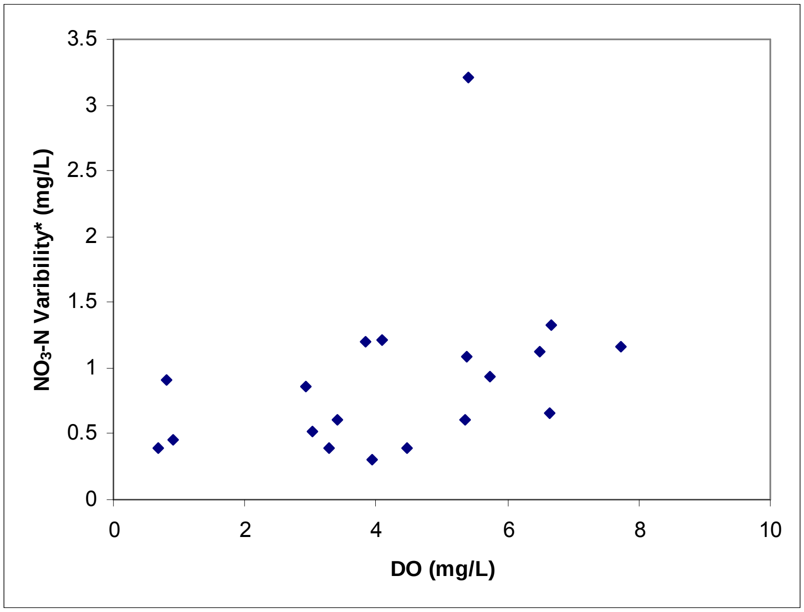


Figure C11. Groundwater nitrate variability versus DO. Nitrate variability was calculated as the nonparametric equivalent of a standard deviation, which is the spread between the 30.85th and 69.15th percentile of a well's sample population.

Table C5. Monthly depth to groundwater observed in the 4 monitoring wells and precipitation between sampling events. Precipitation data is from the Corvallis Agrimet site (US Bureau of Reclamation, 2005).

Month	Depth to water (m)				Precipitation (mm)
	3	6	12	16	
July-04	3.51	3.71	4.88	2.22	0.8
Aug-04	3.71	3.91	4.98	2.29	2.5
Sep-04	3.80	4.11	5.11	2.39	99.5
Oct-04	3.78	4.27	5.22	2.48	43.2
Nov-04	3.71	4.14	5.11	2.44	84.3
Dec-04	3.60	4.06	4.97	1.64	111.4
Jan-05	3.28	3.49	4.65	1.46	58.7
Feb-05	3.30	3.43	4.72	1.57	45.4
Mar-05	3.32	3.56	4.83	2.03	6.6
Apr-05	2.94	3.15	4.18	0.86	163.0
May-05	2.84	3.23	4.25	1.24	76.0
Jun-05	2.88	3.21	4.42	1.53	94.3
Jul-05	3.16	3.39	4.65	1.80	3.8
Aug-05	3.54	3.61	4.91	2.11	0.5
Sep-05	3.76	3.85	5.09	2.34	7.1
Oct-05	3.86	4.06	5.22	2.55	57.9

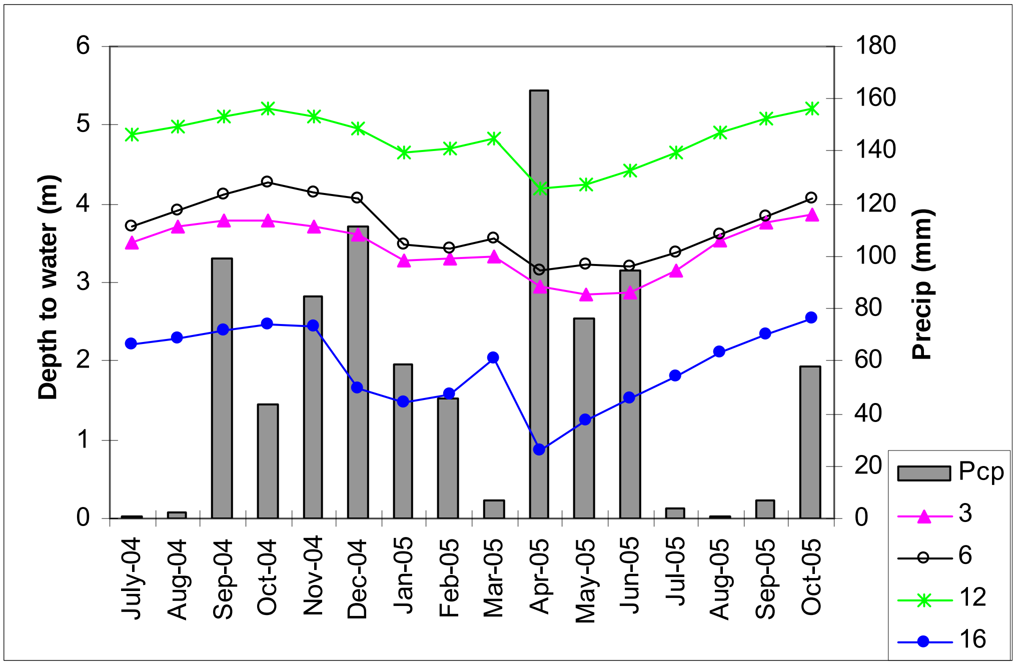


Figure C12. Graph of monthly depth to groundwater observed in wells 3, 6, 12, and 16. All data are presented in Table C5. Data points are connected to guide the eye and do not imply interpolation.

Appendix D

An Analysis of Taxlot Density to Understand Rural Population Density

An analysis of taxlot densities was performed for the GWMA to gain a better understanding of rural population densities in the three GWMA counties and different geologic units. Understanding rural population densities yields a more informed regional understanding for septic loading in different areas. A principal assumption of this analysis is that rural taxlot size correlates to population density, with smaller taxlots (often associated with houses) being associated with higher population densities, while large taxlots (often associated with fields) are associated with lower densities. An example of this is shown in Figures D1 and D2.

Results of the analysis, shown in Table D1, indicate that for rural areas (rural being defined as regions outside of city limits) in the GWMA, taxlot densities are greater over the Willamette aquifer (Qal of Gannett and Caldwell (1998)) than the Willamette silt (Qs). Additionally, taxlot densities are much lower in Linn County than either Lane or Benton, indicating that impacts on groundwater nitrate due to wide-spread septic loading is less threatening in Linn County than other counties. Lane County was found to have a disproportionately high taxlot density overlying the unconfined Willamette aquifer, which could be manifested by the high groundwater concentrations observed near Coburg and Junction City. Relatively low taxlot densities were observed for the Willamette silt, except in Benton County, where higher densities are associated with housing on the southern outskirts of Corvallis.

Figure D1. Satellite photo of a rural area in the GWMA located in Lane County.

Figure D2. Taxlots of the same region shown in Figure D1. Note that houses in the satellite photo are generally on smaller taxlots.

Table D1. Rural taxlot densities for different counties and geologic units within the GWMA. Qal is the Willamette aquifer and Qs is the Willamette silt, with both geologic units defined by Gannett and Caldwell (1998).

Rural Taxlot Densities by Geology

	Qal	Qs
Rural Area (km^2)	234.67	299.53
Rural Taxlots	2952	3036
Rural Taxlot Size (acres)	19.64	24.38
Rural Taxlot Size (km^2)	0.08	0.10
Taxlots/km^2	12.58	10.14

# of Rural Taxlots per County & Avg Taxlot Density			
	Lane	Benton	Linn
# Taxlots	2592	2232	1164
Rural Area (km^2)	176.59	159.51	197.6
Taxlots/km^2	14.68	13.99	5.89

# of Rural Taxlots per County & Avg Taxlot Density on Qal			
	Lane	Benton	Linn
# Taxlots	1151	1484	317
Rural Area (km^2)	66.08	133.74	36.5
Taxlots/km^2	17.42	11.10	8.68

# of Rural Taxlots per County & Avg Taxlot Density on Qs			
	Lane	Benton	Linn
# Taxlots	1441	748	847
Rural Area (km^2)	110.51	25.77	161.1
Taxlots/km^2	13.04	29.03	5.26

Appendix E

SWAT Land Use Assignments, New Crops, and Crop Values Calibrated

Land use assignments applied in SWAT for present and future scenarios are presented in this appendix. New crops created for the SWAT database and their input parameters are also included.

Table E1 displays all land use/land cover (LULC) assignments used by Hulse et al. (2002) for the Willamette Valley. Additionally, the SWAT crop and management scenario assigned to each LULC class is presented. The assigned values apply to the present (mid-1990s), Development 2050, Conservation 2050, and Plan Trend 2050 LULC maps used in SWAT. The mid-1990s LULC map was used in SWAT for the present (instead of a map for the year 2000) because most of the available nitrate leaching data is from the 1990s and because groundwater age dating in the SWV indicates that groundwater nitrate concentrations reflect land management from the 1990s and earlier (Craner, 2006). Therefore, since the available soil nitrate and groundwater nitrate values reflect land use from the 1990s or earlier, a mid-1990s LULC map was used. A sensitivity analysis using a LULC map for the year 2000 was not performed due to constraints on time and resources.

Crop assignments on Table E1 are in some cases misleading (such as OATS being assigned for Oak Savanna), but in SWAT the LULC classes can be reassigned in the management scenario editor. Therefore, the actual crops being harvested for a particular land use are those listed in the management scenario column.

Error propagation from the LULC maps are thought to have affected outputs of the SWAT scenarios. Notably, most SWV crops have between 50 and 88% identification accuracy (based on Landsat identifications versus ground-truth observations) (Hulse et al., 2002). Based on the regional scope of this study and available data sources, no alternatives existed to using the Landsat LULC maps.

Table E2 shows new SWAT land use classes created based on modifications of pre-existing land use classes, while Table E3 shows input values and sources used for creating the peppermint land use class for SWAT.

Additionally, three SWAT crop parameters were slightly modified during calibration. Nitrogen uptake parameters (BN1, BN2, and BN3) were adjusted for tall fescue (0.0560, 0.0252, and 0.0144), annual ryegrass (0.0660, 0.0304, and 0.0176) and peppermint (0.0470, 0.0220, and 0.0187). For fescue and ryegrass, BN2 and BN3 were increased by 20% of their database values, while all peppermint values were increased by 10%. Nitrogen uptake parameters were increased to more accurately reflect nitrate leaching rates in the SWV.

Table E1. Modified Land use/land cover (LULC) legend from Hulse et al. (2002) for mid-1990s LULC map and the three future scenarios. Assigned SWAT land use classes for each LULC map class are indicated as well as the management scenarios applied. All management scenarios (other than those listed as "Default" or "no HRUs") are presented in Appendix F. "Default" management scenarios simulate growth according to the default management scenario assigned to it by SWAT. Default scenarios were generally used in non-agrarian regions. The management scenario "no HRUs" indicates that a particular LULC class is present in the SWV, but accounted for less than 5% of the total area in any subbasin and thus does not compose an HRU.

SWAT CLASS	DATA VALU	CATEGORY NAME	SWAT MANAGEMENT SCENARIO
URMD	1	Residential 0-4 dwelling unit/ac	Default
URMH	2	Residential 4-9 dwelling unit/ac	Default
URHD	3	Residential 9-16 dwelling unit/ac	Default
URHD	4	Residential >16 dwelling unit/ac	Default
none	5	Vacant	none
UCOM	6	Commercial	Default
UCOM	7	Comm/Industrial	Default
UIDU	8	Industrial	Default
none	9	Industrial & Comm.	none
none	10	Residential & Comm.	none
UINS	11	Urban non-vegetated unknown	Default
URLD	16	Rural structures	Default
URTN	18	Railroad	Default
URTN	19	Primary roads	Default
URTN	20	Secondary roads	Default
URTN	21	Light duty roads	Default
UTRN	24	Rural non-vegetated unknown	Default
WETN	29	Main channel non-vegetated	no HRUs
WATR	32	Stream orders 5-7	Default
WATR	33	Permanent lentic water	Default
FRST	39	Topographic Shadow	Default
none	40	Snow	none
none	42	Barren	none
UINS	49	Urban tree overstory	Default
FRSE	51	Upland Forest open	Default
FRST	52	Upland Forest Semi-closed mixed	Default
FRSD	53	Forest Closed hardwood	Default
FRST	54	Forest Closed mixed	Default
FRSE	55	Upland Forest Semi-closed conifer	Default
FRSE	56	Conifers 0-20 yrs	Default
FRSE	57	Forest closed conifer 21-40 yrs	Default
FRSE	58	Forest closed conifer 41-60 yrs.	Default
FRSE	59	Forest closed conifer 61-80 yrs	Default
FRSE	60	Forest closed conifer 81-200 yrs	Default
FRSE	61	Forest closed conifer older than 200 yrs	Default
FRSD	62	Upland Forest Semi-closed hardwood	Default

Table E1. (Continued)

POPL	66	Hybrid poplar	Default
RYEG	67	Grass seed rotation	Ryegrass
CUCM	68	Irrigated annual rotation	Row Rotation
WWHT	71	Grains	Wheat Rotation
POPL	72	Nursery	Default
STRW	73	Berries & Vineyards	no HRUs
CLVR	74	Double cropping	Field Crop Rotation
AGRC	75	Hops	Default
PPMT	76	Mint	Peppermint
none	77	Radish seed	none
SBFS	78	Sugar beet seed	Row Rotation
AGRR	79	Row crop	Row Rotation
FESC	80	Grass	Tall Fescue
RYEG	81	Burned grass	Italian (Annual) Ryegrass
CLVR	82	Field crop	Field Crop Rotation
SWGR	83	Hayfield	Tall Fescue
CANA	84	Late field crop	no HRUs
WPAS	85	Pasture	Tall Fescue
NATG	86	Natural grassland	Default
NATS	87	Natural shrub	Default
AGRC	88	Bare/fallow	Default
WETL	89	Flooded/marsh	Default
PPMT	90	Irrigated perennial	Peppermint
BROC	91	Turfgrass	no HRUs
ORCD	92	Orchard	Orchard
FRSE	93	Christmas trees	Default
FRSE	95	Conifer Woodlot	Default
OATS	98	Oak savanna	Default, growing as Nat. Grassland
none	101	Wet shrub	none
none	102	Unknown	none

Table E2. New land use classes created for the Southern Willamette Valley by modifying SWAT database land use classes.

New Land Use Classes Created from SWAT Database Land Use Classes			
New Land Use Database Entry	Parent Land Use^	Changes	Source
Urban Residential Medium-High (URMH)	Urban Medium Density (URMD), Urban High Density (URHD)	Averaged URMD and URHD. URMD is 1-4 unit per acre, URHD is > 8 unit/acre. Needed a 4-8 unit/acre land use class.	Neitsch et al., 2002a
Sugarbeet for Seed (SBFS)	Sugarbeet (SGBT)	IDC*: cool season annual HVSTI*: 0.75 CHTMX*: 2.2m WSYF*: 0.74	Estimate Estimate Estimate Estimate
Natural Grass (NATG)	Slender Wheatgrass (SWGR)	Slender wheatgrass is a natural grass of the Willamette Valley (Daris, 2003). CNs for slender wheatgrass were changed to grassland with good cover values.	Neitsch et al., 2002a
Natural Shrub (NATS)	Slender Wheatgrass (SWGR)	Slender wheatgrass is a natural grass of the Willamette Valley (Daris, 2003). LAI was estimated to be 3.3 (down from 4.0) and the CN values used were those for brush with fair cover.	Neitsch et al., 2002a

^ "Parent Land Use" refers to the original SWAT land use class which was modified to create the new land use class
*Definitions of SWAT input parameters can be found in Table 3 and in Neitsch et al. (2002a) and Neitsch et al. (2002b).

Table E3. Input crop values used for peppermint. Further information on input parameters can be found in Neitsch et al. (2002a) and Neitsch et al. (2002b).

Input Parameter	Units	Definition	Input Value	Source
CPNM	-	Four character code to represent the land cover/plant name.	PPMT	
IDC	-	Land Cover/Plant Classification.	Perennial	Hackett and Carolane, 1982
BIO_E	[(kg/ha)/(MJ/m^2)]	Biomass/Energy Ratio.	10.2	US Bureau of Reclamation, 2005; Sullivan et al., 1999
HVSTI	(kg/ha)/(kg/ha)	Harvest index.	0.95	estimate
BLAI	(m^2/m^2)	Max leaf area index.	5	Singh et al., 1989
FRGRW1	(fraction)	Fraction of the plant growing season corresponding to the 1st. Point on the optimal leaf area development curve.	0.1	Sullivan et al., 1999
LAIMX1	(fraction)	Fraction of the max. leaf area index corresponding to the 1st. point on the optimal leaf area development curve.	0.1	Sullivan et al., 1999
FRGRW2	(fraction)	Fraction of the plant growing season corresponding to the 2nd. point on the optimal leaf area development curve.	0.9	Sullivan et al., 1999
LAIMX2	(fraction)	Fraction of the max. leaf area index corresponding to the 2nd. point on the optimal leaf area development curve.	0.9	Sullivan et al., 1999
DLAI	(heat units/heat units)	Fraction of growing season when leaf area starts declining.	1	Mellbye and Hart, personal communication 2005
CHTMX	(m)	Max canopy height.	0.75	estimate
RDMX	(m)	Max root depth.	0.61	Smesrud et al., 1997
T_OPT	(deg C)	Optimal temp for plant growth.	15	Hackett and Carolane, 1982
T_BASE	(deg C)	Min temp plant growth.	5	Hackett and Carolane, 1982
CNYLD	[kg N/kg seed]	Fraction of nitrogen in yield .	0.0166	Sullivan et al., 1999
CPYLD	[kg P/kg seed]	Fraction of phosphorus in yield.	0.0005	estimate
BN1	[kg N/kg biomass]	Fraction of N in plant at emergence .	0.0425	Sullivan et al., 1999
BN2	[kg N/kg biomass]	Fraction of N in plant at 0.5 maturity.	0.02	Sullivan et al., 1999
BN3	[kg N/kg biomass]	Fraction of N in plant at maturity.	0.017	Sullivan et al., 1999
BP1	[kg P/kg biomass]	Fraction of P at emergence.	0.0050	estimate
BP2	[kg P/kg biomass]	Fraction of P at 0.5 maturity.	0.0010	estimate
BP3	[kg P/kg biomass]	Fraction of P at maturity.	0.0007	estimate
WSYF	[(kg/ha)/(kg/ha)]	Lower limit of harvest index.	0.95	estimate
USLE_C	-	Min value of USLE C factor applicable to the land cover/plant.	0.2	database estimate*
GSI	(m/s)	Max stomatal conductance (in drough condition).	0.0080	database estimate*
VPDFR	(kPa)	Vapor pressure deficit corresponding to the fraction maximum stomatal conductance defined by FRGMAX	4	default^

Table E3. (Continued)

FRGMAX	(fraction)	Fraction of maximum stomatal conductance that is achievable at a high vapor pressure deficit.	0.75	default^
WAVP	[rate]	Rate of decline in radiation use efficiency per unit increase in vapor pressure deficit.	5.5	database estimate*
CO2HI	(microliter/liter)	Elevated CO2 atmospheric concentration.	660	default^
BIOEHI	(ratio)	Biomass-energy ratio corresponding to the 2nd. point on the radiation use efficiency curve. Only used for climate change studies.	20	database estimate*
RSDCO_PL	(fraction)	Plant residue decomposition coefficient.	0.05	default^
Cropname	-	Crop description name.	Peppermint	
CN2	-	SCS runoff curve number for moisture condition II.	67A, 78B, 85C, 89D	Neitsch et al., 2002a
OV_N	-	Manning's "n" value for overland flow.	0.12	Neitsch et al., 2002a
FERTFIELD	-	If checked this crop is going to be fertilized.	Yes	Sullivan et al., 1999

^ Default values are those listed in the SWAT database (Neitsch et al., 2002a) which have the same values for all crops.
* Database estimate are values estimated from the SWAT database (Neitsch et al., 2002a) for crops similar to peppermint.

Appendix F

SWAT Land Management Scenarios

Management scenarios and Groundwater Best Management Practices (GW-BMP) management scenarios used in SWAT simulations are presented in this appendix. Values presented in this appendix are not intended to be definitive on agricultural land practices in SWV, as they are mostly based on a literature review and communication with agricultural extension agents of the Oregon State University Extension Service. Land management scenarios reflect approximate fertilizer rates, irrigation rates, and field management dates for the 1990s and early 2000s. These management scenarios were applied to both present and future SWAT scenarios, and thus future scenarios do not account for higher agricultural efficiencies that may develop by 2050.

In the proceeding tables, aerial percentages of each management scenario are given from the present scenario to help readers assess the relative importance of each management scenario. "Grower" rates reflect those applied in non-GW-BMP scenarios, while "Extension" rates are used in GW-BMP scenarios. The general difference between Grower and Extension rates is that farmers apply 13% more fertilizer and irrigation in Grower scenarios than in Extension scenarios. However, tall fescue fertilization Grower and Extension rates are defined by agricultural extension agents (see Table F1) and do not use a 13% difference between the scenarios.

GW-BMPs were not applied for urban lawn management scenarios because of a high level of uncertainty associated with the modeled urban nitrogen loading. Fertilizer and irrigation values applied in the urban management scenario were estimated by weighting different OSU Extension application rates by lawn care statistics reported by Nielson and Smith (2005) for the Tualitin Watershed (located in the Northern Willamette Valley). Nielson and Smith's study indicates that only 20% of homeowners surveyed fertilize based on professional advice or on product labels, so although reasonable assumptions were applied for urban nitrogen loading, high uncertainty is thought to be associated with fertilizer and irrigation rates.

The SWAT model was run for 15 years for all present and future scenarios, and thus each management scenario repeats itself several times throughout a simulation.

Table F1. Tall fescue management scenario. Values are based on Cross et al. (1992) and Mellbye and Hart (personal communication 2005).

TALL FESCUE: FESC 0.40%, WPAS 7.71%, SWGR 5.70% = 13.81%

				Extension (BMP) Fertilizer Rates			Grower Fertilizer Rates	
Year	Date	Operation	Fraction N	kg/ha	kg N/ha	lbsN/a	kg/ha	lbsN/a
1	1-Mar	Plant Tall Fescue						
	1-Mar	Fertilizer with Planting, 46-0-0, 97.7 kg/ha	0.46	97.70	44.94	40	97.70	40
	1-Jul	Harvest Only						
	15-Sep	Fertilize, Fall, 15-15-15, 262 kg/ha	0.15	299.63	44.94	40	374.53	50
	1-Nov	End of Growing Season						
2	1-Feb	Start of Growing Season, Tall Fescue						
	10-Mar	Fertilizer with 33-0-0, 170 kg/ha	0.33	180.46	59.55	53	255.36	75
	20-Apr	Fertilizer with 33-0-0, 170 kg/ha	0.33	180.46	59.55	53	255.36	75
	7-Jul	Harvest Only						
	15-Sep	Fertilize, Fall, 15-15-15, 262 kg/ha	0.15	299.63	44.94	40	374.53	50
	1-Nov	End of Growing Season						
3	1-Feb	Start of Growing Season, Tall Fescue						
	10-Mar	Fertilizer with 33-0-0, 170 kg/ha	0.33	180.46	59.55	53	255.36	75
	20-Apr	Fertilizer with 33-0-0, 170 kg/ha	0.33	180.46	59.55	53	255.36	75
	7-Jul	Harvest Only						
	15-Sep	Fertilize, Fall, 15-15-15, 262 kg/ha	0.15	299.63	44.94	40	374.53	50
	1-Nov	End of Growing Season						
4	1-Feb	Start of Growing Season, Tall Fescue						
	10-Mar	Fertilizer with 33-0-0, 170 kg/ha	0.33	180.46	59.55	53	255.36	75
	20-Apr	Fertilizer with 33-0-0, 170 kg/ha	0.33	180.46	59.55	53	255.36	75
	7-Jul	Harvest Only						
	15-Sep	Fertilize, Fall, 15-15-15, 262 kg/ha	0.15	299.63	44.94	40	374.53	50
	1-Nov	End of Growing Season						

Table F1. (Continued)

5	1-Feb	Start of Growing Season, Tall Fescue						
	10-Mar	Fertilizer with 33-0-0, 170 kg/ha	0.33	180.46	59.55	53	255.36	75
	20-Apr	Fertilizer with 33-0-0, 170 kg/ha	0.33	180.46	59.55	53	255.36	75
	7-Jul	Harvest and Kill						
	20-Sep	Tillage, Moldboard Plow						
	1-Oct	Tillage, Single Disk						
	5-Oct	Tillage, Roller Harrow						
	8-Oct	Tillage, Roller Harrow						
	11-Oct	Tillage, Roller Harrow						

Table F2. Ryegrass management scenario. Note that the February 15th planting date used in year 1 is unrealistic, but was used to approximate a full year of production for the crop of years 5 and 1. All other values are based on Mellbye et al. (2003) and Mellbye and Hart (personal communication 2005).

ANNUAL (ITALIAN) RYEGRASS: RYEG, 24.68%

Models "GRASS SEED ROTATION" (RYEG 67) and "BURNED GRASS" (RYEG 81) land uses

				Extension (BMP) Fertilizer Rates			Grower Fertilizer Rates	
Year	Date	Operation	Fraction N	kg/ha	kg N/ha	lbsN/a	kg/ha	lbsN/a
1	15-Feb	Plant Italian (Annual) Ryegrass						
	1-Apr	Spring Fertilize, 46-0-0, 293 kg/ha	0.46	293	134.78	119.95	331.09	135.55
	7-Jul	Harvest and Kill						
	1-Sep	Tillage, Single Disk						
	8-Sep	Tillage, Moldboard Plow						
	18-Sep	Tillage, Harrow (tines)						
	18-Sep	Tillage, Seedbed roller						
	19-Sep	Tillage, Harrow (tines)						
	19-Sep	Tillage, Seedbed roller						
	21-Sep	Plant Italian (Annual) Ryegrass						
	21-Sep	Fall Fertilize, 15-15-15, 149.8 kg/ha	0.15	149.8	22.47	20.00	169.27	22.60
2	1-Apr	Spring Fertilize, 46-0-0, 293 kg/ha	0.46	293	134.78	119.95	331.09	135.55
	7-Jul	Harvest and Kill						
	1-Sep	Tillage, Single Disk						
	8-Sep	Tillage, Moldboard Plow						
	18-Sep	Tillage, Harrow (tines)						
	18-Sep	Tillage, Seedbed roller						
	19-Sep	Tillage, Harrow (tines)						
	19-Sep	Tillage, Seedbed roller						
	21-Sep	Plant Italian (Annual) Ryegrass						
	21-Sep	Fall Fertilize, 15-15-15, 149.8 kg/ha	0.15	149.8	22.47	20.00	169.27	22.60
3	1-Apr	Spring Fertilize, 46-0-0, 293 kg/ha	0.46	293	134.78	119.95	331.09	135.55

Table F2. (Continued)

	7-Jul	Harvest and Kill						
	1-Sep	Tillage, Single Disk						
	8-Sep	Tillage, Moldboard Plow						
	18-Sep	Tillage, Harrow (tines)						
	18-Sep	Tillage, Seedbed roller						
	19-Sep	Tillage, Harrow (tines)						
	19-Sep	Tillage, Seedbed roller						
	21-Sep	Plant Italian (Annual) Ryegrass						
	21-Sep	Fall Fertilize, 15-15-15, 149.8 kg/ha	0.15	149.8	22.47	20.00	169.27	22.60
4	1-Apr	Spring Fertilize, 46-0-0, 293 kg/ha	0.46	293	134.78	119.95	331.09	135.55
	7-Jul	Harvest and Kill						
	1-Sep	Tillage, Single Disk						
	8-Sep	Tillage, Moldboard Plow						
	18-Sep	Tillage, Harrow (tines)						
	18-Sep	Tillage, Seedbed roller						
	19-Sep	Tillage, Harrow (tines)						
	19-Sep	Tillage, Seedbed roller						
	21-Sep	Plant Italian (Annual) Ryegrass						
	21-Sep	Fall Fertilize, 15-15-15, 149.8 kg/ha	0.15	149.8	22.47	20.00	169.27	22.60
5	1-Apr	Spring Fertilize, 46-0-0, 293 kg/ha	0.46	293	134.78	119.95	331.09	135.55
	7-Jul	Harvest and Kill						
	1-Sep	Tillage, Single Disk						
	8-Sep	Tillage, Moldboard Plow						
	18-Sep	Tillage, Harrow (tines)						
	18-Sep	Tillage, Seedbed roller						
	19-Sep	Tillage, Harrow (tines)						
	19-Sep	Tillage, Seedbed roller						
	21-Sep	Plant Italian (Annual) Ryegrass						
	21-Sep	Fall Fertilize, 15-15-15, 149.8 kg/ha	0.15	149.8	22.47	20.00	169.27	22.60
	31-Dec	Kill Crop						

Table F3. Wheat management scenario. Values are based on Steiner and Karow (1997), Mellbye and Hart (personal communication 2005), Hart et al. (2000), and Mellbye et al. (2003).

Wheat Rotation: WWHT, 6.28%

Year	Date	Operation	Fraction N	Extension (BMP) Fertilizer Rates			Grower Fertilizer Rates	
				kg/ha	kg N/ha	lbsN/a	kg/ha	lbsN/a
1	3-Apr	Tillage, Offset Disk (light-duty)						
	4-Apr	Plant Winter Wheat						
	4-Apr	Fertilizer, 15-15-15, 149.8 kg/ha	0.15	149.8	22.47	20.00	169.27	22.60
	4-Apr	Tillage, Harrow (Tines)						
	5-Apr	Tillage, Cultipacker						
	6-Apr	Tillage, Harrow (Tines)						
	27-Apr	Fertilize, 46-0-0, 194 kg/ha	0.46	194	89.24	79.42	219.22	89.75
	7-Aug	Harvest & Kill						
	1-Sep	Tillage, Single Disk						
	8-Sep	Tillage, Moldboard Plow						
	18-Sep	Tillage, Harrow (tines)						
	18-Sep	Tillage, Seedbed roller						
	19-Sep	Tillage, Harrow (tines)						
	19-Sep	Tillage, Seedbed roller						
	21-Sep	Plant Italian (Annual) Ryegrass						
	21-Sep	Fall Fertilize, 15-15-15, 149.8 kg/ha	0.15	149.8	22.47	20.00	169.27	22.60
2	1-Apr	Spring Fertilize, 46-0-0, 293 kg/ha	0.46	293	134.78	119.95	331.09	135.55
	7-Jul	Harvest and Kill						
	1-Aug	Tillage, Offset Disk (light-duty)						
	15-Oct	Plant Winter Wheat						
	15-Oct	Fertilize, 46-0-0, 61 kg/ha	0.46	61	28.06	24.97	68.93	28.22
	15-Oct	Fertilize, 0-15-0, total of 44.9 kg P2O5/ha	0.15	299.3	44.90*	39.96*	338.21	45.15*
	16-Oct	Tillage, Harrow (Tine)						

Table F3. (Continued)

3	10-Mar	Fertilizer, Elemental N, 180 kg/ha	1	180	180.00	160.20	203.40	181.03
	7-Aug	Harvest & Kill						
	1-Sep	Tillage, Single Disk						
	8-Sep	Tillage, Moldboard Plow						
	18-Sep	Tillage, Harrow (tines)						
	18-Sep	Tillage, Seedbed roller						
	19-Sep	Tillage, Harrow (tines)						
	19-Sep	Tillage, Seedbed roller						
	21-Sep	Plant Italian (Annual) Ryegrass						
	21-Sep	Fall Fertilize, 15-15-15, 149.8 kg/ha	0.15	149.8	22.47	20.00	169.27	22.60
4	1-Apr	Spring Fertilize, 46-0-0, 293 kg/ha	0.46	293	134.78	119.95	331.09	135.55
	7-Jul	Harvest and Kill						
	1-Aug	Tillage, Offset Disk (light-duty)						
	15-Oct	Plant Winter Wheat						
	15-Oct	Fertilize, 46-0-0, 61 kg/ha	0.46	61	28.06	24.97	68.93	28.22
	15-Oct	Fertilize, 0-15-0, total of 44.9 kg P2O5/ha	0.15	299.3	44.90*	39.96*	338.21	45.15*
	16-Oct	Tillage, Harrow (Tine)						
	31-Dec	Kill Wheat						

*Values are for P2O5, not N

Table F4. Field crop rotation management scenario**.** This rotation plants wheat and ryegrass in opposite years of the "Wheat Rotation" scenario (Table F3) to prevent annual basin-wide leaching differences caused by the same rotation pattern being applied across the SWV. Note that the February 15th planting date used in year 1 is unrealistic, but was used to approximate a full year of production for the crop of years 4 and 1. All other values are based on Mellbye and Hart (personal communication 2005), Hart et al. (2000), and Mellbye et al (2003).

Field Crop Rotation: CLVR 0.34% + AGRC 0.02% = 0.36%

				Extension (BMP) Fertilizer Rates			Grower Fertilizer Rates	
Year	Date	Operation	Fraction N	kg/ha	kg N/ha	lbsN/a	kg/ha	lbsN/a
1	15-Feb	Plant Italian (Annual) Ryegrass						
	1-Apr	Spring Fertilize, 46-0-0, 293 kg/ha	0.46	293	134.78	119.95	331.09	135.55
	7-Jul	Harvest and Kill Annual Ryegrass						
	1-Aug	Tillage, Offset Disk (light-duty)						
	15-Oct	Plant Winter Wheat						
	15-Oct	Fertilize, 46-0-0, 61 kg/ha	0.46	61	28.06	24.97	68.93	28.22
	15-Oct	Fertilize, 0-15-0, total of 44.9 kg P2O5/ha	0.15	299.3	44.90*	39.96*	338.21	45.15*
	16-Oct	Tillage, Harrow (Tine)						
2	10-Mar	Fertilizer, Elemental N, 180 kg/ha	1	180	180.00	160.20	203.40	181.03
	7-Aug	Harvest & Kill Wheat						
	1-Sep	Tillage, Single Disk						
	8-Sep	Tillage, Moldboard Plow						
	18-Sep	Tillage, Harrow (tines)						
	18-Sep	Tillage, Seedbed roller						
	19-Sep	Tillage, Harrow (tines)						
	19-Sep	Tillage, Seedbed roller						
	21-Sep	Plant Italian (Annual) Ryegrass						
	21-Sep	Fall Fertilize, 15-15-15, 149.8 kg/ha	0.15	149.8	22.47	20.00	169.27	22.60
3	1-Apr	Spring Fertilize, 46-0-0, 293 kg/ha	0.46	293	134.78	119.95	331.09	135.55
	7-Jul	Harvest and Kill Annual Ryegrass						
	1-Aug	Tillage, Offset Disk (light-duty)						
	15-Oct	Plant Winter Wheat						
	15-Oct	Fertilize, 46-0-0, 61 kg/ha	0.46	61	28.06	24.97	68.93	28.22
	15-Oct	Fertilize, 0-15-0, total of 44.9 kg P2O5/ha	0.15	299.3	44.90*	39.96*	338.21	45.15*
	16-Oct	Tillage, Harrow (Tine)						

Table F4. (Continued)

4	10-Mar	Fertilizer, Elemental N, 180 kg/ha	1	180	180.00	160.20	203.40	181.03
	7-Aug	Harvest & Kill WWHT						
	1-Sep	Tillage, Single Disk						
	8-Sep	Tillage, Moldboard Plow						
	18-Sep	Tillage, Harrow (tines)						
	18-Sep	Tillage, Seedbed roller						
	19-Sep	Tillage, Harrow (tines)						
	19-Sep	Tillage, Seedbed roller						
	21-Sep	Plant Italian (Annual) Ryegrass						
	21-Sep	Fall Fertilize, 15-15-15, 149.8 kg/ha	0.15	149.8	22.47	20.00	169.27	22.60
	31-Dec	Kill Annual Rye						

*Values are for P2O5, not N

Table F5. Orchard management scenario**.** Values are based on Rackham (1996), Olson (2001), and McGrath (personal communication 2005).

ORCHARD: ORCD 0.23%

				Extension (BMP) Fertilizer & Irrigation Rates				Grower Rates		
Year	Date	Operation	Fraction N	kg/ha	kg N/ha	lbsN/a	Irrigation	kg/ha	lbsN/a	Irrigation (mm)
1	1-Feb	Plant Orchard/begin growing season								
	13-May	43.4 mm irrigation					43.4			49.04
	5-Jun	Fertilize, Urea	0.46	291.30	134	119.26		329.17	134.76	
	5-Jun	54 mm irrigation					54			61.02
	27-Jun	54 mm irrigation					54			61.02
	18-Jul	84mm irrigation					84			94.92
	19-Jul	84mm irrigation					84			94.92
	9-Aug	66mm irrigation					66			74.58
	30-Aug	66mm irrigation					66			74.58
	19-Sep	17.5mm irrigation					17.5			19.78
	25-Sep	Harvest Only								
	31-Oct	End of Growing Season								

Table F6. Peppermint management scenario (without GW-BMPs). Values based on Mellbye and Hart (personal communication 2005), Mitchell (1997), Taylor et al. (1992), Smesrud et al. (1997), and Hart et al. (2000).

PEPPERMINT: PPMT (1.02%), modeling peppermint and irrigated perennial

				Grower Rates			
Year	Date	Operation	Fraction N	kg/ha	kg N/ha	lbsN/a	Irrigation (mm)
1	1-Sep	Tillage: Moldboard plow					
	10-Sep	Tillage: Harrow (tines)					
	20-Sep	Tillage: Harrow (tines)					
	14-Oct	Plant PPMT					
	14-Oct	Fert 10-20-20 @ 226 lbs/a	0.1	253.93	25.39	22.60	
	14-Oct	Irrigate 1.13" (28.7 mm)					28.70
	21-Oct	Tillage: marker (cultivator)					
	21-Oct	Irrigate 1.13" (28.7 mm)					28.70
	25-Oct	Tillage: marker (cultivator)					
	30-Oct	Tillage: marker (cultivator)					
2 through 5	1-Apr	Fertilize: 15-15-15 @ 226 lbs/a	0.15	253.93	38.09	33.90	
	7-Apr	.71" or 18.08 mm					18.08
	14-Apr	.71" or 18.08 mm					18.08
	21-Apr	.71" or 18.08 mm					18.08
	28-Apr	.71" or 18.08 mm					18.08
	5-May	Fertilize: 113 lbs N/a of 46-0-0	0.46	276.01	126.97	113.00	
	5-May	.95" or 24.7 mm					24.07
	12-May	.95" or 24.7 mm					24.07
	19-May	.95" or 24.7 mm					24.07
	26-May	.95" or 24.7 mm					24.07
	2-Jun	Fertilize: 113 lbs N/a of 46-0-0	0.46	276.01	126.97	113.00	
	2-Jun	1.58" or 40.23 mm					40.23
	9-Jun	1.58" or 40.23 mm					40.23
	16-Jun	1.58" or 40.23 mm					40.23
	23-Jun	1.58" or 40.23 mm					40.23
	1-Jul	1.66" or 42.15 mm					42.15
	8-Jul	1.66" or 42.15 mm					42.15
	15-Jul	1.66" or 42.15 mm					42.15
	23-Jul	1.66" or 42.15 mm					42.15
	29-Jul	1.66" or 42.15 mm					42.15
	1-Aug	Harvest Only					
	4-Aug	Fertilize: 22.6 lbs N/a of 46-0-0	0.46	55.20	25.39	22.60	
	4-Aug	1.10" or 28.02 mm					28.02
	20-Aug	1.10" or 28.02 mm					28.02
	7-Sep	1.10" or 28.02 mm					28.02
	23-Sep	1.10" or 28.02 mm					28.02

Table F6. (Continued)

6	1-Apr	Fertilize: 15-15-15 at 226 lbs/a	0.15	253.93	38.09	33.90	
		Irrigation and Fertilzation as in yr 2					
	1-Aug	Harvest and Kill					
	5-Aug	Tillage, Offset Disk (light-duty)					
	15-Oct	Plant Winter Wheat					
	15-Oct	Fertilize, 46-0-0, 25.39 kg/ha	0.46	55.20	25.39	22.60	
	15-Oct	Fertilize, 0-15-0 at 338.21 kg/ha	0.15	338.21	50.73*	45.15*	
	16-Oct	Tillage, Harrow (Tine)					
	31-Dec	Kill Wheat					

*Values are for P2O5, not N

Table F7. Peppermint GW-BMP management scenario. Not only were peppermint fertilization and irrigation rates reduced to recommended extension rates, wheat is planted as a cover crop in year 6 (instead of as a harvested crop). Values are based on Mellbye and Hart (personal communication 2005), Mitchell (1997), Taylor et al. (1992), Smesrud et al. (1997), and McGrath (personal communication 2005).

PEPPERMINT: PPMT (1.02%), modeling peppermint and irrigated perennial

				Extension (BMP) Fertilizer & Irrigation Rates			
Year	Date	Operation	Fraction N	kg/ha	kg N/ha	lbsN/a	Irrigation (mm)
1	1-Sep	Tillage: Moldboard plow					
	10-Sep	Tillage: Harrow (tines)					
	20-Sep	Tillage: Harrow (tines)					
	14-Oct	Plant PPMT					
	14-Oct	Fert 10-20-20 @ 200 lbs/a	0.1	224.72	22.47	20	
	14-Oct	Irrigate 1" (25.4 mm)					25.4
	21-Oct	Tillage: marker (cultivator)					
	21-Oct	Irrigate 1" (25.4 mm)					25.4
	25-Oct	Tillage: marker (cultivator)					
	30-Oct	Tillage: marker (cultivator)					
2 through 5	1-Apr	Fertilize: 15-15-15 at 200 lbs/a	0.15	224.72	33.71	30	
	7-Apr	.63" or 16 mm					16
	14-Apr	.63" or 16 mm					16
	21-Apr	.63" or 16 mm					16
	28-Apr	.63" or 16 mm					16
	5-May	Fertilize: 100 lbs N/a of 46-0-0	0.46	244.26	112.36	100	
	5-May	.838" or 21.3 mm					21.3
	12-May	.838" or 21.3 mm					21.3
	19-May	.838" or 21.3 mm					21.3
	26-May	.838" or 21.3 mm					21.3
	2-Jun	Fertilize: 100 lbs N/a of 46-0-0	0.46	244.26	112.36	100	
	2-Jun	1.40" or 35.6 mm					35.6
	9-Jun	1.40" or 35.6 mm					35.6
	16-Jun	1.40" or 35.6 mm					35.6
	23-Jun	1.40" or 35.6 mm					35.6
	1-Jul	1.47" or 37.3 mm					37.3
	8-Jul	1.47" or 37.3 mm					37.3
	15-Jul	1.47" or 37.3 mm					37.3
	23-Jul	1.47" or 37.3 mm					37.3
	29-Jul	1.47" or 37.3 mm					37.3
	1-Aug	Harvest Only					
	4-Aug	Fertilize: 20 lbs N/a of 46-0-0	0.46	48.85	22.47	20	
	4-Aug	.978" or 24.8 mm					24.8
	20-Aug	.978" or 24.8 mm					24.8
	7-Sep	.978" or 24.8 mm					24.8
	23-Sep	.978" or 24.8 mm					24.8

Table F7. (Continued)

6	REPEAT YEAR 2 UP UNITL AUG						
	1-Aug	Harvest and Kill					
	5-Aug	Tillage, Offset Disk (light-duty)					
	15-Oct	Plant Winter Wheat					
	31-Dec	Kill Wheat					

Table F8. Row crop management scenario (without GW-BMPs). Values adapted from Steiner and Karow (1997), Hart et al. (2000), Lisec et al. (1995), Smesrud et al. (1997), Cuenca et al. (1992), Cross et al. (1988), Draycott and Christenson (2003), and Cooke and Scott (1993).

Row Rotation: CUCM (3.43%) + AGRR (0.03%)

				Grower Rates			
Year	Date	Operation	Fraction N	kg/ha	kg N/ha	lbsN/a	Irrigation (mm)
1	3-Apr	Tillage, Offset Disk (light-duty)					
	4-Apr	Plant Winter Wheat					
	4-Apr	Fertilizer, 15-15-15, 169.27 kg/ha	0.15	169.27	25.39	22.60	
	4-Apr	Tillage, Harrow (Tines)					
	5-Apr	Tillage, Cultipacker					
	6-Apr	Tillage, Harrow (Tines)					
	27-Apr	Fertilize, 46-0-0, 219.22 kg/ha	0.46	219.22	100.84	89.75	
	7-Aug	Harvest & Kill					
	15-Aug	Tillage: Moldboard plow					
	22-Aug	Tillage: Offset Disc					
	27-Aug	Tillage:Harrow (tines)					
	27-Aug	Tillage: seedbed roller					
	30-Aug	Fertilizer: 130 kg/ha P2O5	0.52	250	130.00*	115.70*	
		N Applied in 11-52-0-->	0.11		27.50	24.48	
	30-Aug	Fertilizer, 46-0-0, 92.39 kg/ha	0.46	92.39	42.50	37.83	
	3-Sep	Plant SBFS					
2	20-Mar	Fertilizer, 70 kg/ha N with Urea	0.46	152.17	70.00	62.30	
	25-May	Fertilizer, 70 kg/ha N with Urea	0.46	152.17	70.00	62.30	
	12-Jun	apply 3" or 76.2 mm					76.20
	18-Jun	76.2 mm					76.20
	24-Jun	76.2 mm					76.20
	30-Jun	76.2 mm					76.20
	6-Jul	76.2 mm					76.20
	12-Jul	76.2 mm					76.20
	18-Jul	76.2 mm					76.20
	24-Jul	76.2 mm					76.20
	7-Aug	Harvest and kill					
3	1-Mar	Tillage, Offset Disk (light-duty)					
	15-Mar	Tillage, Offset Disk (light-duty)					
	30-Mar	Tillage, Offset Disk (light-duty)					
	15-Apr	Plant Broccoli					
	15-Apr	Fertilize 222.16 kg of P2O5, using 11-52-0	0.52	427.23	222.16*	197.72*	
		N applied in 11-52-0-->	0.11		47.00	41.83	
	15-Apr	Fertilize 23.05 kg elemental N	1	23.05	23.05	20.52	

Table F8. (Continued)

	17-Apr	.30" or 7.68 mm					7.68
	20-Apr	.30" or 7.68 mm					7.68
	23-Apr	.30" or 7.68 mm					7.68
	26-Apr	.30" or 7.68 mm					7.68
	29-Apr	.30" or 7.68 mm					7.68
	3-May	.48" or 12.09 mm					12.09
	6-May	.48" or 12.09 mm					12.09
	9-May	.48" or 12.09 mm					12.09
	12-May	.48" or 12.09 mm					12.09
	15-May	.48" or 12.09 mm					12.09
	18-May	.48" or 12.09 mm					12.09
	21-May	.48" or 12.09 mm					12.09
	24-May	.48" or 12.09 mm					12.09
	27-May	.48" or 12.09 mm					12.09
	30-May	.48" or 12.09 mm					12.09
	3-Jun	.81" or 20.68 mm					20.68
	5-Jun	Fertilize, Elemental N, 247.58 kg/ha	1	247.58	247.58	220.35	
	6-Jun	.81" or 20.68 mm					20.68
	9-Jun	.81" or 20.68 mm					20.68
	12-Jun	.81" or 20.68 mm					20.68
	15-Jun	.81" or 20.68 mm					20.68
	18-Jun	.81" or 20.68 mm					20.68
	21-Jun	.81" or 20.68 mm					20.68
	24-Jun	.81" or 20.68 mm					20.68
	27-Jun	.81" or 20.68 mm					20.68
	30-Jun	.81" or 20.68 mm					20.68
	10-Jul	Harvest & Kill					
	1-Aug	Tillage, Offset Disk (light-duty)					
	15-Oct	Plant Winter Wheat					
	15-Oct	Fertilize, 46-0-0, 68.93 kg/ha	0.46	68.93	31.71	28.22	
	15-Oct	Fertilize, 0-15-0, total of 50.73 kg P2O5/ha	0.15	338.21	50.73*	45.15*	
	16-Oct	Tillage, Harrow (Tine)					
4	10-Mar	Fertilizer, Elemental N, 203.4 kg/ha	1	203.4	203.40	181.03	
	7-Aug	Harvest & Kill					
5	9-May	Tillage, Chisel Plow					
	15-May	Tillage, Single Disk					
	20-May	Tillage, Harrow (tines)					
	20-May	Tillage, Seedbed Roller					
	21-May	Plant Corn					
	21-May	Fertilize, 327.05 kg/ha of 11-52-00	0.52	327.05	170.07*	151.36*	
		N applied in 11-52-0-->	0.11		35.98		
	21-May	Fertilize, Urea	0.46	58.52	26.92	23.96	

Table F8. (Continued)

22-May	34.47 mm					34.47
1-Jun	28.59 mm					28.59
10-Jun	28.59 mm					28.59
19-Jun	28.59 mm					28.59
28-Jun	28.59 mm					28.59
5-Jul	69.61 mm					69.61
19-Jul	69.61 mm					69.61
22-Jul	Fertilize, Urea, 413.6 kg/ha	0.46	413.58	190.25	169.32	
23-Jul	69.61 mm					69.61
1-Aug	43.73 mm					43.73
10-Aug	43.73 mm					43.73
19-Aug	43.73 mm					43.73
28-Aug	43.73 mm					43.73
6-Sep	40.79 mm					40.79
15-Sep	Harvest & Kill					
16-Sep	Tillage, Offset Disk (light-duty)					
15-Oct	Plant Winter Wheat					
31-Dec	Kill Wheat					

*Values are for P2O5, not N

Table F9. Row crop GW-BMP management scenario. Crop irrigation and fertilization rates were lowered to Extension rates and cover crops were planted between years 2-3, 4-5, and 5-1. In the non-GW-BMP row crop scenario, cover crops were only used between years 5 and 1. Fertilization and irrigation rates were not changed for sugarbeet for seed (SBFS) between GW-BMP and non-GW-BMP scenarios as sugarbeet growers are closely regulated by local processors. Values have been adapted from Steiner and Karow (1997), Lisec et al. (1995), Smesrud et al. (1997), Cuenca et al. (1992), Cross et al. (1988), Draycott and Christenson (2003), Cooke and Scott (1993), and McGrath (personal communication 2005).

Row Rotation BMP: CUCM (3.43%) + AGRR (0.03%)

				Extension (BMP) Fertilizer & Irrigation Rates			
Year	Date	Operation	Fraction N	kg/ha	kg N/ha	lbsN/a	Irrigation (mm)
1	3-Apr	Tillage, Offset Disk (light-duty)					
	4-Apr	Plant Winter Wheat					
	4-Apr	Fertilizer, 15-15-15, 149.8 kg/ha	0.15	149.80	22.47	20.00	
	4-Apr	Tillage, Harrow (Tines)					
	5-Apr	Tillage, Cultipacker					
	6-Apr	Tillage, Harrow (Tines)					
	27-Apr	Fertilize, 46-0-0, 194 kg/ha	0.46	194.00	89.24	79.42	
	7-Aug	Harvest & Kill					
	15-Aug	Tillage: Moldboard plow					
	22-Aug	Tillage: Offset Disc					
	27-Aug	Tillage:Harrow (tines)					
	27-Aug	Tillage: seedbed roller					
	30-Aug	Fertilizer: 130 kg/ha P2O5	0.52	250	130.00*	115.70*	
		N Applied in 11-52-0-->	0.11		27.50	24.48	
	30-Aug	Fertilizer, 46-0-0, 92.39 kg/ha	0.46	92.39	42.50	37.83	
	3-Sep	Plant SBFS					
2	20-Mar	Fertilizer, 70 kg/ha N with Urea	0.46	152.17	70.00	62.30	
	25-May	Fertilizer, 70 kg/ha N with Urea	0.46	152.17	70.00	62.30	
	12-Jun	apply 3" or 76.2 mm					76.20
	18-Jun	76.2 mm					76.20
	24-Jun	76.2 mm					76.20
	30-Jun	76.2 mm					76.20
	6-Jul	76.2 mm					76.20
	12-Jul	76.2 mm					76.20
	18-Jul	76.2 mm					76.20
	24-Jul	76.2 mm					76.20
	7-Aug	Harvest and kill					
	21-Aug	Tillage, Single Disk					
	1-Sep	Tillage, Moldboard Plow					
	18-Sep	Tillage, Harrow (tines)					
	18-Sep	Tillage, Seedbed roller					
	19-Sep	Tillage, Harrow (tines)					
	19-Sep	Tillage, Seedbed roller					
	21-Sep	Plant Italian (Annual) Ryegrass					

Table F9. (Continued)

3	1-Mar	kill rye					
	1-Mar	Tillage, Offset Disk (light-duty)					
	15-Mar	Tillage, Offset Disk (light-duty)					
	30-Mar	Tillage, Offset Disk (light-duty)					
	15-Apr	Plant Broccoli					
	15-Apr	Fertilize 196.6 kg of P2O5, using 11-52-0	0.52	378.08	196.60*	174.98*	
		N applied in 11-52-0-->	0.11		41.59	37.01	
	15-Apr	Fertilize 20.4 kg elemental N	1	20.40	20.40	18.16	
	17-Apr	.27" or 6.8 mm					6.80
	20-Apr	.27" or 6.8 mm					6.80
	23-Apr	.27" or 6.8 mm					6.80
	26-Apr	.27" or 6.8 mm					6.80
	29-Apr	.27" or 6.8 mm					6.80
	3-May	.42" or 10.7 mm					10.70
	6-May	.42" or 10.7 mm					10.70
	9-May	.42" or 10.7 mm					10.70
	12-May	.42" or 10.7 mm					10.70
	15-May	.42" or 10.7 mm					10.70
	18-May	.42" or 10.7 mm					10.70
	21-May	.42" or 10.7 mm					10.70
	24-May	.42" or 10.7 mm					10.70
	27-May	.42" or 10.7 mm					10.70
	30-May	.42" or 10.7 mm					10.70
	3-Jun	.72" or 18.3 mm					18.30
	5-Jun	Fertilize, Elemental N, 219.1 kg/ha	1	219.10	219.10	195.00	
	6-Jun	.72" or 18.3 mm					18.30
	9-Jun	.72" or 18.3 mm					18.30
	12-Jun	.72" or 18.3 mm					18.30
	15-Jun	.72" or 18.3 mm					18.30
	18-Jun	.72" or 18.3 mm					18.30
	21-Jun	.72" or 18.3 mm					18.30
	24-Jun	.72" or 18.3 mm					18.30
	27-Jun	.72" or 18.3 mm					18.30
	30-Jun	.72" or 18.3 mm					18.30
	10-Jul	Harvest & Kill, CNOP					
	1-Aug	Tillage, Offset Disk (light-duty)					
	15-Oct	Plant Winter Wheat					
	15-Oct	Fertilize, 46-0-0, 61 kg/ha	0.46	49.00	22.54	20.06	
	15-Oct	Fertilize, 0-15-0, total of 44.9 kg P2O5/ha	0.15	299.30	44.90*	39.96*	
	16-Oct	Tillage, Harrow (Tine)					
4	10-Mar	Fertilizer, Elemental N, 180 kg/ha	1	123.00	123.00	109.47	
	7-Aug	Harvest & Kill					
	1-Sep	Tillage, Single Disk					
	8-Sep	Tillage, Moldboard Plow					
	18-Sep	Tillage, Harrow (tines)					
	18-Sep	Tillage, Seedbed roller					

3	1-Mar	kill rye					
	1-Mar	Tillage, Offset Disk (light-duty)					
	15-Mar	Tillage, Offset Disk (light-duty)					
	30-Mar	Tillage, Offset Disk (light-duty)					
	15-Apr	Plant Broccoli					
	15-Apr	Fertilize 196.6 kg of P2O5, using 11-52-0	0.52	378.08	196.60*	174.98*	
		N applied in 11-52-0-->	0.11		41.59	37.01	
	15-Apr	Fertilize 20.4 kg elemental N	1	20.40	20.40	18.16	
	17-Apr	.27" or 6.8 mm					6.80
	20-Apr	.27" or 6.8 mm					6.80
	23-Apr	.27" or 6.8 mm					6.80
	26-Apr	.27" or 6.8 mm					6.80
	29-Apr	.27" or 6.8 mm					6.80
	3-May	.42" or 10.7 mm					10.70
	6-May	.42" or 10.7 mm					10.70
	9-May	.42" or 10.7 mm					10.70
	12-May	.42" or 10.7 mm					10.70
	15-May	.42" or 10.7 mm					10.70
	18-May	.42" or 10.7 mm					10.70
	21-May	.42" or 10.7 mm					10.70
	24-May	.42" or 10.7 mm					10.70
	27-May	.42" or 10.7 mm					10.70
	30-May	.42" or 10.7 mm					10.70
	3-Jun	.72" or 18.3 mm					18.30
	5-Jun	Fertilize, Elemental N, 219.1 kg/ha	1	219.10	219.10	195.00	
	6-Jun	.72" or 18.3 mm					18.30
	9-Jun	.72" or 18.3 mm					18.30
	12-Jun	.72" or 18.3 mm					18.30
	15-Jun	.72" or 18.3 mm					18.30
	18-Jun	.72" or 18.3 mm					18.30
	21-Jun	.72" or 18.3 mm					18.30
	24-Jun	.72" or 18.3 mm					18.30
	27-Jun	.72" or 18.3 mm					18.30
	30-Jun	.72" or 18.3 mm					18.30
	10-Jul	Harvest & Kill, CNOP					
	1-Aug	Tillage, Offset Disk (light-duty)					
	15-Oct	Plant Winter Wheat					
	15-Oct	Fertilize, 46-0-0, 61 kg/ha	0.46	49.00	22.54	20.06	
	15-Oct	Fertilize, 0-15-0, total of 44.9 kg P2O5/ha	0.15	299.30	44.90*	39.96*	
	16-Oct	Tillage, Harrow (Tine)					
4	10-Mar	Fertilizer, Elemental N, 180 kg/ha	1	123.00	123.00	109.47	
	7-Aug	Harvest & Kill					
	1-Sep	Tillage, Single Disk					
	8-Sep	Tillage, Moldboard Plow					
	18-Sep	Tillage, Harrow (tines)					
	18-Sep	Tillage, Seedbed roller					
	19-Sep	Tillage, Harrow (tines)					
	19-Sep	Tillage, Seedbed roller					
	21-Sep	Plant Italian (Annual) Ryegrass					

Table F9. (Continued)

	19-Sep	Tillage, Harrow (tines)					
	19-Sep	Tillage, Seedbed roller					
	21-Sep	Plant Italian (Annual) Ryegrass					
5	1-May	Kill rye					
	9-May	Tillage, Chisel Plow					
	15-May	Tillage, Single Disk					
	20-May	Tillage, Harrow (tines)					
	20-May	Tillage, Seedbed Roller					
	21-May	Plant Corn					
	21-May	Fertilize,11-52-00, 289.42 kg/ha	0.52	289.42	150.50*	133.95*	
		N applied in 11-52-0-->	0.11		31.84		
	21-May	Fertilize, Urea	0.46	51.79	23.82	21.20	
	22-May	30.5 mm					30.50
	1-Jun	25.3 mm					25.30
	10-Jun	25.3 mm					25.30
	19-Jun	25.3 mm					25.30
	28-Jun	25.3 mm					25.30
	5-Jul	61.6mm					61.60
	19-Jul	61.6mm					61.60
	22-Jul	Fertilize, Urea, 366 kg/ha	0.46	366.00	168.36	149.84	
	23-Jul	61.6mm					61.60
	1-Aug	38.7 mm					38.70
	10-Aug	38.7 mm					38.70
	19-Aug	38.7 mm					38.70
	28-Aug	38.7 mm					38.70
	6-Sep	36.1 mm					36.10
	15-Sep	Harvest & Kill					
	16-Sep	Tillage, Offset Disk (light-duty)					
	15-Oct	Plant Winter Wheat					
	31-Dec	Kill Wheat					

*Values are for P2O5, not N

Table F10. Urban lawn management scenario. Fertilizer and irrigation rates were weighted based on values reported in Nielsen and Smith (2005). GW-BMPs were not applied for urban areas. Values adapted from VanDerZanden and Cook (2001), Cook and McDonald (2005), and Nielsen and Smith (2005).

Urban Fertilizer: URMD (4.83%) + URMH (0.02%) + URHD (0.01%) + UINS (1.38%) + UCOM (0.22%)+ UIDU (0.13%) = 6.59%

Year	Date	Operation	Fraction N	kg/ha applied	kg N/ha	lbsN/a	Irrigate (in)	Irrigate (mm)
1	1-Feb	Plant/begin growing season						
	1-May	Fertilize, 127.7 kg/ha Urea	0.46	127.67	58.73	52.27		
	1-May	Irrigate .53"					0.53	13.46
	8-May	Irrigate .53"					0.53	13.46
	15-May	Irrigate .53"					0.53	13.46
	22-May	Irrigate .53"					0.53	13.46
	1-Jun	Irrigate .9"					0.90	22.86
	8-Jun	Irrigate .9"					0.90	22.86
	15-Jun	Irrigate .9"					0.90	22.86
	22-Jun	Irrigate .9"					0.90	22.86
	1-Jul	Fertilize, 127.7 kg/ha Urea	0.46	127.67	58.73	52.27		
	1-Jul	Irrigate .95"					0.95	24.13
	8-Jul	Irrigate .95"					0.95	24.13
	15-Jul	Irrigate .95"					0.95	24.13
	22-Jul	Irrigate .95"					0.95	24.13
	1-Aug	Irrigate .95"					0.95	24.13
	8-Aug	Irrigate .95"					0.95	24.13
	15-Aug	Irrigate .95"					0.95	24.13
	22-Aug	Irrigate .95"					0.95	24.13
	1-Sep	Irrigate .63"					0.63	16.00
	8-Sep	Irrigate .63"					0.63	16.00
	15-Sep	Fertilize, 127.7 kg/ha Urea	0.46	127.67	58.73	52.27		
	15-Sep	Irrigate .63"					0.63	16.00
	22-Sep	Irrigate .63"					0.63	16.00
	1-Oct	Irrigate .26"					0.26	6.60
	8-Oct	Irrigate .26"					0.26	6.60
	15-Oct	Irrigate .26"					0.26	6.60
	22-Oct	Irrigate .26"					0.26	6.60

Appendix G

Algorithm Used to Fill Large Hydrograph Data Gaps

A modeling approach using hydrograph ratios was used to fill large data gaps in input and calibration/validation hydrographs for SWAT. The following approach was found to have Nash-Sutcliffe values ranging from 0.79 – 0.94. The approach used is as follows:

1) Determine what nearby rivers have data for the missing time period on River A (the river of interest that has missing hydrograph data). The river should be similar to River A in flow magnitude, proximity (so both rivers are likely to experience the same storms), and geographic/geologic province (so that drastically different flow regimes, such as a baseflow dominated river and a flashy surface flow dominated river, are not compared).

2) Calculate daily flow ratios between River A with other rivers identified in step 1 for time periods where flow data exists for both rivers. For the entire set of daily ratios, determine which of the rivers has the lowest coefficient of variation relative to River A. This is the river that the ratio model will be applied to (henceforth referred to as River B).

3) For the periods where both River A and River B have hydrograph data, create a table to determine the average daily ratio between rivers. Ideally, 10 or more years of data should be used to determine average daily ratios.

4) For periods where river data does not exist for River A, model the expected flow by multiplying the daily flows in River B by the corresponding average daily flows calculated in step 3.

Appendix H

Southern Willamette Valley Stream Data
(Nitrate Concentrations and Flow Data for Muddy Creek)

Stream data were collected in this study for the SWV because it was initially thought that they would be necessary for calibrating and validating the SWAT model. Data collected include nitrate data for rivers near hydrograph locations, as well as continuous flow data for Muddy Creek at Stahlbush Island Road near Corvallis. Nitrate data are presented in Table H1 and Figure H1, while the rating curve created for Muddy Creek, stage heights, and flow values are presented in Figures H2-H4.

Table H1. Chart of NO_3-N (mg/L) values observed in rivers of the Southern Willamette Valley.

ID	Description	11/04	12/04	1/05	2/05	3/05	4/05	5/05	6/05	7/05	8/05	9/05	10/05	Median
22	Marys River Corvallis	0.37	1.3	1.01	0.66	0.20	0.54	0.47	0.29	0.14	0.08	0.09	0.02	0.33
23	Willamette in Corvallis, Super 8	0.18	0.5	0.48	0.35	0.26	0.32	0.22	0.09	0.14	0.06	0.13	0.12	0.20
24	Willamette @ Irish Bend		0.3	0.31	0.23	0.23	0.17	0.18	0.04	0.11	0.03	0.08	0.08	0.17
25	Long Tom @ Fern Ridge	0.09	0.3	0.37	0.22	0.05	0.21	0.11	<0.01	0.05	0.04	0.04	0.03	0.09
26	Long Tom @ Monroe	0.15	1.0	0.70	0.52	0.33	0.49	0.3	0.15	0.14	0.04	0.02	0.15	0.23
27	Willamette N of Eugene	0.08	0.5	0.24	0.14	0.09	0.16	0.21	0.10	0.08	0.07	0.13	0.12	0.13
28	Muddy Creek, E of Corvallis		2.6	3.03	2.49	1.84	2.75	1.48	1.31	0.95	0.39	0.43	0.85	1.48

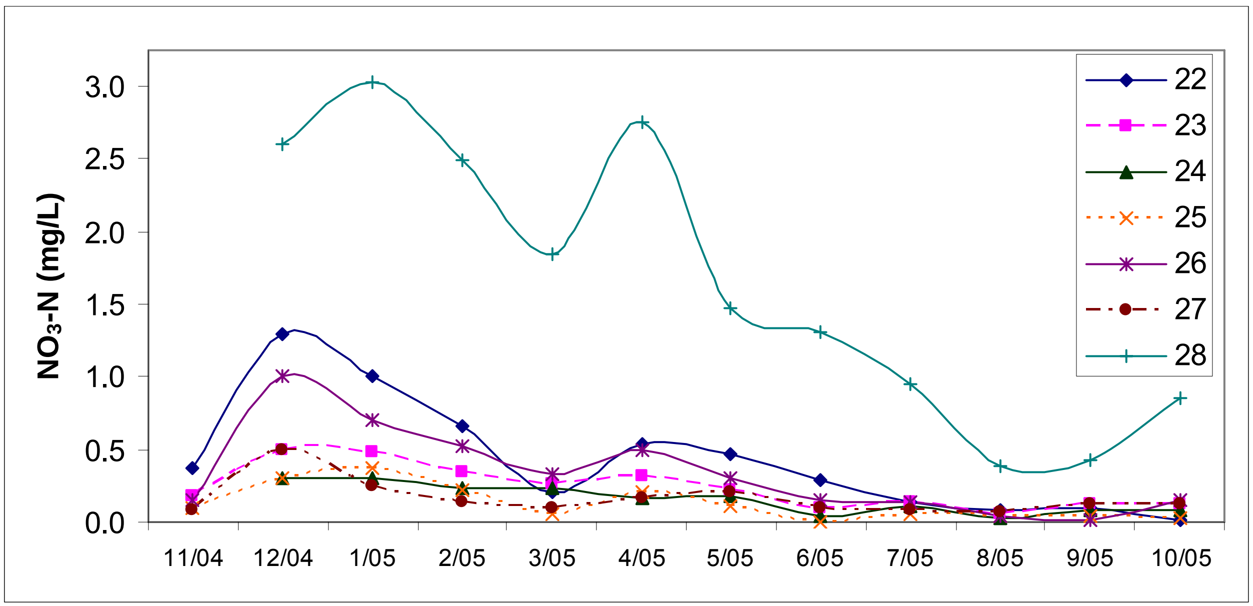


Figure H1. Graph of riverine nitrate concentrations for the SWV. All data presented are found in Table H1.

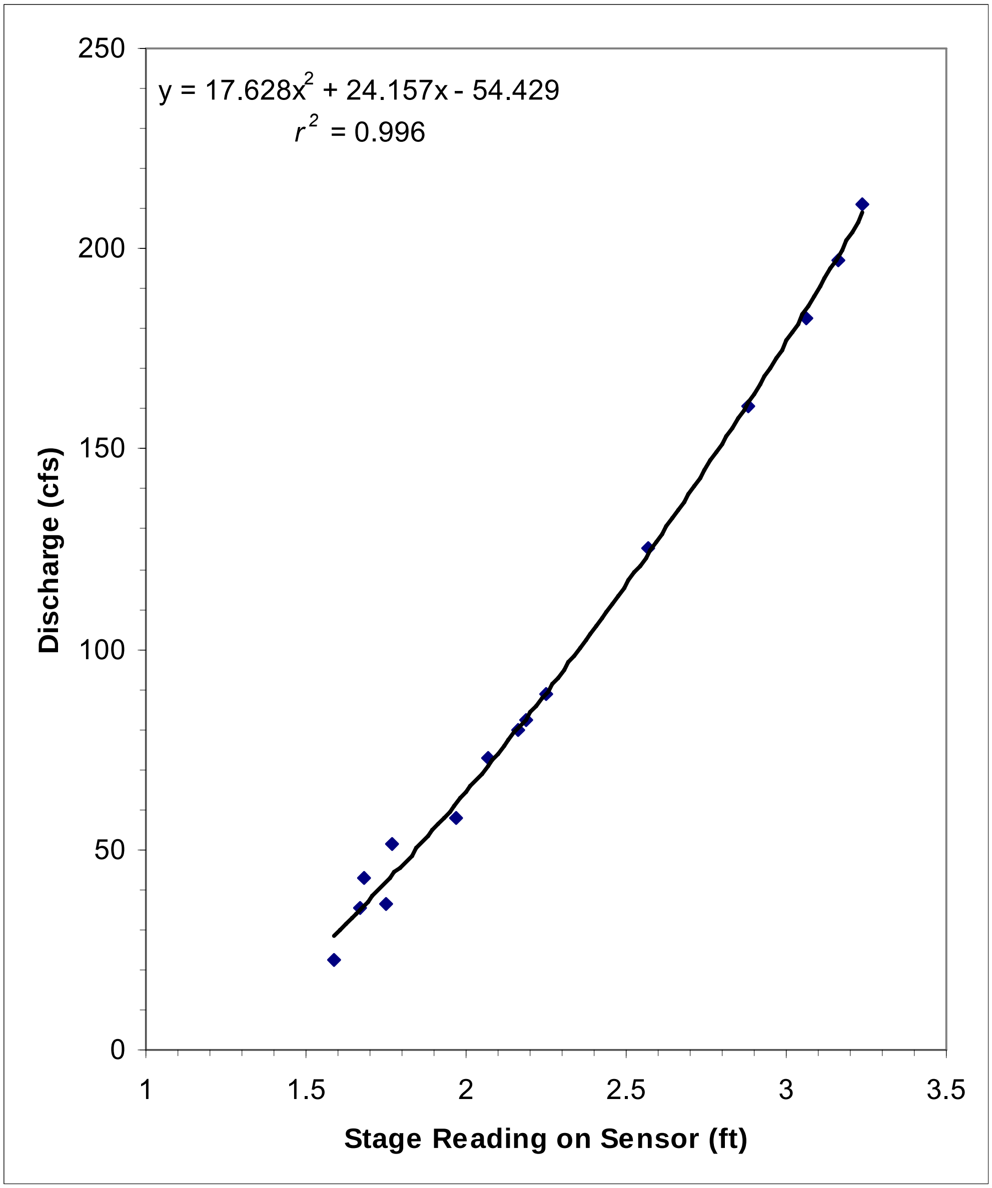


Figure H2. Rating curve for Muddy Creek in at Stahlbush Island Road near Corvallis. Data point collection occurred from January through July 2005

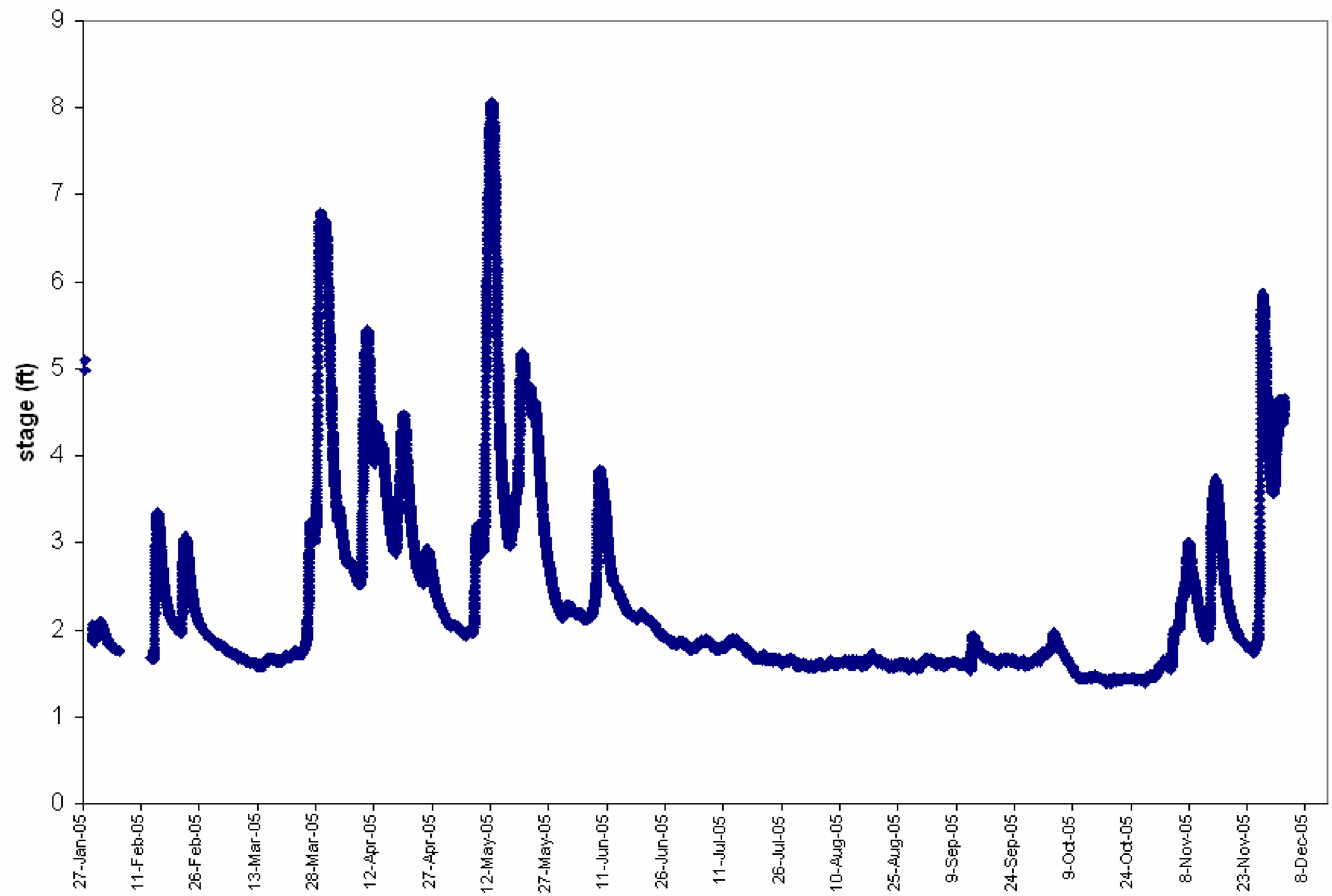


Figure H3. Stage data from the pressure transducer installed in Muddy Creek.

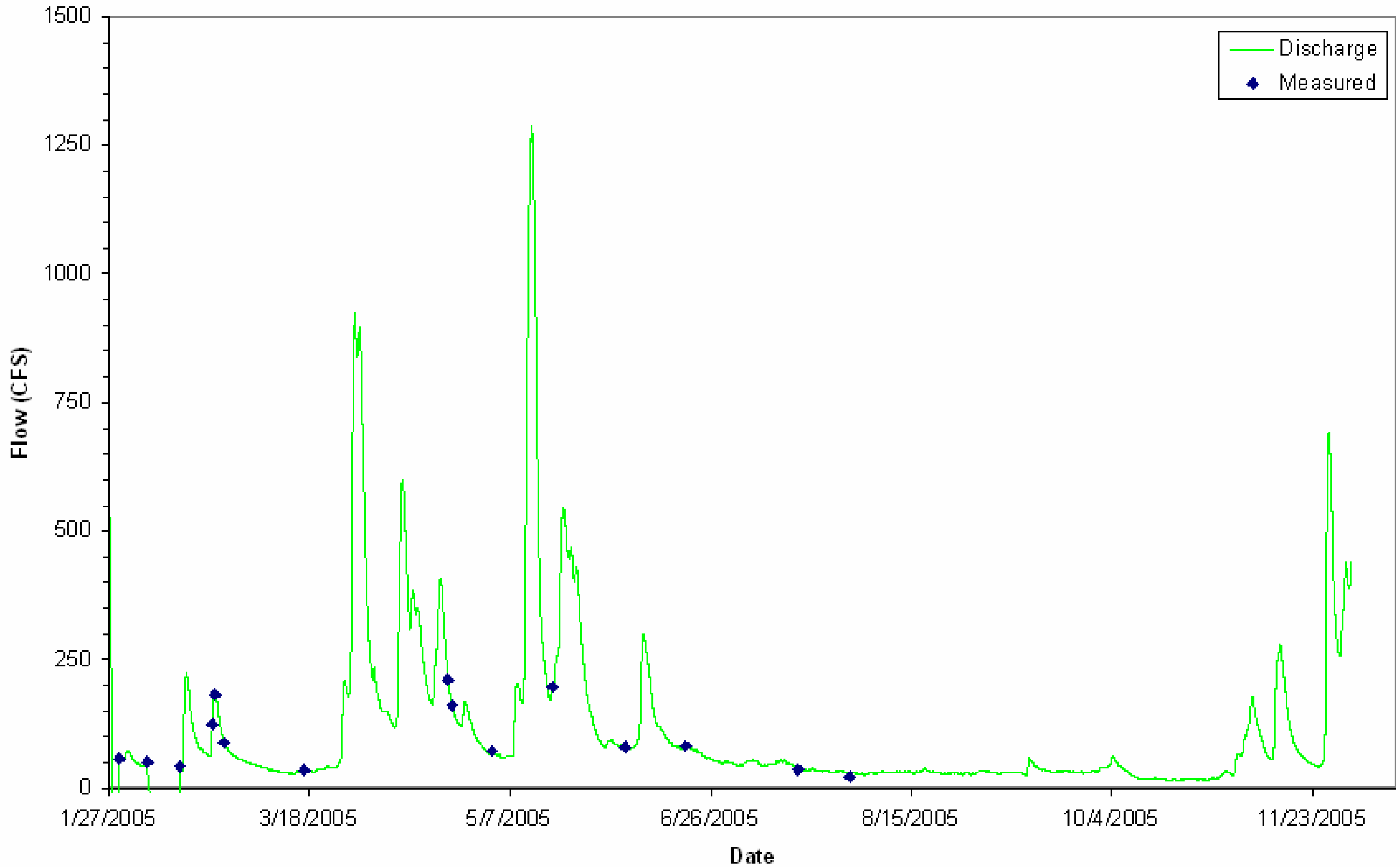


Figure H4. Flow data for Muddy Creek, with the rating curve from Figure H2 being applied to the stage data in Figure H3. Flow above ~ 250 cfs has low confidence associated with it because higher flows are not defined on the rating curve (the creek was not crossable when stage readings were > 3.5 ft). Dots indicate dates when flow data were collected.