

Water Quality: Abbotsford-Sumas Final Report

Dr. Robert J. Mitchell, Dr. R. Scott Babcock,
Ms. Heather Hirsch, and Ms. Leslie McKee
Geology Department

and

Dr. Robin A. Matthews and Ms. Joan Vandersypen
Institute for Watershed Studies,
Western Washington University *

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Executive Summary

We monitored the groundwater and surface water in a small agricultural region (6 sq-mi) south of Judson Lake in Whatcom County from July 2002 to June 2004 to establish temporal and spatial distributions of nitrate. The study area lies within the bounds of the Abbotsford-Sumas aquifer that spans regions in British Columbia and Whatcom County Washington. Groundwater flows south from British Columbia into Whatcom County in the aquifer. The study area was chosen because it has a history of persistent groundwater-nitrate contamination and two surface-water creeks that are on the approved Total Maximum Daily Limit list. Our primary objectives were to compare nitrate and other related water quality species to local agronomic loading information provided by regional agricultural agencies, and to estimate concentrations of nitrate transported south across the international border using data from Environment Canada's monitoring wells north of the study area in British Columbia.

Data were collected and analyzed according to the protocol outlined in an approved Quality Assurance Project Plan. A total of 466 groundwater samples from 26 wells were processed over the 24-month period. An additional 217 samples were analyzed for quality control. Most of the groundwater samples (87%) had nitrate levels¹ greater than 3 mg N/L, and 21 of the study wells had median nitrate values greater than 3 mg N/L. These levels indicate probable anthropogenic contamination. In addition, 64% of the nitrate samples and 14 of the well medians were greater than the EPA maximum contaminant level (MCL) of 10 mg N/L.

Nitrate concentrations were higher in the northern half of the study area due to a combination of transport from British Columbia and leaching from agricultural lands in Whatcom County. The majority (12) of the 15 wells monitored in the northern half had median nitrate concentrations above the 10 mg N/L MCL. The low nitrate concentrations in the southern half of the study area were due to denitrification that occurs in reducing, iron-rich peat that exists in a bog and along portions of a creek in the central portion of the study area. Water quality parameters indicate that it is also likely that denitrification in unmapped peat deposits was responsible for anomalously low nitrate concentrations measured from wells dispersed throughout the study area. When wells with low nitrate values in the study area (likely due to denitrification) were eliminated, the greatest contamination occurred in shallow wells (<25 ft), which produced a median value of 15.7

¹Nitrate was measured as nitrate+nitrite for this study. The predominant form was nitrate.

mg N/L. The deep well (>25 ft) nitrate concentrations were lower in magnitude, and resulted in a median value of 11.3 mg N/L. This stratification in nitrate concentrations is in agreement with historical observations.

The agricultural land use in the study area is primarily raspberries and crops (grass and corn) receiving dairy manure. Raspberries are fertilized in February and March and dairy manure can be spread on corn/grasses anytime between February and October. Irrigation water can be applied between May and August on raspberries and between April and September on grasses. Land use to the north in British Columbia is primarily raspberries, which have been receiving a mix of poultry manure and inorganic fertilizer in recent years.

We used a combination of land-use maps, nutrient-loading information, and nitrate-nitrogen isotopes (measured quarterly) to determine nitrogen sources of the groundwater nitrate. The ratio of stable nitrogen isotopes from nitrate was used to differentiate inorganic-commercial fertilizers from animal manure. Most of the 123 groundwater samples analyzed from the 25 wells over the 24-month period indicated organic manure, or a mix of organic and inorganic nitrogen as the source of the nitrate. These results are consistent with the fertilization practices that occur in the study area, and north of the study area in British Columbia. Nitrogen isotope results from two wells were consistently within the inorganic, commercial fertilizer range, whereas median results from 10 wells were within the animal manure range. It is possible that enrichment due to denitrification was responsible for elevated values in some of these wells. Samples from the other 13 wells had nitrogen isotope values that indicated a mix of organic and inorganic nitrogen.

The nitrate-time series from nearly all the wells displayed some kind of pattern, many showing highs and lows of varying degrees, and at inconsistent times. This variability was due to the multiple factors that may be influencing nitrate concentrations in each well, including: southerly flowing groundwater containing nitrate that originated in British Columbia; vadose zone thickness and residence time; depth below the water table; up gradient land use, including timings and types of nutrient loading; seasonal precipitation and irrigation; vertical and lateral mixing due to intermingling plumes along flow paths in the aquifer; mixing due to excessive well pumping; nitrate recirculation due to irrigation; and denitrification, especially due to peat deposits in the area.

Two Class A creeks were sampled every other month for two years (13 times). The water quality in the creeks was influenced by a combination of groundwater discharge and surface, or shallow subsurface, runoff associated with precipitation and irrigation events. A total of 109 surface water samples from 10 stream sites were analyzed over the 24-month period: 49 samples from 5 sites along Johnson Creek, and 61 samples from 5 sites along Pangborn Creek. An additional 61 samples were analyzed for quality control. Two sites in Johnson Creek (Sites 4 and 5) were not sampled during certain summer months because of low-flow conditions. The nitrate levels measured in both Pangborn and Johnson Creeks were above 5 mg N/L throughout the duration of the study (except for one site near Pangborn Bog), indicating anthropogenic sources. The nitrate values in general were usually lower in Johnson Creek, which had a median nitrate concentration of 6.5 mg N/L. The median nitrate concentration for Pangborn Creek was 9.0 mg N/L. Both creeks had higher summer nitrate and peaks in other water quality parameters that correlated to heavy precipitation events.

Fecal coliform concentrations ranged from 5 to 1200 cfu/100 mL in Johnson Creek and from 2 to 12,000 cfu/100 mL in Pangborn Creek. About 50% of the samples measured in both creeks exceeded the target level of 100 cfu/100 mL for fecal coliforms in Class A streams. The highest fecal counts occurred in the summer months in both creeks. Approximately 27% of the samples measured in Johnson Creek and 30% of the samples from Pangborn exceeded the 8.0 mg/L target level for dissolved oxygen in Class A streams. The lowest dissolved oxygen concentrations were usually measured during the summer.

The nitrogen isotope values from both creeks (33 samples) indicated animal manure as the source of the nitrate, which is consistent with the land use in the vicinity of the creeks. It is also possible that the isotopes were enriched due to denitrification processes in the peat.

Based on our results, the nitrate concentrations in both ground water and surface water in the study area do not appear to be decreasing. Because many spatial and temporal factors may influence nitrate concentrations, further study would be needed to assess the effects of specific nutrient-loading plans in the study area. In particular, there needs to be more extensive soil and soil pore-water analyses, shallow ground-water monitoring, and numerical modeling of surface and subsurface nitrate movement.

1 Introduction

1.1 Background

The Abbotsford-Sumas aquifer is located in southwest British Columbia, Canada and extends across the international boundary into Whatcom County in Washington State, USA (Figure 1, page 42). The predominately unconfined aquifer lies within the Fraser and Nooksack Lowlands and is comprised primarily of unconsolidated glacial sands and gravels. The aquifer occupies approximately 77 sq-mi (200 sq-km) and serves as a water supply for more than 100,000 people in the United States and British Columbia (Hii et al., 1999). The fertile, well-drained soils in the Fraser and Nooksack lowlands make it one of the more intensely farmed regions in British Columbia and Whatcom County. Unfortunately, when combined with high precipitation, these agricultural activities make the unconfined aquifer susceptible to nitrate contamination (Tesoriero and Voss, 1997).

Nitrate is a common pollutant of shallow groundwater that has been linked to agricultural practices in the Abbotsford-Sumas aquifer, specifically dairy and poultry manure and inorganic commercial fertilizer application. Nitrate concentrations above the Canadian Drinking Water Guidelines of 10 milligrams per liter as nitrogen (10 mg N/L) have been consistently documented over the past decade in British Columbia (Leibschner et al., 1992; Wassenaar, 1995; Hii et al., 1999). Groundwater flows south from British Columbia into Whatcom County. Nitrate concentrations above the United States Environmental Protection Agency (EPA) maximum contaminant level (MCL) of 10 mg N/L have also been reported in Whatcom County (Ryan, 1994; Garland and Erickson, 1994; Erickson, 1998; Cox and Kahle, 1999; Mitchell et al., 2000; Tesoriero et al., 2000; Mitchell et al., 2003). In addition, degraded surface water quality in the region also suggests contamination due to agricultural land use (Wills, 1998; Butkus et al., 2000; Mitchell et al. 2003).

From April 1997 through February 1999, researchers at Western Washington University (WWU) quantified groundwater and surface water nitrate magnitudes and identified flow paths in a study area south of Judson Lake near the International border in Whatcom County (Figure 1, page 42). They analyzed hydrogeologic data, nitrogen isotope signatures and water quality data, and concluded that a two-layered stratification of nitrate contamination existed in the study area (Mitchell et al., 2000; 2003). The shallow layer (< 25 feet) contained nitrate values greater

than 20 mg N/L, which was considered to be a result of local agricultural practices. The deeper layer (> 25 feet) had nitrate values greater than 10 mg N/L, which was considered to be a result of British Columbia agricultural practices. Additionally, samples from a number of wells had low nitrate concentrations presumed to be associated with denitrification. This assumption was supported by the presence of anoxic conditions and elevated iron and manganese concentrations in these wells.

In a step to improve water quality, the State of Washington signed into law the Dairy Nutrient Management Act (RCW 90.64) in April 1998. As part of this Act, all dairy farmers had to register with the Whatcom Conservation District (WCD) and have an approved Dairy Nutrient Management Plan implemented by December of 2003. Unfortunately, the task of managing nutrient loading in the county is hindered by southerly flowing groundwater that transports nitrate from agricultural sources in British Columbia. Therefore, accurate knowledge of nitrate concentration distributions and source separation in the aquifer is important to manage nutrient loading in the agricultural areas of the county.

The primary objective of this study was to monitor the groundwater and surface water in a small region south of Judson Lake in Whatcom County to establish temporal and spatial distributions of nitrate and related species and correlate these to agronomic loading information. The results of the monitoring will serve as a means to assess the effectiveness of the newly established Dairy Nutrient Management Plans resulting from the 1998 Dairy Nutrient Management Act.

The study area near Judson Lake was chosen because it has a history of persistent nitrate contamination and a well characterized hydrostratigraphy and groundwater flow regime. Dairy Nutrient Management Plans have been developed for farms in the area and there are numerous domestic wells accessible for monitoring. In addition, nitrate data are available from monitoring wells north of the study area in British Columbia. There are also two surface-water creeks in the study area that are on the approved Total Maximum Daily Limit (TMDL) list (listed parameters are dissolved oxygen and fecal coliform).

The objectives of the project were to:

1. Monitor water quality parameters from 25 domestic drinking water wells in the study area to establish temporal and spatial trends for nitrate concentrations. Monitoring took place monthly (July 2002 to June 2004) in 10–12 of the 25 wells, and every-other month in all 25 wells, for 2 years. In general, we chose shallow and deep wells along Halverstick Road, Pangborn Road, and Van Buren Road.

2. Quantify nitrate-nitrogen isotope magnitudes 3 times per year in 25 wells to assess nitrogen sources in the study area.
3. Collect available agronomic loading information from the USDA-Natural Resources Conservation Service and the Whatcom Conservation District and correlate the data to temporal and spatial groundwater nitrate concentrations.
4. Monitor select surface water quality parameters, including fecal coliforms, every other month for 2 years in the study area to assess surface water and groundwater interactions. Monitoring included 5 sites along a reach in Pangborn Creek and 5 sites along a reach in Johnson Creek.
5. Educate the community by offering a workshop for local teachers, developing a web site, and offering presentations at research and Whatcom County forums.

2 Study Area Description

2.1 Location

The study area for this project consisted of approximately 6 square miles (15.5 sq km) in Northern Whatcom County, west of the city of Sumas and bordering British Columbia Canada in the vicinity of Judson Lake (Figure 2, page 43). The township Sections and Ranges include: T41N, Sec. 36, R3E; T41N, Sec. 31 & 32, R4E; T40N, Sec. 1 & 12, R3E and T40N, Sec. 6, 7 & 8, R4E.

2.2 Hydrogeology

Regionally, the Abbotsford-Sumas Aquifer is predominately unconfined and covers an area of about 77 sq-mi (Liebscher et al., 1992). The aquifer is comprised largely of glacial outwash sand and gravel deposits that gently slope southward from Abbotsford, British Columbia to the Nooksack floodplain near Lynden, Washington. Kovanen and Easterbrook (2002) documented four phases (SI-SIV) of the Sumas Stade in the study area. Phase SIII (ca. 10,980–10,250 ¹⁴C years

B.P.) deposited the Sumas Outwash, the gravelly water-bearing portion of the aquifer. It includes recessional and advance glaciofluvial and glaciolacustrine deposits (Kovanen, 2002). Phase SIV is marked by glacial readvance into the study area (ca. <10,250–10,000 ^{14}C years B.P.) and more deposits of sand and gravel. Concurrently, blocks of ice melted forming kettles in what are now Pangborn and Judson Lakes. In time, Pangborn Lake evolved into a peat formation (ca. 10,245 90 and 10,265 65 ^{14}C years B.P.; Kovanen and Easterbrook, 2002). It is hypothesized that peat formed in various other unmapped locales and depths throughout the region. These peat deposits are considered to be key locations where denitrification is occurring in the aquifer (McKee, 2004).

According to Cox and Kahle (1999) the thickness of the unconfined aquifer is typically about 50 to 75 ft, but can range from 15 to 230 ft. They also used data from 170 completed wells in the aquifer to estimate a mean hydraulic conductivity of 270 feet per day and a median lateral groundwater velocity on the order of 2 feet per day. For our study area, Stasney (2000) used sediment-size analyses and empirical correlations to estimate the average hydraulic conductivity to be about 900 feet per day, which falls within the range of values determined by Cox and Kahle (1999). Stasney (2000) also estimated an average hydraulic gradient of 0.0075 and an average southerly horizontal flow velocity of 20 feet per day for the study area. Underlying the Abbotsford-Sumas Aquifer is a confining unit, primarily clay with an unsorted mixture of pebbly silt of glaciomarine origin (13,500–11,000 yr ago). The clay layer is believed to be continuous at depth and is interpreted as Bellingham glaciomarine drift (Easterbrook, 1976) in northwestern Washington and as the Fort Langley Formation to the north in British Columbia (Halstead, 1986).

Researchers in the Earth Science Department at Simon Fraser University have developed a three dimensional groundwater flow model for the Abbotsford-Sumas Aquifer (Scibek and Allen, 2005). The multi-layer Modflow model was developed using data from 2500 well logs on both sides of the border. The spatial distribution of hydraulic conductivities of the sediments in the aquifer was produced using an inverse interpolation method. They estimate the hydraulic conductivity of the gravels and sands in the majority of our study area to range from about 175 to 500 feet per day.

The outwash makes an excellent high yield aquifer that supplies water for both residential use and irrigation. Water levels in the study area typically fluctuate about 5 ft or less over the course of a year. They are generally lowest in October and highest in April (Stasney, 2000).

2.3 Climate, Soils, and Recharge

The temperate climate in the Abbotsford-Sumas area is influenced by Pacific-maritime air. The National Climatic Data Center's Clearbrook COOP gauging station (latitude 48° 58'N, longitude 122° 20'W and 64 ft above sea level) is just a couple of miles from the study area. The 30-year average annual temperature and precipitation values recorded at the station are 50.4°F (10.2°C) and 45.6 in (115.7 cm), respectively. About 70% of the precipitation typically occurs between October and March. Temperature and precipitation values from January 2002 through the end of the sampling period (June 2004) are plotted along with 30-year averages (Figures 3–4, pages 44–45). Most notable is the drier than normal spring in 2002 and an extremely wet October in 2003. Note that 18 daily values used to estimate the monthly averages were unavailable from the Clearbrook station, so they were replaced by values from the Bellingham International airport at latitude 48° 48'N, longitude 122° 32'W and 149 ft (45.4 m) above sea level.

The soil survey of Whatcom County Area, Washington (1992) summarizes the soil types within the study area. The soils are, in general, poorly developed and thin (<6 ft). Silt loam and gravelly loam soils are the most productive and widespread soils in the study area (Figure 5, page 46). The silt loams formed in a mixture of loess and volcanic ash over glacial outwash; therefore, the substratum of this unit is very gravelly to extremely gravelly. The permeabilities (0.6 to 2 in/hr) are generally greater than typical precipitation rates (Cox and Kahle, 1999). Peat, which has moderate to low permeability (0.06 to 0.6 in/hr), is abundant in the central portion of the study area.

Recharge to the aquifer is primarily by precipitation. Because of the permeable soils and coarse-grained aquifer material, recharge is on the order of 60% of the annual precipitation (Cox and Kahle, 1999). Recharge in many parts of the region is supplemented by irrigation in the summer months. Irrigation magnitudes are typically on the order of 1 to 1.5 ft during the summer (J. Gillies, personal communication, USDA Natural Resource Conservation Service, 2001).

2.4 Surface Water

The study area lies within the Johnson Creek watershed which includes small creeks, lakes, and engineered irrigation ditches that serve as discharge zones for the aquifer (Figure 5, page 46). Judson Lake lies in the northern portion of the study area and crosses the international boundary into Canada. A small lake of unknown depth occupies the center of Pangborn Bog and is only visible from aerial photographs. Two small springs feed Pangborn Bog from the north and west as well as several engineered drainage ditches to the north and east. Pangborn Creek discharges Pangborn Bog to the east, and is a principal tributary to Johnson Creek. The average stream discharge of Pangborn Creek is about 12 cubic feet per second (Wills, 1998). Johnson Creek is the main stream within the Johnson Creek Watershed and has an average annual stream discharge is about 50 cubic feet per second (Butkus et al., 2000). Discharge is usually highest in mid-winter and lowest in late summer resulting from variation in seasonal precipitation. The Johnson Creek watershed (with Class A streams) is on the approved TMDL 303(d) list. Listed parameters include dissolved oxygen and fecal coliforms (Butkus et al., 2000; Henderson, 2003).

2.5 Land Use

The land use in the study area is primarily agriculture, dominated by pasture, grassland, and corn associated with dairy operations, and raspberry and blueberry fields. Through the cooperation of the Whatcom Conservation District and The United States Department of Agriculture Natural Resources Conservation Service (USDA-NRCS), we have compiled land-use maps for the study area for the years 2002, 2003, and 2004 (Figures 6–8, pages 47–49). The land use changes that occurred during this time period include a transition from corn to grasslands just north of Pangborn Creek and a change from grassland to corn in a small region southeast of Judson Lake. The raspberry acreage remained constant.

We compiled monthly loading and irrigation activities for these crops, with assistance from the Whatcom Conservation District, the USDA-NRCS and the Washington State University Cooperative Extension (Tables 1–3, pages 37–39). The data in these tables are not fixed; they represent possible loading and irrigation requirements for optimum yields. For example, the lands associated with dairy operations can receive dairy manure fertilization between the months of February

and October in a given year—it doesn't mean that loading does occur each month. Berries are typically fertilized in February with dairy manure and in March with commercial, inorganic fertilizers. The irrigation magnitudes are possible requirements and may change if adequate rainfall is received during the summer.

Land use to the north in British Columbia is primarily raspberries, which have been receiving a mix of poultry manure and inorganic fertilizer in recent years (G. Hughes-Games, personal communication, BC Ministry of Agriculture, Food and Fisheries, 2004).

3 Groundwater Nitrogen Cycle

Nitrates can enter the groundwater system by natural or anthropogenic sources. Natural sources include soil nitrogen, nitrogen rich geologic deposits, and atmospheric deposition. Anthropogenic sources include inorganic commercial fertilizers, animal manure, septic tank drainage, and leaching of the soil due to irrigation (Madison and Brunett, 1985).

When nitrates enter the soil, a number of hydrologic, chemical or biological processes take place to convert nitrogen into nitrate, nitrite, ammonia, N_2 gas, or nitrous oxide. This process is called the nitrogen cycle and includes five stages (Canter, 1997; Kendall, 1998):

1. **Nitrogen fixation** occurs when di-nitrogen gas (N_2) is converted to ammonia (NH_3) by aerobic or anaerobic bacteria. Other fixation pathways include industrial, atmospheric (lightning) and combustion;
2. **Ammonification or mineralization** is a biochemical process where bacteria alter organic nitrogen into ammonium during the decomposition of plants, animals, and fecal matter;
3. **Assimilation or Synthesis** is a biochemical process, where plants take up either ammonium or nitrate (NO_3^-) to form proteins or other nitrogen containing compounds. This process does not involve permanent nitrogen loss because these compounds are re-released into the subsurface;
4. **Nitrification** is a biological process by which ammonium ions become oxidized. Nitrification occurs in two steps: first, ammonium and oxygen com-

bine to form nitrite (NO_2^+) and oxygen; and second, nitrite and oxygen combine to form nitrate. Nitrite is an intermediate stage that is not often found in most environments; and

5. **Denitrification**, which occurs when bacteria respire anaerobically and convert nitrate into nitrite, and possibly into nitrous oxide, nitrogen dioxide and finally nitrogen gas. Dissimilatory nitrate reduction occurs in anaerobic environments when nitrate is temporarily converted to ammonium.

A decrease in nitrate concentrations in groundwater may result from dilution, denitrification, or assimilatory reduction to ammonia or ammonium. The assimilation process of transforming nitrate or ammonium into organic nitrogen is reversible, the organic nitrogen can later be transformed back into nitrate. As a result, assimilation alone will not cause significant, long-term increases or decreases in nitrate concentrations in the soil. Denitrification is the only mechanism that results in a significant, long-term reduction of nitrate concentrations because the reactants are unlikely to be transferred back into nitrate (Wassenaar, 1995). The environmental conditions necessary for denitrification to occur include the availability of organic carbon, iron or manganese as electron donors, the presence of denitrifying bacteria, and reducing (anoxic) conditions (Starr and Gillham, 1993).

4 Methods

4.1 Field Sampling

All field sampling and laboratory analyses followed an approved Quality Assurance Project Plan (Mitchell et al., 2002). The quality control sections for this study are presented in Appendix A, beginning on page 110.

The criteria used to select groundwater sampling locations included the spatial distribution of wells, depth below the water table, road access, land owner agreement, existence of a well log, detection of nitrate levels from previous studies, and presence of hose-barb fittings for flow cell chamber.

The well-depth assignments were based on median depth below the water table, where shallow is <25 ft and deep is >25 ft (Figure 2, page 43; Table 4, page 40). The well identification letters were abbreviated based on the road names nearest

the site, where H is for Halverstick Road, P is for Pangborn Road, V is for Van Buren Road, T is for Trap Line Road, and K is for Kraght Road. The shallow wells included: H1, H2, H5, H6, H8, P3, T1, V1, V4, V5, V6, V9, and V10. The deep wells included: H3, H4, H7, K1, P1, P2, T2, V2, V3, V7, V8, V11, and V12. Well H7, a deep well, could be considered a shallow well because data indicate it is likely breached or has a leaking seal. In November of 2003, a 26th well was added, well V12, which is a deep well. Data at well H6 were collected sporadically due to problems with the onsite groundwater pump.

The static water level was measured at 18 accessible wells before water sampling commenced (H2–H6, P1, P3, T1, T2, V1, V2, V4, V5, V7–V9, V11 and V12). Groundwater samples were extracted from a standpipe at 25 residential wells (H1–H8, T1, T2, K1, P1–P3, and V1–V11) every other month for a 24-month period, beginning in July 2002. Thirteen of the 25 wells were monitored monthly and were chosen based on high nitrate values (H1, H2, H3, H4, H5, H7, T1, T2, K1, V5, V6, V8, and V9). The standpipe closest to the well was purged for approximately ten minutes, until the dissolved oxygen, specific conductance, and temperature values were stable. Samples and measurements were collected from a one-liter flow cell chamber in order to minimize atmospheric interactions. Dissolved oxygen, specific conductance, and temperature were measured using a YSI model 85 analyzer. Dissolved oxygen was calibrated at every site to the nearest 100 feet of elevation and within 95–105% saturation. Before each day of sampling, conductivity was checked using a known standard and temperature was checked using a mercury thermometer.

Three bottles were used to collect samples at each site. The samples that were analyzed in the laboratory for nutrients, including nitrate, ammonium, total phosphorus, and total nitrogen, were collected in 500 mL 2 N hydrochloric acid-washed Nalgene bottles. The samples analyzed for chloride were collected in 500 mL soap-washed Nalgene bottles. Those analyzed for iron and manganese were collected in 60 mL 20% nitric acid-washed Nalgene bottles. The 500 mL bottles were rinsed three times with the sample prior to collection; the 60 mL bottles were not rinsed as they contained ultra pure nitric acid for metal preservation. All samples were packed on ice for transport.

Nitrate-nitrogen isotope samples were collected in November of 2002 (23 wells), February 2003 (4 wells), May of 2003 (24 wells), September 2003 (22 wells), January 2004 (24 wells) and in May 2004 (25 wells). Well V2 was never sampled because of low nitrate values. Samples with nitrate ranging from 0.3–2.0 mg

N/L were collected in 2 N hydrochloric acid-washed 500 mL bottles; samples in excess of 2.0 mg N/L were collected in 2 N hydrochloric acid-washed 250 mL bottles. After collection, samples were filtered in the lab with a 0.45-micron 2 N hydrochloric acid- soaked filter, preserved with sulfuric acid to a pH of 2, and shipped on ice to the Colorado Plateau Stable Isotopes Laboratory at Northern Arizona University for analysis.

The ratio of stable nitrogen isotopes from nitrate (nitrate-nitrogen isotopes) can be used to differentiate among inorganic-commercial fertilizers, septic field leaks, and animal wastes versus in-situ soil nitrogen (Aravena et al., 1993; Wassenaar et al., 1995). Nitrogen is composed of two isotopes, the lighter ^{14}N , with a natural abundance of approximately 1% and the heavier ^{15}N , with a natural abundance of approximately 99%.

The notation used to express the high abundance of ^{15}N is “delta N-15” ($\delta^{15}\text{N}$). Delta values are the relative differences between the sample ratio ($^{15}\text{N}/^{14}\text{N}_{\text{sample}}$) and the standard ratio ($^{15}\text{N}/^{14}\text{N}_{\text{standard}}$) (Kendall, 1998). Measurements are in per mil ($^{\circ}/_{\text{oo}}$) and the standard is atmospheric air, where:

$$\delta^{15}\text{N} = \frac{\frac{^{15}\text{N}}{^{14}\text{N}_{\text{sample}}} - \frac{^{15}\text{N}}{^{14}\text{N}_{\text{standard}}}}{\frac{^{15}\text{N}}{^{14}\text{N}_{\text{standard}}}} \times 10^3$$

A wide range of $\delta^{15}\text{N}$ values can occur from a single source because of soil microbial activities, such as ammonia volatilization and nitrification, making values hard to interpret. Ranges presented by Wassenaar (1995) provide a guideline to determine source identification and serve as a basis for comparing results from this study to previous isotope work done by the 1997–1999 studies. In groundwater, inorganic commercial fertilizers result in values ranging from $-1^{\circ}/_{\text{oo}}$ to $+2^{\circ}/_{\text{oo}}$. Animal wastes have delta values from $+8^{\circ}/_{\text{oo}}$ to $+16^{\circ}/_{\text{oo}}$. Values between $+2^{\circ}/_{\text{oo}}$ and $+8^{\circ}/_{\text{oo}}$ should then indicate a mixture of inorganic commercial and animal fertilizers or derived from natural soil nitrate. Human waste from septic tank effluent would fall in the category of animal waste. However, Cox and Kahle (1999) determined that $<4\%$ of nitrogen in the region is related to septic tanks. The $\delta^{15}\text{N}$ technique can be inaccurate where denitrification occurs because fractionation due to nitrate reduction causes enrichments in $\delta^{15}\text{N}$ from the original signature.

Surface water samples were collected at 5 sites along Pangborn Creek (PB1–PB5) and 5 sites along Johnson Creek (JN1–JN5) every other month for 24 months,

beginning in July 2002 (Figure 2, page 43). The criteria used to determine surface water sampling locations were spatial location, land owner permission, and data from previous studies (e.g., Wills, 1998 and Nanus, 2000).

Dissolved oxygen, specific conductance, and temperature were measured by placing the YSI model 85 analyzer probe in the center of the channel and the center of the water column (when possible). Dissolved oxygen was calibrated at every site to the nearest 100 ft of elevation. Calibration checks for specific conductance and temperature were performed in the laboratory prior to sampling. Dissolved oxygen was calibrated in the laboratory before each day of sampling. The acceptable calibration range was 95–105% saturation.

Four bottles were used to collect samples at each site to analyze pH, nutrients, chlorides, metals, and fecal coliform. Fecal coliform samples were collected in a 125 mL plastic container and placed on ice until transport to Avocet Environmental Laboratories in Bellingham. Sample bottles were rinsed with sample three times prior to collection (except for fecal coliform and metals). All water samples were collected facing upstream in the deepest part of the channel or the area with maximum flow velocity. When necessary, a reaching pole with sample-bottle at the end of it was used to create a minimum amount of disturbance to bottom sediments. When possible, samples were collected at elbow depth. All samples were packed on ice for transport.

Nitrogen isotope samples were collected from Sites PB1, PB3, PB4, PB5, JN1, and JN4 in May and October 2003 and February and June of 2004 (PB1 was not sampled in October 2003). Sites PB2, JN2 and JN3 were sampled in October 2003 and February and June of 2004. Site JN5 was sampled in February 2004. Samples were collected according to the previous months nitrate concentrations: nitrate from 0.3–2.0 mg N/L were collected in 2 N hydrochloric acid-washed 500 mL bottles and samples in excess of 2.0 mg N/L were collected in 2 N hydrochloric acid-washed 250 mL bottles. After collection, samples were filtered in the lab with 0.45 micron 2 N hydrochloric acid-soaked filter, preserved with sulfuric acid to a pH of 2, and shipped on ice to the Colorado Plateau Stable Isotopes Laboratory at Northern Arizona University for nitrogen isotope analysis.

4.2 Analytical Methods

All water samples except fecal coliforms and stable isotopes were analyzed at the IWS laboratory.² Fecal coliforms were analyzed by Avocet Environmental Testing (Bellingham, Washington), nitrogen isotope samples were analyzed by Colorado Plateau Stable Isotopes Laboratory (Northern Arizona University, Flagstaff, Arizona). Standard operating procedures adapted from APHA (1998) were used for all IWS analytical work (see Appendix A, Table 5). Temperature, conductivity, and dissolved oxygen were measured in the field using a calibrated YSI field meter. Quality control samples for Winkler dissolved oxygen analyses were collected using 300 mL BOD bottles; Winkler reagents were added in the field. Sterile Nalgene bottles provided by Avocet were used to collect fecal coliform samples. Combined water samples for lab conductivity (QC), pH, total phosphorus, total persulfate nitrogen, soluble reactive phosphate, ammonia, and nitrate+nitrite were collected in acid-washed 1-L Nalgene bottles. All samples were transported back to the laboratory on ice. The water quality data are included in electronic format on the CD that accompanies this report and in ASCII files linked to the web version of this report (<http://www.ac.wvu.edu/~iws>).

5 Groundwater Quality Results

A total of 466 groundwater samples from 26 wells were analyzed over the 24-month period. An additional 217 samples were analyzed for quality control. Due to lab error, nitrate was not determined in May 2004 for wells K1, P1, P2, P3, V1, V2, V3, V4, V5, V6, V11, and V12; and in June 2004 for wells H1, H2, H3, H4, H5, H7, K1, T1, T2, V5, V6, V8, V9, and V12. Total nitrogen was determined for these wells, which is highly correlated to nitrate. Boxplots were used to present summaries of the data (Figure 9, page 50)

²The Institute for Watershed Studies is accredited by the Washington State Department of Ecology (Accreditation #A006).

5.1 Nitrate

Most (87%) of the nitrate values and 21 well-median nitrate values were greater than 3 mg N/L and indicate probable anthropogenic contamination. Both Tesoriero and Voss (1997) and Cox and Kahle (1999) used 3 mg N/L as a conservative estimate to represent nitrate concentrations in the aquifer due to anthropogenic sources. Background nitrate concentrations are probably less than 1 mg N/L (Cox and Kahle, 1999). In addition, 64% of all values and 14 of the well-median values were greater than the 10 mg N/L MCL (Figure 10, page 51). The strong positive correlation between groundwater nitrate and total nitrogen indicates that the dominant form of nitrogen in the study area is nitrate (Figure 11, page 52). The correlation between total nitrogen and nitrate was expected because total nitrogen is predominantly nitrate in most aerobic environments (Figure 12, page 53).

5.2 Ammonia

Ammonia concentrations were often lower than their detection limits of 21 $\mu\text{g N/L}$ (Figure 13, page 54). The higher ammonia values were likely associated with denitrification byproducts. For example, well V12 had unusually high concentrations of ammonia on more than one sampling date. Denitrification was substantiated by the low nitrate and dissolved oxygen, and high iron and manganese levels at V12.

5.3 Nitrogen Isotopes

Most of the 123 groundwater samples analyzed from the 25 wells over the 24-month period indicated a mix of organic and inorganic nitrogen as the source of the nitrate (Figure 14, page 55). Nitrogen isotopes were never determined at well V2 because of its historically low nitrate concentrations. With the exception of wells H6, V5, and V12, each well was sampled at least 5 times. Wells that demonstrated variability in $\delta^{15}\text{N}$ values were H1, H2, H6, and V4. The median $\delta^{15}\text{N}$ from wells V7 and V11 were within the inorganic, commercial fertilizer range ($\delta^{15}\text{N} < +2\text{‰}$). The median $\delta^{15}\text{N}$ from ten wells (H6, H7, K1, P1, P2, V1, V5, V6, V9, and V12) were within the animal manure range ($\delta^{15}\text{N} > +8\text{‰}$). It is possible that enrichment due to denitrification was responsible for the elevated

$\delta^{15}\text{N}$ values at wells K1, P1, P2 and V12. Samples from the rest of the wells (H1, H2, H3, H4, H5, H8, P3, T1, T2, V3, V4, V8 and V10) had $\delta^{15}\text{N}$ values in the mixed-source Range ($-1\text{‰} < \delta^{15}\text{N} < +8\text{‰}$).

5.4 Chloride

The chloride concentrations were typically $<10\text{ mg/L}$ (Figure 15, page 56). Cox and Kahle (1999) estimated background chloride levels to be 0.5 to 4 mg/L for the Sumas aquifer. They also state that chloride concentrations above background and below 25 mg/L were slightly elevated. The highest values measured in our study were below 25 mg/L (K1, T2, V5 and V6). Chloride is a component in animal manure and domestic sewage. Wells that demonstrated similar nitrate and chloride trends (highs and lows) include H6, H7, V4, V5 and V6 in the central portion of the study area. In general, there was a weak correlation between nitrate and chloride (Figure 16, page 57).

5.5 Conductivity

Median conductivities ranged from $<125\text{ }\mu\text{S/cm}$ in well V11 to $>400\text{ }\mu\text{S/cm}$ in well V5 (Figure 17, page 58). Wells that had high chloride concentrations also had high conductivities. Temperature changes, dilution from precipitation, and ions in fertilizer also cause the variation.

5.6 pH

The well median pH values fell within the range of 5.93 to 8.25 (Figure 18, page 59). Many of the wells had pH levels lower than the EPA secondary standards for drinking water (6.5 – 8.5); however, the ranges were fairly typical for groundwater.

5.7 Temperature

The median groundwater temperature value for all the wells was 10.60 ° C. There was a high variability in groundwater temperatures for some wells, which is not atypical when sampling domestic wells, especially from outside taps (Figure 19, page 60).

5.8 Dissolved Oxygen

The dissolved oxygen concentrations were variable at almost all sites (Figure 20, page 61). The range in groundwater dissolved oxygen concentrations is not unusual. Higher oxygen concentrations are common as temperatures decrease (less bacterial activity and more oxygen dissolved in the colder water) or when groundwater becomes aerated from pumping activities. Low oxygen concentrations are common in groundwater due to bacterial respiration during denitrification. Denitrification may be responsible for the low dissolved oxygen values measured in wells V1, V2, V3, V7, V11, V12, P1, P2, and K1 (and V4 during the summer). There was a weak correlation between nitrate and dissolved oxygen (Figure 21, page 62).

5.9 Phosphorus

Phosphorus in water is not considered directly toxic to humans so no drinking water standards have been established by the EPA. The EPA recommends that total phosphorus concentrations should be less than 0.1 mg/L in rivers, and less than 0.05 mg/L where rivers enter lakes and reservoirs to avoid contributing to eutrophication. Except for some outliers, the groundwater phosphorus concentrations were typically less than 0.020 mg/L (Figure 22, page 63).

5.10 Iron and Manganese

A condition that can enhance denitrification potential is the presence of desorbed metals. Well P2 had the highest levels of iron and manganese (0.24 mg/L and 0.09 mg/L, respectively). Wells V1, V2, V10, and V12 also had elevated (>0.10 mg/L)

median metals concentrations (Figures 23–24, pages 64–65). Field notes indicated that residents at Site P1 installed a filter to mitigate the high iron concentrations. A likely source of iron and manganese are iron-manganese nodules known to exist in peat in the region. Abiotic iron corrosion and microbial iron reduction are possible pathways for nitrate loss in the peat deposits.

5.11 Water Level

Water table levels were measured at 18 wells before extracting samples. The water table at a respective well was deepest in the fall (usually October) and shallowest in the spring (usually April). The median annual water table fluctuation was 5.60 ft, with a range between 2.0 ft at V12 and 15.0 ft at T2 (Figure 25, page 66). The high value at T2 was unique because T2 is an irrigation well that could not be turned off during sampling. The drawdown during summer pumping at well T2 was 8 to 10 ft. Water-level hydrographs for the northern (H2) and southern (T1) regions of the study area typify seasonal fluctuations (Figure 26, page 67). The median depth to the water table was used to estimate the depth of a respective well below the water table (Table 4, page 40).

We assumed that groundwater flow is primarily horizontal. Accordingly, we applied a regression technique described in Kelly and Bogardi (1989) to estimate the magnitude and direction of the hydraulic gradient. We used water-table elevations and UTM coordinates for 18 wells in our study area and 4 wells north of the study area in BC. The resulting hydraulic gradient for a high water-table condition (November 2003) was 0.0053 with a direction of 155.5° . The hydraulic gradient for a low water-table state (April 2004) was 0.0056 with a direction of 159.1° . These values are consistent with Hii et al., (1999) who estimated the hydraulic gradient to be between 0.003 and 0.008 north of our study area in BC.

We estimated the horizontal-hydraulic conductivity of the aquifer material in the study area using specific capacity data from 8 wells, and the techniques described in Cox and Kahle (1999). The mean, median and geometric-mean hydraulic conductivities were 475, 585, and 532 feet per day, respectively. These values correspond to a sand/gravel (Freeze and Cheery, 1979), which is typical of our study area. The average horizontal pore-water velocity is 10 feet per day ($\approx 157^\circ$) using a hydraulic conductivity of 532 feet per day, an estimate of porosity of 0.30, and a hydraulic gradient of 0.0056.

6 Surface Water Quality Results

Creeks were sampled every other month for two years (13 times). A total of 109 surface water samples from 10 stream sites were analyzed over the 24-month period: 49 samples from 5 sites along Johnson Creek and 61 samples from 5 sites along Pangborn Creek (Figure 2, page 43). An additional 61 samples were analyzed for quality control. Two sites in Johnson Creeks (Sites 4 and 5) were not sampled during certain summer months because of low-flow conditions.

6.1 Nitrate

With the exception of Site PB1, all the stream nitrate values in Pangborn Creek were greater than 3 mg N/L, indicating probable anthropogenic contamination (Figure 27, page 68). The low nitrate values measured at Site PB1 were likely the result of denitrification that occurs in the peat bog adjacent to the site. The median nitrate value for Pangborn Creek was 8.95 mg N/L, with minimum and maximum values of 0.94 and 19.06 mg N/L, respectively. Nitrate values were usually lower in Johnson Creek. The median nitrate value for Johnson Creek was 6.46 mg N/L. The minimum and maximum values from Johnson Creek median were 3.53 and 8.80 mg N/L, respectively. Nitrate was not as highly correlated to total nitrogen in the surface water because of higher ammonia concentrations, especially in Pangborn Creek (Figures 28–29, pages 69–70). The creek sites show a temporal pattern of high summer nitrate concentrations and lower winter and spring concentrations (Figures 30–31, pages 71–72).

6.2 Ammonia

Ammonia concentrations ranged from non-detectable ($<21 \mu\text{g N/L}$) in both creeks to a maximum of $293.38 \mu\text{g N/L}$ in Pangborn Creek and $691.14 \mu\text{g N/L}$ in Johnson Creek (Figure 32, page 73). About 85% of the all the values were below $70 \mu\text{g N/L}$ and they tended to decrease down gradient in Pangborn Creek and up gradient in Johnson Creek. Ammonia concentrations were typically higher in Pangborn Creek (median = $25.70 \mu\text{g N/L}$) than in Johnson Creek (median = $8.42 \mu\text{g N/L}$). When an ammonia outlier measured at well JN5 (2.37 mg/L) is eliminated, the difference between the two creeks is more distinguishable (Figure 33,

page 74). The higher ammonia concentrations measured at PB1 were likely the result of organic decomposition of peat in the anoxic environment. A secondary source of ammonium is via dissimilatory nitrate reduction to ammonium in the peat. The timing of peaks in ammonia concentrations along the creeks varied and likely coincided with agronomic loading and precipitation or irrigation events (Figures 34–35, pages 75–76). For example, two peaks in ammonia concentrations (which correlate to peaks in other measured surface water parameters) occur on sampling dates in April 2003 and October 2003. The amount of rainfall was large during the week prior to these sampling dates, 1.1 inches in April and 2.0 inches in October.

6.3 Nitrogen Isotopes

A total of 33 surface water samples were analyzed for nitrogen isotope magnitudes. The median nitrogen isotope value from Pangborn Creek was $10.9 \delta^{15}\text{N}$ (Figure 36, page 77). The range was between 7.4 and $12.3 \delta^{15}\text{N}$. The isotope values were similar in Johnson Creek (median of $10.9 \delta^{15}\text{N}$ and a range between 8.3 and $11.9 \delta^{15}\text{N}$). These $\delta^{15}\text{N}$ values are indicative of a manure source. It is also possible that the isotopes were enriched due to denitrification processes in the bog.

6.4 Chloride

The chloride concentrations ranged from 5.72 mg/L to 14.70 mg/L in Pangborn Creek and between 10.0 mg/L and 14.70 mg/L in Johnson Creek (Figure 37, page 78). The median value in Johnson Creek (11.20 mg/L) was higher than the median in Pangborn Creek (9.85 mg/L). The chloride values increased slightly downstream in Pangborn Creek. In most cases, the peaks in chloride concentrations correlated to peaks in stream nitrate (Figures 38–39, pages 79–80).

6.5 Conductivity

The conductivity values in both creeks increased and decreased with chloride concentrations. Measurements in Pangborn Creek ranged from 198 to 359 $\mu\text{S}/\text{cm}$ and

resulted in a median of 285 $\mu\text{S}/\text{cm}$. Conductivity values in Johnson Creek ranged from 282 to 332 $\mu\text{S}/\text{cm}$ and resulted in a median value of 300 $\mu\text{S}/\text{cm}$ (Figure 40, page 81).

6.6 Fecal Coliforms

The streams in the Johnson Creek watershed are Class A, as such, fecal coliform concentrations should not exceed 100 cfu/100 mL (Butkus et al., 2000). About 50% of the samples measured in both creeks exceeded 100 cfu/100 mL. Fecal coliform concentrations ranged from 5 to 1200 cfu/100 mL in Johnson Creek and from 2 to 12,000 cfu/100 mL in Pangborn Creek (Figure 41, page 82). The geometric mean values for Pangborn and Johnson Creeks were 125.1 and 95.8 cfu/100 mL, respectively. There was a much larger variability in fecal counts in Pangborn Creek. The highest fecal counts occurred in the summer months, in both creeks. The fecal geometric mean exceeded the target value of 37 cfu/100 mL in Pangborn Creek (Henderson, 2003) 10 out of the 13 sampling months. The fecal geometric mean exceeded the target value of 100 cfu/100 mL in Johnson Creek (Henderson, 2003) 7 out of the 13 sampling months. As in the chloride results, fecal coliforms increase downstream (Figures 42–43, pages 83–84). The timing of peaks in fecal colonies (like the ammonia concentrations) varied, and likely coincide with agronomic loading and precipitation or irrigation events. For example, the peaks in fecals in Pangborn Creek occur during the aforementioned heavy rain events in April and October 2003.

6.7 Dissolved Oxygen

The median dissolved oxygen concentration in Pangborn Creek was 5.66 mg/L and ranged between 1.02 mg/L to 12.51 mg/L (Figure 44, page 85). The streams in the Johnson Creek watershed are Class A, as such, dissolved oxygen concentrations should exceed 8.0 mg/L (Butkus et al., 2000). Of the samples measured in Johnson Creek, 27% exceeded 8.0 mg/L and 30% of the samples from Pangborn exceeded 8.0 mg/L. The lower values of dissolved oxygen occurred in the summer months when the biological oxygen demand was highest. The dissolved oxygen values were low near the head of the creek (in the anoxic peat bog) and increased down gradient due to oxygenated groundwater discharging into the stream and

stream aeration. The median dissolved oxygen concentration in Johnson Creek was 6.73 mg/L and ranged between 1.82 mg/L to 10.45 mg/L (Figure 44, page 85). The seasonal trends in dissolved oxygen in Johnson Creek were similar to those observed in Pangborn Creek. Although the up gradient concentrations were lower, there was a smaller difference between up gradient and down gradient dissolved oxygen values in Johnson Creek (Figures 45– 46, pages 86–87).

6.8 Temperature

The surface water temperature trends and magnitudes in both creeks were similar (Figure 47, page 88). The median temperature for Pangborn Creek was 49.73 °F (9.85 °C) and ranged between 38.12 °F (3.40 °C) and 65.84 °F (18.80 °C). Whereas the median for Johnson Creek was 50.54 °F (10.30 °C) and ranged between 40.64 °F (4.80 °C) and 58.82 °F (14.90 °C). Both creeks peaked in temperature in July and were coolest in December (opposite of the dissolved oxygen concentrations).

6.9 Phosphorus

The phosphorus concentrations were much greater in Pangborn Creek (median of 217.3 µg P/L) than in Johnson Creek (median of 89.7 µg P/L; Figure 48, page 89). The highest value (1502 µg P/L) was measured at Site PB1 near the mouth of Pangborn Bog in October 2003 (coincident with the heavy rainfall and fecal peak). The maximum value in Johnson Creek (265.5 µg P/L) occurred at Site JN5 in April 2003 during another heavy rainfall event, in concert with a second peak in Pangborn Creek (1015 µg P/L).

6.10 Iron and Manganese

Iron and manganese concentrations are elevated in the surface water primarily due to the iron and manganese concentrations in the peat that exists in the area. The mean iron concentrations in Pangborn and Johnson Creeks were 0.12 and 0.15 mg/L, respectively (Figure 49, page 90). The manganese medians were 0.01 and 0.04 mg/L, respectively (Figure 50, page 91). The iron (and manganese) concen-

trations increased in the winter and peaked in the spring, presumably because the rising water table encountered a larger area of peat prior to discharging into the creeks. The peak iron values in Pangborn and Johnson Creeks were 0.57 and 0.49 mg/L, respectively.

6.11 pH

The water discharging from Pangborn Bog is more acidic, therefore the median pH value for Pangborn Creek was lower (6.74) than the 7.14 median pH for Johnson Creek (Figure 51, page 92). In general, the pH tended to be higher in the summer and lower in the winter in both creeks. The highest pH values were recorded at Site PB5, whereas the lowest were measured near the bog (Site PB1). The EPA secondary standard for pH of drinking water ranges between 6.5–8.5.

7 Groundwater Quality Discussion

Numerous factors may influence a nitrate concentration measured at a particular well in the study area, including: southerly flowing groundwater having nitrate originating in British Columbia, vadose zone thickness and residence time, depth below the water table, up gradient land use including timings and types of nutrient loading, seasonal precipitation and irrigation, vertical and lateral mixing due to intermingling plumes along flow paths in the aquifer, mixing due to excessive well pumping, nitrate recirculation due to irrigation (Guimera, 1998), and denitrification (especially due to peat deposits in the area).

7.1 Nitrate Distribution

In general, nitrate concentrations were higher north of Pangborn Bog and Pangborn Creek due to a combination of transport from British Columbia and leaching in Whatcom County (Figure 52, page 93). All the wells north of the creek (except V12) had median nitrate values greater than 3 mg N/L, indicating anthropogenic sources (Tesoriero and Voss, 1997). Furthermore, 12 of the 15 wells north of the creek had median nitrate concentrations above the EPA MCL of 10 mg N/L. The nitrate concentrations from V12 were consistently below 1.0 mg N/L, likely due

to denitrification as suggested by the low dissolved oxygen and high iron concentrations measured at the well. Well V12 was added to the monitoring list in the fall of 2003 and was monitored 5 times between 11/03 and 3/04.

We assumed that deep wells H3, H4, H7, V7, and V8, and shallow wells V9 and V10 north of the creek withdraw groundwater derived upgradient in British Columbia. The median nitrate concentrations for these wells ranged from 4.0 to 18.6 mg N/L (Figure 10, page 51).³ Due to their depth and proximity to the border, it is unlikely that the elevated nitrate concentrations at these wells were caused by Whatcom County sources. It would require large vertical gradients to deliver nitrate to these wells and the vertical hydraulic gradients are minimal in the region (Zebarth et al., 1998). Moreover, the high nitrate values from these wells are consistent with the nitrate concentrations measured from four piezometers in the British Columbia Study Area that Environment Canada monitors on a monthly basis (Figure 52, page 93). Median nitrate concentrations for our study period from piezometers BC3, BC4, BC5, and BC6 just north of our study area were 17.0, 8.2, 13.5 and 22.4 mg N/L, respectively (Figure 53, page 94).

The peat bog and creek serve as a discharge zone (and nitrate sink) for the shallow, nitrate-contaminated groundwater from the north (McKee, 2004). The low nitrate concentrations south of Pangborn Bog and Pangborn Creek are due to denitrification that occurs in the reducing, iron-rich peat in the bog and along portions of the creek. McKee (2004) used a combination of chemical indicators to document denitrification in the peat. These include low nitrate concentrations, low dissolved oxygen concentrations, presence of iron and manganese, enriched $\delta^{15}\text{N}$ signatures, a linear relationship between nitrogen and oxygen isotope values, and detection of N_2 gas measured in and around the bog (e.g., P1, P2, and K1).

Hydrostratigraphic data suggest that peat deposits may occur in other areas of the study area, at various unmapped depths (e.g., Rigg, 1958; Kovanen and Easterbrook, 2002; D. Easterbrook, personal communication, WWU Geology Dept., 2003). Hence, groundwater nitrate may be reduced locally in the vicinity of unmapped peat deposits. Because of indicators such as low dissolved oxygen, high metals, or enriched $\delta^{15}\text{N}$ values, it is likely that such deposits and denitrification are responsible for low nitrate concentrations measured from wells P3, V1, V2, V3, V4, V7, V11, V12, and possibly V10.

³The anomalously low nitrate values measured at V7 may have been caused by denitrification, which was suggested by the low dissolved oxygen values recorded at the well.

7.2 Nitrate Stratification

When wells with low nitrate values in the entire study area (likely due to denitrification) are eliminated, the greatest contamination occurred in shallow wells (<25 ft), which produced a median value of 15.7 mg N/L. The deep well (>25 ft) nitrate concentrations were lower in magnitude, and resulted in a median value of 11.3 mg N/L. This stratification in nitrate concentrations is in agreement with historical nitrate measurements in the study area (e.g., Mitchell et al., 2003) and the British Columbia study area (Hii et al., 1999). The highest nitrate concentrations (>20 mg N/L) occurred in shallow wells T1, V5, V6, and V9 (and H7, a deep well assumed to be breached). Except for V9, the high nitrate concentrations in shallow wells were likely due to agricultural practices in Whatcom County that contributed to the regional nitrate contamination originating in British Columbia.

The high nitrate concentrations measured in a number of the deep wells (and shallow well V9 near the border) were in part due to British Columbia transport. Large pumping wells can also draw shallow, more contaminated water deeper into the aquifer, thus skewing the deeper nitrate concentrations. This process was observed in an irrigation well, T2. Distinct summer drawdowns were observed at T2, which correspond to summer nitrate peaks (Figure 54, page 95).

7.3 Nitrate Sources

The agricultural land use in the study area is primarily raspberries and crops grown for dairy cow forage (grasses and corn). Raspberries are fertilized in February (inorganic fertilizer) and March (manure) and dairy manure can be spread on corn/grasses anytime between February and October. Irrigation water can be applied between May and August on raspberries and between April and September on grasses; the largest magnitudes are applied between June and August.

Median values of $\delta^{15}\text{N}$ measured from wells in the study area ranged from -0.7 to 28.6‰ (Figure 14, page 55). Because the $\delta^{15}\text{N}$ values were derived from groundwater samples, it is probable that denitrification and mixing are in part responsible for the range of $\delta^{15}\text{N}$ values. Most of $\delta^{15}\text{N}$ values suggest organic manure, or a mix of organic and inorganic nitrogen as the source of the nitrate (between $+2\text{‰}$ and $+8\text{‰}$), an interpretation consistent with the mixed fertilization practices in the study area (Figure 55, page 96). Wells having $\delta^{15}\text{N}$ in the mixed range include

H1, H2, H3, H4, H5, H8, P3, T1, T2, V3, V4, V8 and V10. Most wells showed little variability in $\delta^{15}\text{N}$ values in time, except for wells H1, H2, and H6 which had values in the animal source range ($>+8\text{‰}$) in August of 2004, and V4 had values in the animal source range in December of 2002 and 2003.

The median $\delta^{15}\text{N}$ values measured at 10 wells (H6, H7, K1, P1, P2, V1, V5, V6, V9, and V12) were within the animal manure range ($\delta^{15}\text{N} > +8\text{‰}$). Wells H6, H7, V5 and V6 are in the central portion of the study area that is dominated by lands that receive dairy manure and irrigation water. Well V9 is close to the British Columbia border, its $\delta^{15}\text{N}$ value likely indicates poultry manure as the nitrogen source. Although the $\delta^{15}\text{N}$ values measured at wells K1, P1, P2, V1 and V12 are in the animal range, they are excessively high. Because of their location to peat, it is likely that isotopic enrichment from denitrification was responsible for the high values.

Wells V7 and V11 consistently had $\delta^{15}\text{N}$ values indicative of an inorganic fertilizer (between -1‰ and $+2\text{‰}$). The land use upgradient of well V7 is mixed, but raspberries dominate the landscape further to the north in Whatcom County, and across the border in British Columbia. Since well V7 is categorized as a deep well, it is likely that the low $\delta^{15}\text{N}$ values are indicative of inorganic, commercial fertilizers applied to these raspberry fields. The $\delta^{15}\text{N}$ values measured at well V7 during the 1997–1999 study were also below $+2\text{‰}$. The shallow well V11 however, is not directly down gradient from fields which receive dominantly inorganic fertilizers. Since the nitrate concentrations were low ($<3\text{ mg N/L}$) at well V7, the source is not of key importance.

Deep well V8 and shallow wells V9 and V10 near the British Columbia border in the northeast portion of the study area are likely receiving groundwater from British Columbia, as are H3 and H4 along Halverstick Road. The $\delta^{15}\text{N}$ signatures from all of these wells revealed a mix of organic and inorganic commercial fertilizers, consistent with the land use north of the border, which is primarily raspberry fields. There has been an increasing trend toward the reduction of poultry manure and an increase in inorganic fertilizers on raspberries in this region in recent years (G. Hughes-Games, personal communication, BC Ministry of Agriculture, Food and Fisheries, 2004). Poultry manure was used dominantly in the British Columbia region during our previous monitoring study (1997–1999). As such, the $\delta^{15}\text{N}$ signatures were useful for distinguishing British Columbia sources of nitrate from Whatcom County sources. For example, the $\delta^{15}\text{N}$ values measured in 1997–1999 from V10 were in the manure range (Mitchell et al., 2003), as ex-

pected. Because of the introduction of inorganic, commercial fertilizers in recent years in the British Columbia region, nitrogen isotopes are not as useful for isolating British Columbia sources in Whatcom County. Moreover, because of the potential for denitrification and $\delta^{15}\text{N}$ enrichment, a combination of nitrogen and oxygen isotope analyses on the nitrate should be applied to distinguish organic from inorganic sources the future.

7.4 Nitrate Trends

Nitrate concentrations were plotted by month for each well to check for site-specific, seasonal patterns (Figures 56–60, page 97– 101). The nitrate-time series from nearly all the wells displayed some kind of pattern, many showing highs and lows of varying degrees, and at sporadic times. The time series for the four British Columbia wells also show varying patterns (Figure 61, page 102). In many cases it is difficult to interpret the pattern observed at a single well because it can be controlled by any or all of the aforementioned factors. The time-series patterns were divided into three groups: wells having nitrate peaks in the fall/winter/spring; wells having nitrate peaks in the summer; and wells with no consistent nitrate pattern.

Fall/Winter/Spring Nitrate Peaks Wassenaar (1995) concluded that nitrification in the Abbotsford-Sumas aquifer occurred during the summer months; in the fall and winter, nitrates would be flushed through the unsaturated zone due to high precipitation rates and a rising water table. This process produces low nitrate concentrations in the summer and peaks in the fall and winter.

The nitrate time series from only two wells, P3 and H8 in our study area revealed this pattern. Both are shallow wells down gradient of raspberry fields. Raspberry fields are not fertilized after May or irrigated in late summer, so it is possible that the soil is storing nitrogen and releasing nitrate in the fall/winter. The $\delta^{15}\text{N}$ values for these two wells indicate that the nitrate source is mixed, consistent with the mix of organic and inorganic fertilizers the raspberries receive. An attenuated version of this pattern occurred at well P1. The attenuation was due to denitrification, which was supported by low nitrate, low dissolved oxygen, and enriched $\delta^{15}\text{N}$ values.

The temporal pattern at well H7 depicted a nitrate spike from December to March, consistent with the fall leaching process discussed above; however, after the 2003 spike (43 mg N/L maximum), the nitrate concentrations dropped and hovered around 10 mg N/L. A nitrate spike of this magnitude is not typical of a well this deep; therefore, a breach likely exists in the casing of well H7 that allowed localized up gradient sources to influence the groundwater quality sooner. The high $\delta^{15}\text{N}$ values indicate that manure was the source of nitrate in the groundwater sampled from H7. Historical evidence of this pattern is provided by data from an unpublished study by Ecology from 1994–1997. Groundwater nitrate of approximately 40 mg N/L in well H7 was observed shortly after heavy manure loading to the same grass field occurred (D. Garland, personal communication, Washington State Department of Ecology, 2004). Within months after loading and during no loading intervals, the nitrate levels dropped to approximately 10 mg N/L, consistent with the other deeper wells along Halverstic Road. Similar large spikes were observed at H7 during the 1997–1999 study (Mitchell et al., 2003).

Summer Nitrate Peaks The nitrate concentrations in five wells (K1, T1, V1, V4, and V7) had summer peaks. One factor that was responsible for the summer peaks was (as discussed above) higher summer pumping rates which drew shallow, more contaminated water deeper into the aquifer. Because of the conductive aquifer material this probably only occurred in irrigation wells that create larger cones of depression (typically < 10 feet). For example, the summer nitrate peaks observed at irrigation well T2 correlate with the summer water-level drawdowns recorded at T2 (Figure 54, page 95). A similar summer pattern was observed at K1, a water association well that also likely has higher summer pumping volumes.

Water-level data indicate that excessive pumping may not be responsible for the summer peaks recorded at shallow wells V1, V4, and V7. The large nitrate variation and summer peaks in V4 were unique, and mimicked the same nitrate pattern observed in Pangborn Creek (Figure 62, page 103). Well V4 is extremely shallow and close (< 200 feet) to Pangborn Creek; therefore, well V4 likely intercepts groundwater discharging to the creek. One hypothesis is that summer irrigation and early fall precipitation on the grasslands upgradient of V4 may be flushing stored nitrates through the thin, highly permeable unsaturated zone. This hypothesis is substantiated by a similar high summer chloride trend observed at the V4. Chloride is a constituent in manure which is the nutrient applied to the grasslands. The dissolved oxygen concentrations were also low (<2 mg/L) during the sum-

mer at well V4, meaning that denitrification may be occurring in the vicinity of the well, thus lowering the (already high) summer magnitudes. We cannot offer a reasonable explanation for the low, summer peaks observed in V1 and V7.

Inconsistent Trends The nitrate time series from wells H6, T1, V6, V8 and V10 showed periodic peaks with decreasing trends. Wells T1, V8 and V10 are down gradient of berry fields, and H6 and V6 are down gradient of mixed raspberry and grass/corn. Wells with random patterns, but with increasing trends include wells near the border, H1, H2, H5 and V9. Wells having inconsistent patterns and low nitrate concentrations (<5 mg N/L) include V2, V3, V11 and V12. Nonagricultural land use and denitrification were likely influencing the low concentrations at these wells. Wells H3 and H4 had unpredictable patterns and consistently high nitrate concentrations (>5 mg N/L). The inconsistent patterns reflect the variability in the timing of loading, irrigation, and mixing in the aquifer, or a combination of the aforementioned factors.

8 Surface Water Quality Discussion

The nitrate levels measured in both Pangborn and Johnson Creeks were above 5 mg N/L throughout the duration of the study (except for Site PB1 near the bog), indicating anthropogenic sources (Figure 27, page 68). Both creeks had higher summer nitrate concentrations and lower winter and spring concentrations (Figures 30–31, pages 71–72). The nitrate values in general, were usually lower in Johnson Creek. The creeks also had peaks in other water quality parameters that correlated to heavy precipitation events. The median nitrate concentration for Pangborn Creek was 8.95 mg N/L. The median nitrate value resulting from measurements along Pangborn Creek in November of 1997, February 1998 and May 1998 was 8.07 mg N/L (Nanus 2000). Both creeks are on the approved TMDL 303(d) list. Listed parameters include dissolved oxygen and fecal coliforms (Butkus et al., 2000; Henderson, 2003).

Pangborn Creek will be the focus of the surface water discussion because of its proximity to the peat deposits and location within the domestic-well sampling network. Water flow in Pangborn creek is sustained in part by discharge from the bog, groundwater, and by runoff from bordering lands (mainly shallow, subsurface flow). The surface material at the headwaters of the creek, along with Sites PB1

and PB2, is primarily peat. Field observations indicate that surface peat deposits extend as far east as stream Site PB3. As the creek flows eastward from Site PB3, the surface material changes to cobbles, gravels, sand, and silt up through Site PB5. The majority of the land use flanking the creek is agricultural (Figure 63, page 104).

The water discharged from Pangborn Bog was high in ammonia, but low in nitrate and dissolved oxygen due to denitrification in the peat (McKee, 2004). The sources of ammonia in the bog may include un-nitrified ammonia-based fertilizer supplied by drainage ditches to the north of Pangborn Bog and ammonia inherent in peat deposits (from organic decomposition in an anoxic environment). A secondary source of ammonia is via dissimilatory nitrate reduction to ammonia in the peat. Ammonia levels decreased downstream in the creek, with some months below the detection limit of 0.02 mg N/L (Figure 32, page 73).

Ammonia levels decreased downstream due to nitrification, or incomplete redox reactions, such as ammonium oxidizing to nitrite, and dilution by groundwater discharge. Nanus (2000) attributed the downstream decrease in ammonium concentrations to nitrification in the more aerobic part of Pangborn Creek. Higher surface-water temperatures likely amplify nitrification reactions, thus causing the higher summer nitrate concentrations in the creeks and a reduction in ammonia (Figure 64, page 105). Groundwater discharge to the creek is also diluting bog discharge, as confirmed by a decrease in ammonia and increase in dissolved oxygen downstream, and by water quality similarities between the creek and extremely shallow well V4 close to Pangborn Creek. Nitrate and chloride (and ammonia, to a lesser degree) at Site PB4 and from V4 followed comparable trends. Moreover, these values became nearly identical in the late summer and early fall when groundwater is the dominant source of water in the creek (Figure 62, page 103).

The water quality in Pangborn Creek (and Johnson Creek) was also influenced by surface or shallow subsurface runoff due to precipitation and irrigation events. Winter rains and runoff elevated the winter creek chloride and nitrate concentrations in the creeks. These high concentrations were not observed in the groundwater at shallow well V4 in the winter. Peaks in ammonia, fecal coliform, and chloride concentrations were also recorded in the creeks just after heavy rainfall events in April 2003 (1.1 in of rain) and in October 2003 (2.0 in of rain). Smaller spikes in these parameters may have been due to irrigation, or lighter rainfall events which transported the chemicals via shallow groundwater to the stream. The fecal counts were higher in the summer months due to irrigation and summer

rain. Irrigation magnitudes for grasses in July reached an estimated value of 4 to 6 in (J. Gillies, personal communication, USDA Natural Resource Conservation Service, 2004).

The $\delta^{15}\text{N}$ values measured in the creeks are indicative of a manure source (Figure 36 and Figure 65, pages 77 and 106), which is consistent with the grass and corn crops that flank Pangborn creek. It is also possible that the isotopes were enriched due to denitrification processes in the bog. The correlation between creek nitrate and chloride also suggest manure as the source (Figure 66, page 107).

9 DNMP Assessment

One of our objectives was to assess the effectiveness of Dairy Nutrient Management Plans (DNMPs). These plans were to be implemented by December of 2003 in accordance with the State of Washington Dairy Nutrient Management Act (RCW 90.64). While we were able to compare surface and ground water nitrate at sites measured before and after the implementation of the DNMPs, direct correlations between nitrate concentrations measured at a well and nutrient-loading data from specific fields cannot be performed without extensive soil and soil pore-water data, shallow peizometer data, and numerical modeling. This was beyond the scope of our project.

The first method used to assess the effectiveness of DNMPs was to compare ground water nitrate concentrations measured between 1997–1999 to those measured in 2002–2004. Fourteen of the wells sampled between July 2002 and June 2004 were also sampled monthly between April 1997 and February 1999 (H1, H2, H5, H6, H7, H8, T1, T2, V5, V6, V7, V8, V10, and V11). The 2002–2004 nitrate concentrations increased in 7 of the 14 wells (Figure 67, page 108). The overall nitrate median from these wells increased from 11.5 mg N/L in 1997–1999 to 13.6 mg N/L in 2002–2004. We also assessed the effectiveness of DNMPs by comparing ground and surface water nitrate before and after the DNMPs were implemented within the 2002–2004 sampling period. Assuming that most plans were implemented by November of 2003, we determined the median nitrate concentration of each well for November 2002 to April 2003 and compared it to the medians from November 2003 to April 2004. Out of 24 wells, 15 wells had median nitrate concentrations that increased after implementation (Figure 68, page 109). The study area median (median of all the wells except H6 and V12) for

November 2002 to April 2003 was 11.3 mg N/L. The median increased to 12.1 mg N/L after implementation (November 2003 to April 2004). We also determined the nitrate medians after excluding data from the wells that are suspected of being influenced by denitrification. The wells that did not appear to be influenced by denitrification (H1, H2, H3, H4, H5, H7, H8, T1, T2, V5, V6, V7, V9, and V10) had median nitrate concentrations of 13.5 mg N/L in 1997–1999 compared to 13.2 mg N/L during 2002–2004.

In order to explore the influence of BC transport on nitrate concentrations in our study area, we estimated the background concentration from BC by determining the median nitrate concentration from the 4 piezometers just north of the study area in BC (Sites BC3–6). The median nitrate concentration decreased from 17.60 mg N/L (November 2002 through April 2003) to 16.0 mg N/L (November 2003 to April 2004), which is consistent with the general trend in BC (B. Hii personal communication, Environment Canada). Because of the transport time (\approx 10 feet per day) the higher 2003 concentrations in BC may be responsible for the elevated concentrations in our study area in 2004.

We also compared surface water nitrate medians for the following two time periods: December 2002 to April 2003 and December 2003 to April 2004 (sampling was not performed in November). Johnson Creek had a median nitrate concentration of 5.3 mg N/L for the year before implementation of the DNMPs (December 2002 to April 2003). The median nitrate concentration increased to 7.2 mg N/L after implementation (December 2003 to April 2004). The median nitrate concentration for Pangborn Creek increased from 7.5 mg N/L before implementation to 9.2 mg N/L after implementation. The total rainfall during these two time periods was roughly equivalent: 21.9 inches between December and April 2002–2003 and 19.1 inches between December and April 2003–2004.

Although our results suggest that nitrate concentrations are not decreasing in the study area, the nitrate levels in 5 wells (H6, T1, V6, V8 and V10) did have decreasing trends in 2002–2004. Because there are many spatial and temporal factors that influence the nitrate concentration at individual wells, we recommend further study, particularly at the sites that showed decreasing nitrate levels, to help identify whether specific DNMPs have been effective at reducing nitrate.

10 Conclusions

Elevated nitrate concentrations as a result of agricultural sources have persisted in groundwater and surface water in the region, despite the implementation of best management practices. Our results suggest that nitrate in excess of 10 mg N/L is transported in the aquifer from British Columbia, into Whatcom County. Local sources contribute to this regional plume, resulting in high concentrations (in excess of 20 mg N/L) in shallow groundwater in the central portion of the study area. The nitrate-time series from the wells showed a mix of patterns, mainly due to the mix of spatial and temporal variables that can influence a concentration at a respective well. Only four wells showed a decreasing trend in nitrate and 14 of the 26 wells had medians above the nitrate MCL of 10 mg N/L.

Values of $\delta^{15}\text{N}$ measured from wells validate that the source of the nitrate is a mix of organic manure and inorganic commercial fertilizer; however, the manure component was higher. Only two wells yielded $\delta^{15}\text{N}$ values suggestive of inorganic nitrogen. These results are consistent with the nutrient loading practices that exists in both British Columbia and Whatcom County. The shift toward more inorganic fertilizers in British Columbia in recent years makes it more difficult to distinguish British Columbia sources from Whatcom County sources.

Nitrate concentrations were significantly lower south of Pangborn Bog and Pangborn Creek due to denitrification in the organic and iron rich peat. We also suspect denitrification is responsible for the anomalously low nitrate concentrations measured at other locations in the study area because peat occurs throughout this region at various unmapped depths. The implication of these findings is that a natural mechanism for nitrate reduction exists in this region. Identifying peat deposits and their influence on nitrate concentrations may help facilitate nutrient management in the region.

Nitrate concentrations were also high in Pangborn and Johnson Creeks. With the exception of one site near the bog, the nitrate concentrations were typically above 5 mg N/L and occasionally above 10 mg N/L. Concentrations in Pangborn Creek were higher than those in Johnson Creek. Values of $\delta^{15}\text{N}$ measured from the creeks validate that the source of the nitrate is organic manure which is coming from a combination of surface runoff and shallow groundwater. The fecal geometric mean exceeded the target value of 37 cfu/100ml in Pangborn Creek (Henderson, 2003) 10 out of the 13 sampling months. The fecal geometric mean exceeded the target value of 100 cfu/100ml in Johnson Creek (Henderson, 2003) 7 out of the

13 sampling months. The timing and magnitude of peaks in fecal colonies varied, and coincide with agronomic loading and precipitation or irrigation events.

Because of multiple sources in British Columbia and Whatcom County, conservationists and farmers will continue to face the problematic task and economic challenges of managing nutrient loading. To assess the affect of specific nutrient-loading plans in the study area, extensive soil and soil pore-water data, shallow peizometer data, and numerical modeling will have to be employed.

11 References

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12 Tables

Month	Activity	Loading Type	Monthly Net Net Irrigation Requirement (cm)	Application Method
January	x	x	0.00	x
February	preplant	manure	0.00	x
March	bulb-break	synthetic fertilizer	0.00	x
April	growth	x	0.00	x
May	growth	x	5.69	big-gun sprinkler
June	growth	x	9.73	big-gun sprinkler
July	harvest	x	14.10	big-gun sprinkler
August	x	x	8.89	big-gun sprinkler
September	x	x	1.17	big-gun sprinkler
October	x	x	0.00	x
November	x	x	0.00	x
December	x	x	0.00	x

Table 1: Possible monthly raspberry crop loading and irrigation activities.

Month	Activity	Loading Type	Monthly Net Net Irrigation Requirement (cm)	Application Method
January	x	x	0.00	x
February	nutrient up-take	manure	0.00	x
March	growth	manure	0.00	big-gun sprinkler
April	growth	manure	1.35	big-gun sprinkler
May	growth	manure	4.72	big-gun sprinkler
June	growth	manure	6.45	big-gun sprinkler
July	dormancy	manure	10.29	big-gun sprinkler
August	dormancy	manure	7.42	big-gun sprinkler
September	growth	manure	2.26	big-gun sprinkler
October	growth	manure	0.00	big-gun sprinkler
November	x	x	0.00	x
December	x	x	0.00	x

Table 2: Possible monthly grasses crop loading and irrigation activities.

Month	Activity	Loading Type	Monthly Net Net Irrigation Requirement (cm)	Application Method
January	x	x	0.00	x
February	x	x	0.00	x
March	preplant	manure	0.00	x
April	preplant	manure	0.00	x
May	preplant/starter	manure/synthetic	0.00	x
June	x	x	0.00	x
July	sidedress fertilizer	synthetic	5.69	big-gun sprinkler
August	x	x	8.43	big-gun sprinkler
September	harvest	x	3.40	big-gun sprinkler
October	x	x	0.00	x
November	x	x	0.00	x
December	x	x	0.00	x

Table 3: Possible monthly corn crop loading and irrigation activities.

Well ID	Ground Surface Elevation (ft above sea level)	Well Depth Below Ground Surface (ft)	Median Depth to Water Table (ft)	Median Depth Below Water Table (ft)	Shallow (<25 ft) or Deep (>25 ft)
H1	149.3	30.0	20.0*	10.0	S
H2	159.1	34.0	20.4	13.6	S
H3	163.2	80.0	24.5	55.5	D
H4	165.9	65.0	30.0	35.0	D
H5	161.4	31.0	24.7	6.3	S
H6	169.1	38.0	32.6	5.4	S
H7	165.7*	71.0	33.0*	38.0	D
H8	167.8	50.0	33.0*	17.0	S
K1	126.3*	49.0	20.0*	29.0	D
P1	137.6	55.0	28.4	26.6	D
P2	150.9	67.0	29.0*	38.0	D
P3	148.2	37.0	30.3	6.7	S
T1	142.1	43.0	28.0	15.0	S
T2	142.3	80.0	26.9	53.1	D
V1	91.1	36.0	18.3	17.7	S
V2	94.5	122.0	17.0	105.0	D
V3	146.0*	85.0	27.0*	58.0	D
V4	121.7	12.0	4.2	7.8	S
V5	151.2	20.0	18.6	1.4	S
V6	165.7*	41.5	34.0	7.5	S
V7	175.2	77.0	43.2	33.8	D
V8	183.5	95.0	57.3	37.7	D
V9	207.0	91.0	72.7	18.3	S
V10	198.5	90.0	73.0*	17.0	S
V11	86.2	45.0	14.5	30.5	D
V12	141.2	57.5	9.4	48.1	D

*Estimated values

Table 4: Abbotsford-Sumas well depth summary.

13 Figures

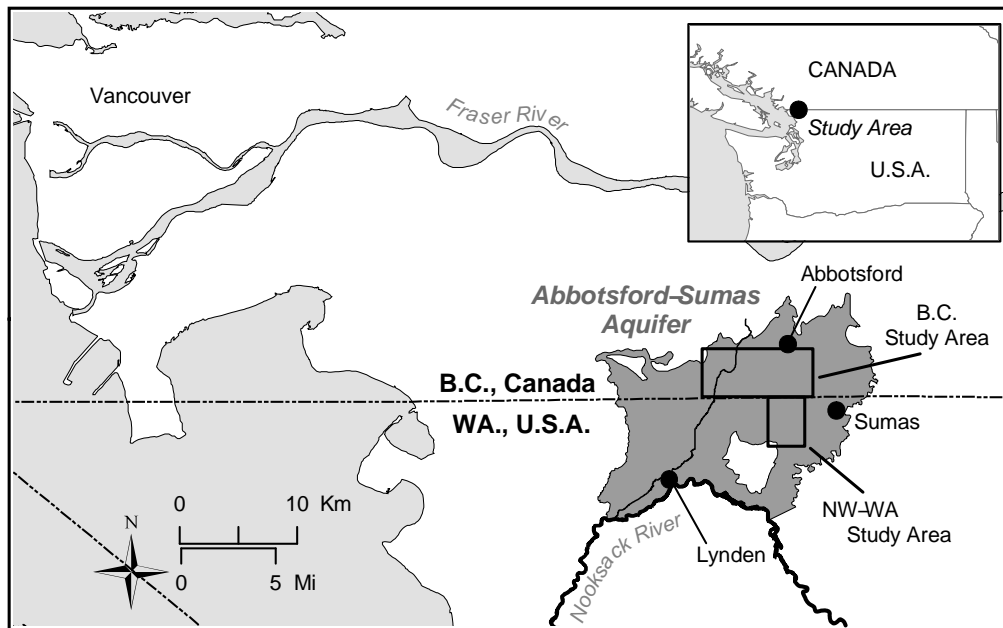


Figure 1: Location of the Abbotsford-Sumas aquifer and the Western Washington University and Environment Canada study sites.

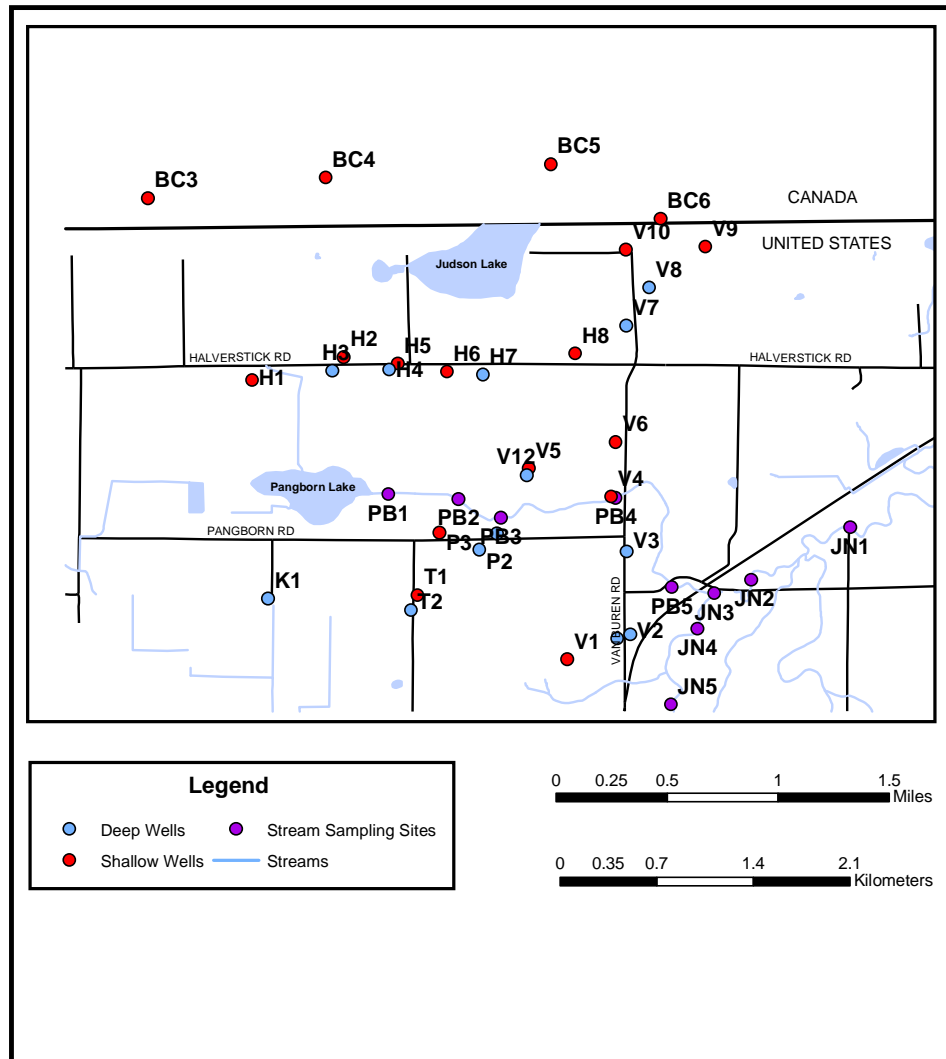


Figure 2: Groundwater and stream sampling sites in the Western Washington University study area.

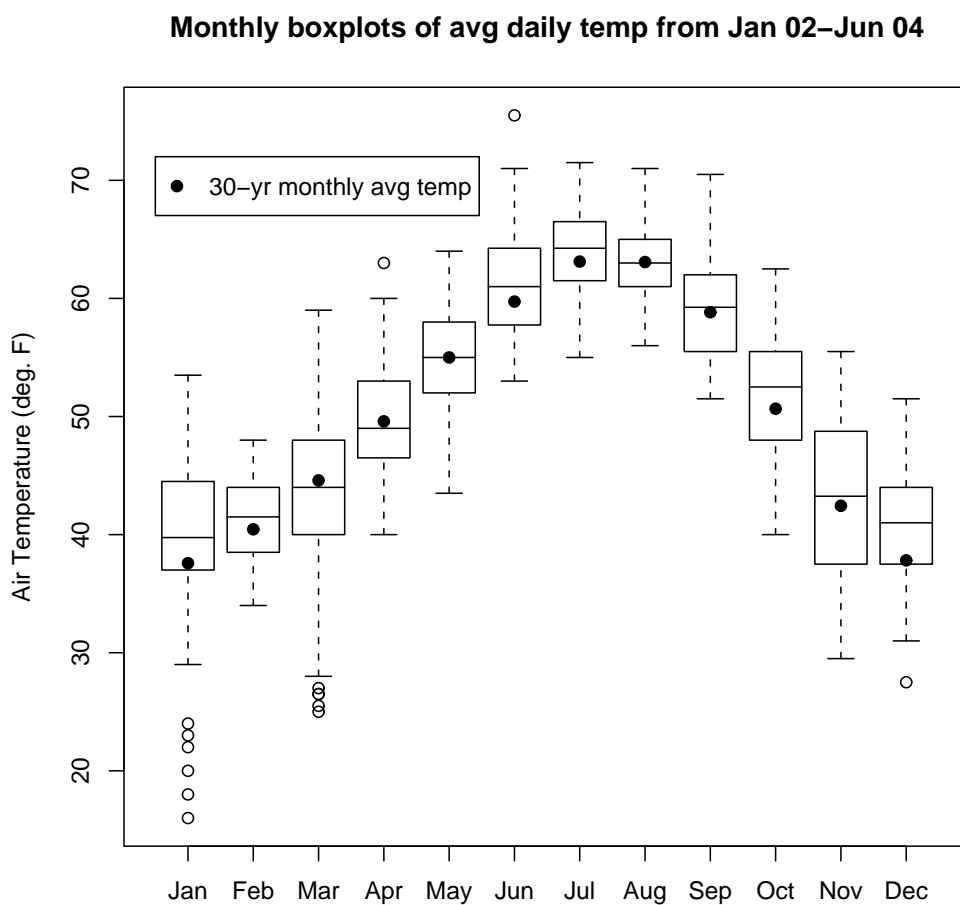


Figure 3: Boxplot of monthly temperature averages for January 2002 through June 2004 compared to the 30-year monthly averages (1973–2003; shown as filled circles). Data obtained from the Clearbrook, WA weather station (<http://www.ncdc.noaa.gov/ol/climate/climatedata.html>).

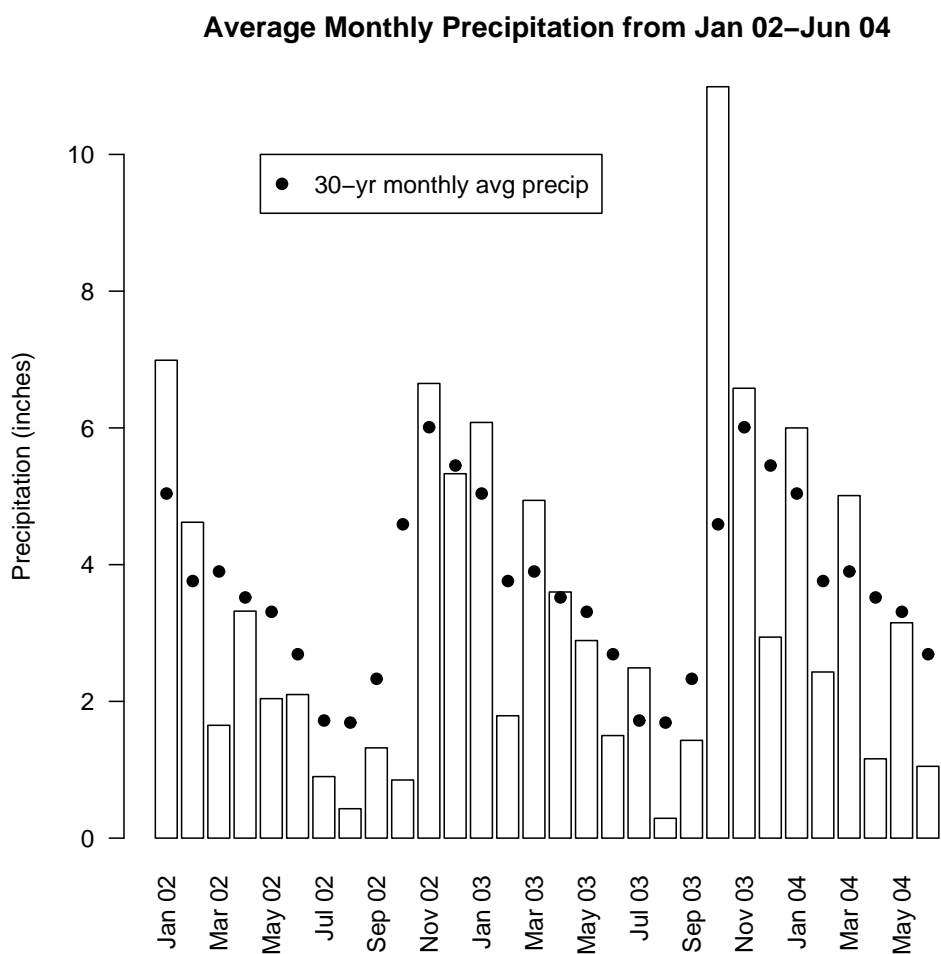


Figure 4: Monthly precipitation averages for January 2002 through June 2004 compared to the 30-year monthly averages (1973–2003; shown as points). Data obtained from the Clearbrook, WA weather station.

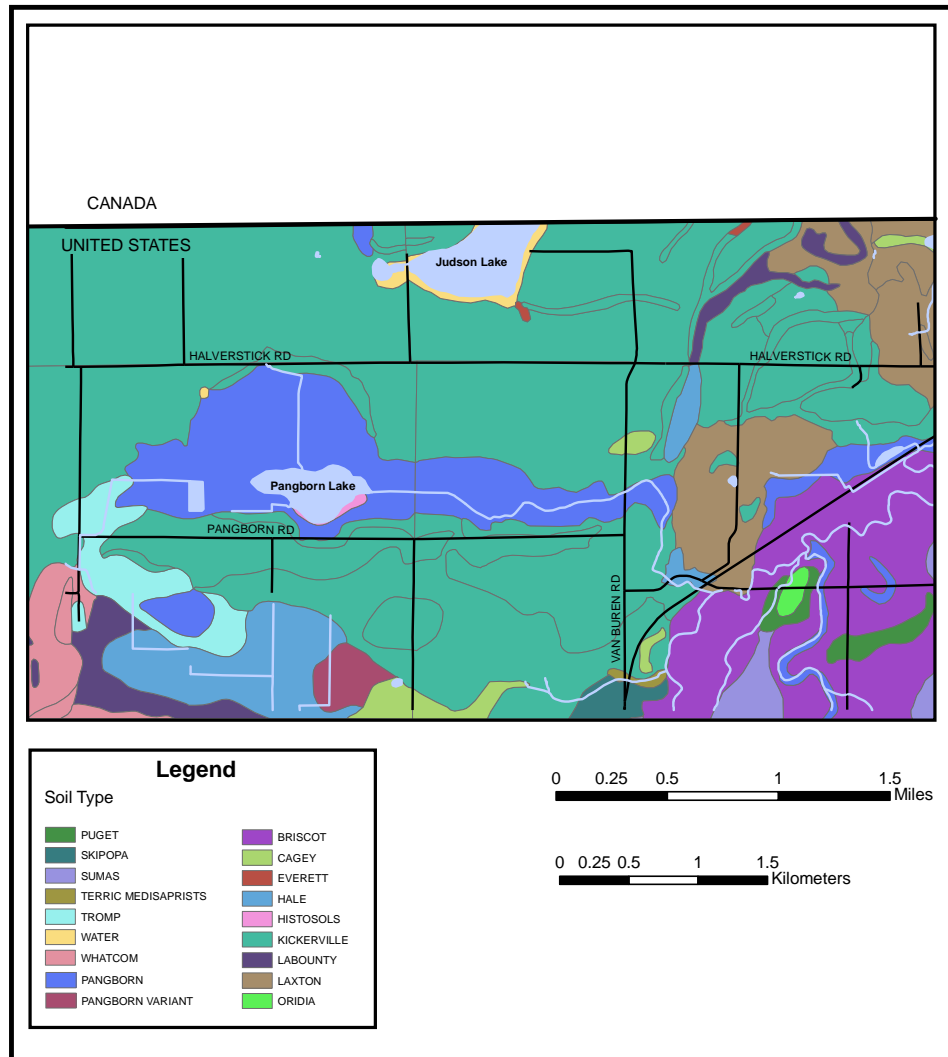


Figure 5: Soils and surface water in the Western Washington University study site.

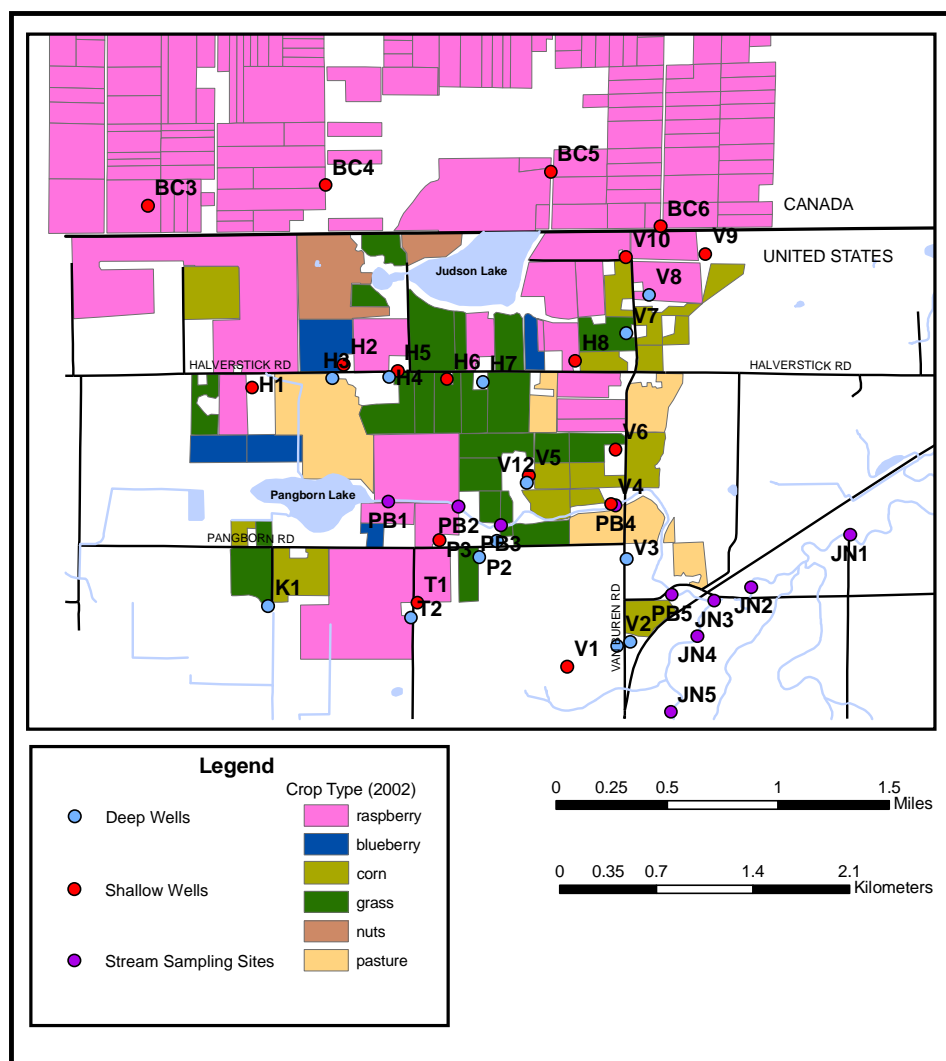


Figure 6: Land use in the Western Washington University study site in 2002.

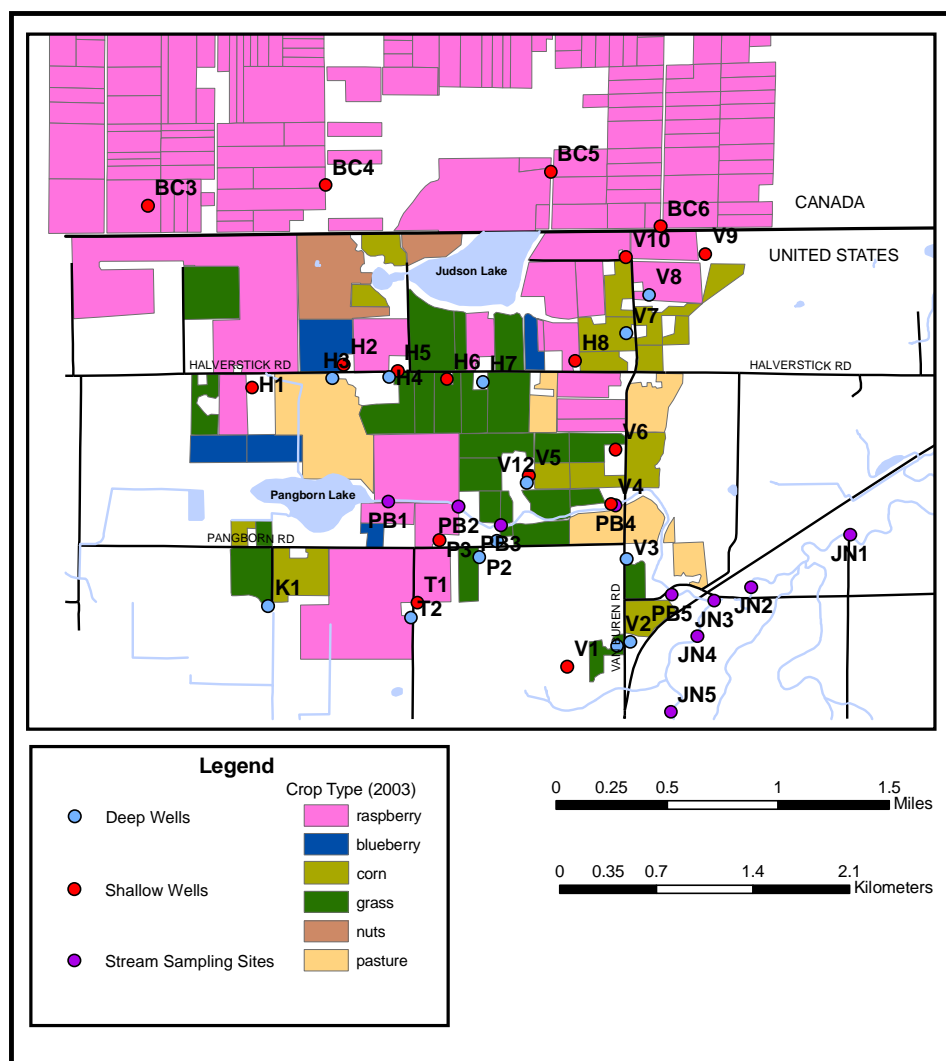


Figure 7: Land use in the Western Washington University study site in 2003.

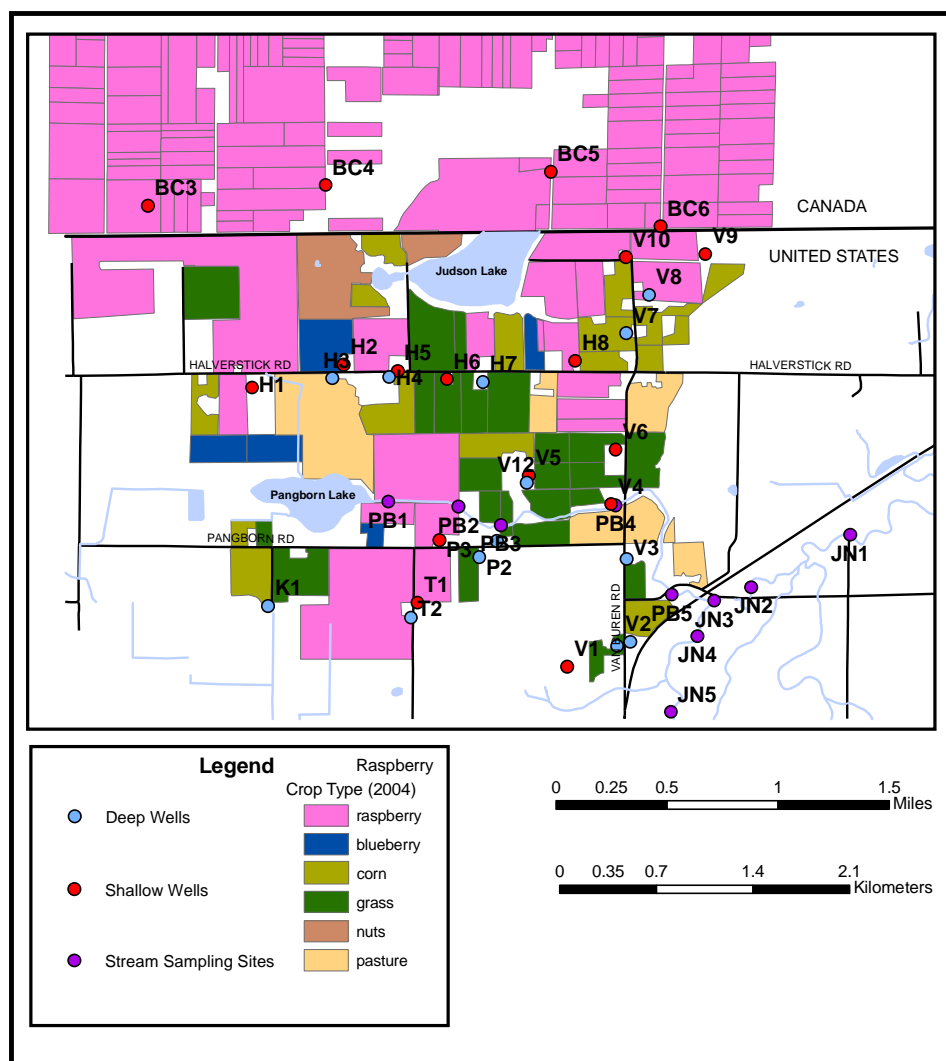


Figure 8: Land use in the Western Washington University study site in 2004.

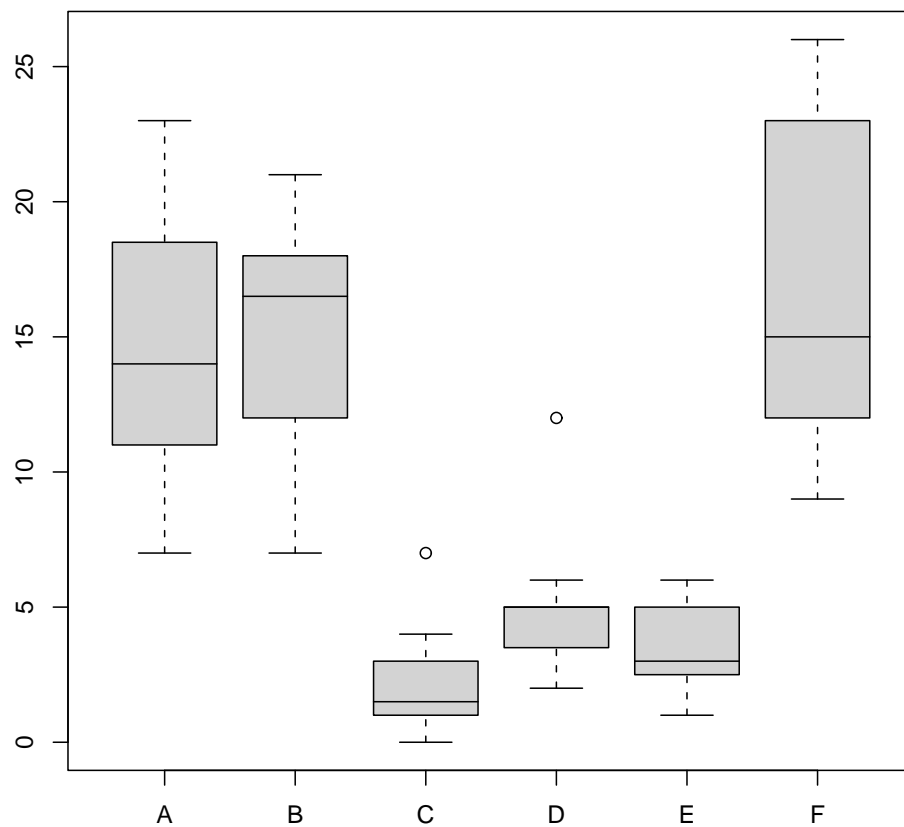


Figure 9: An example boxplot. Boxes indicate the median and upper/lower 25% quartiles; whiskers show upper/lower 50% quartiles; outliers (shown as open circles) are $\geq 1.5 \times$ interquartile range.

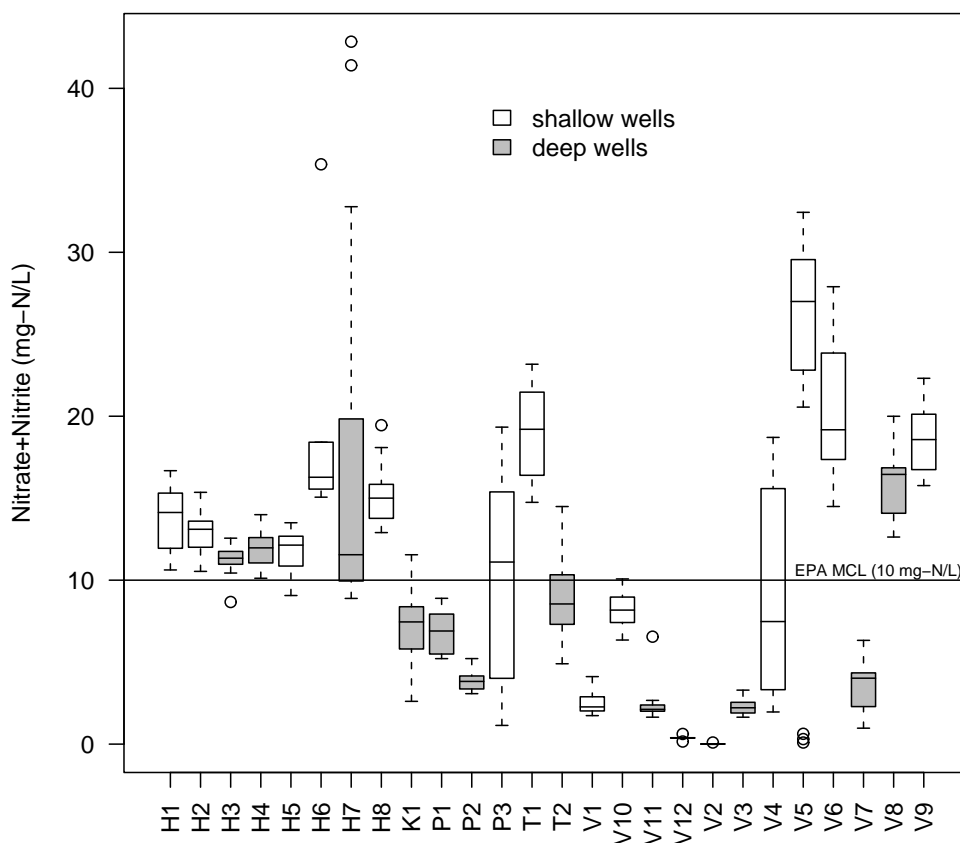


Figure 10: Boxplot of nitrate+nitrite concentrations for the well sites. The EPA's Maximum Contaminant Limit (MCL) of 10 mg-N/L for nitrate in drinking water is shown as a line. Shading indicates a shallow vs. deep well classification (see text for classification criteria).

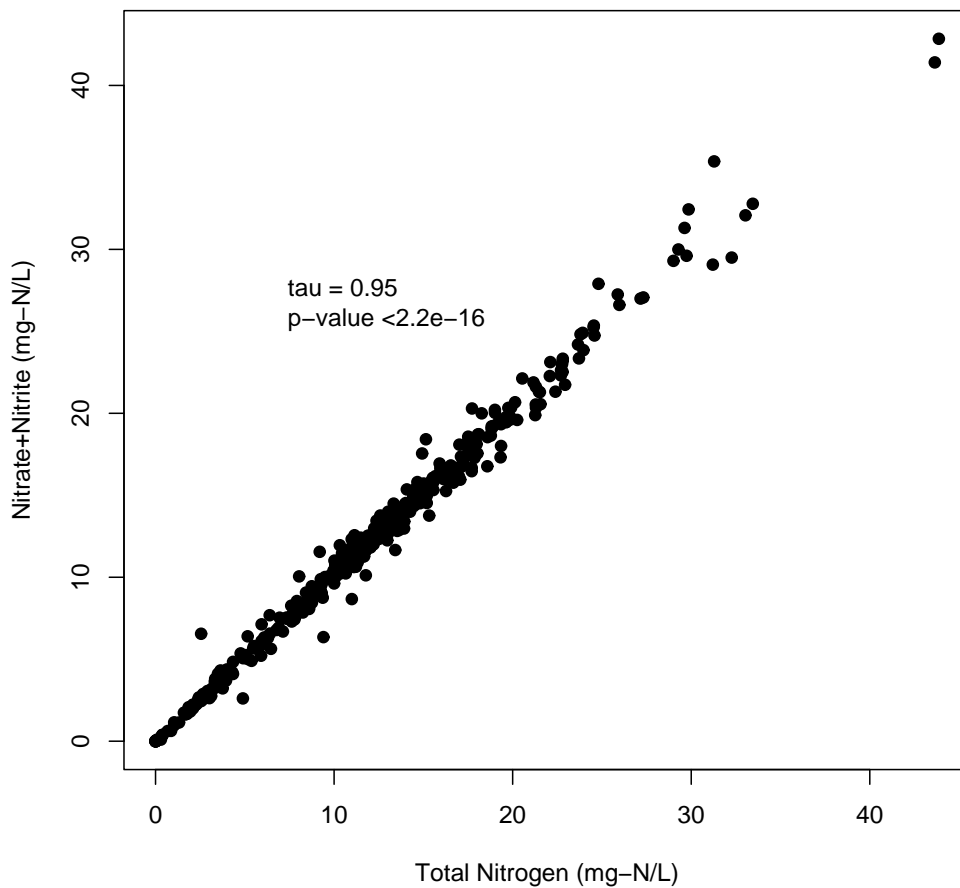


Figure 11: Scatterplot of nitrate+nitrite vs. total nitrogen concentrations for the well sites. Kendall's tau correlation results (shown here) indicate a significant correlation.

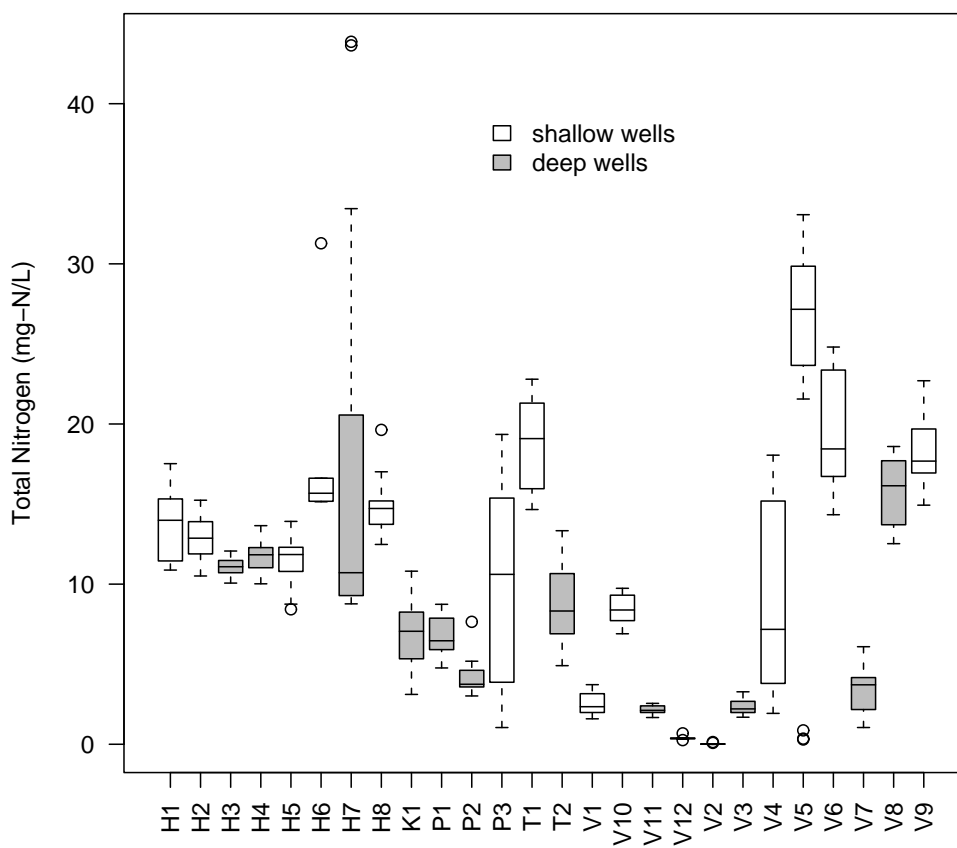


Figure 12: Boxplot of total nitrogen concentrations for the well sites. Shading indicates a shallow vs. deep well classification (see text for classification criteria).

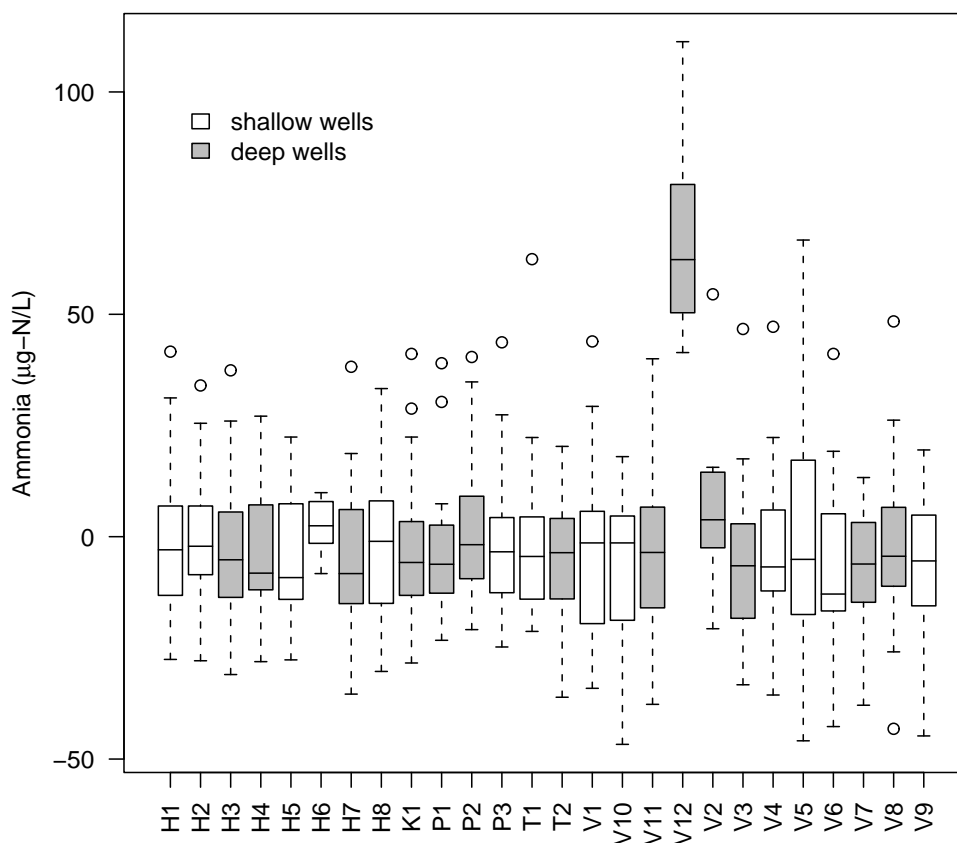


Figure 13: Boxplot of ammonia concentrations for the well sites. Note that the ammonia values are in $\mu\text{g-N/L}$, which differs from the mg-N/L scale used in plots of the other nutrient variables. Shading indicates a shallow vs. deep well classification (see text for classification criteria).

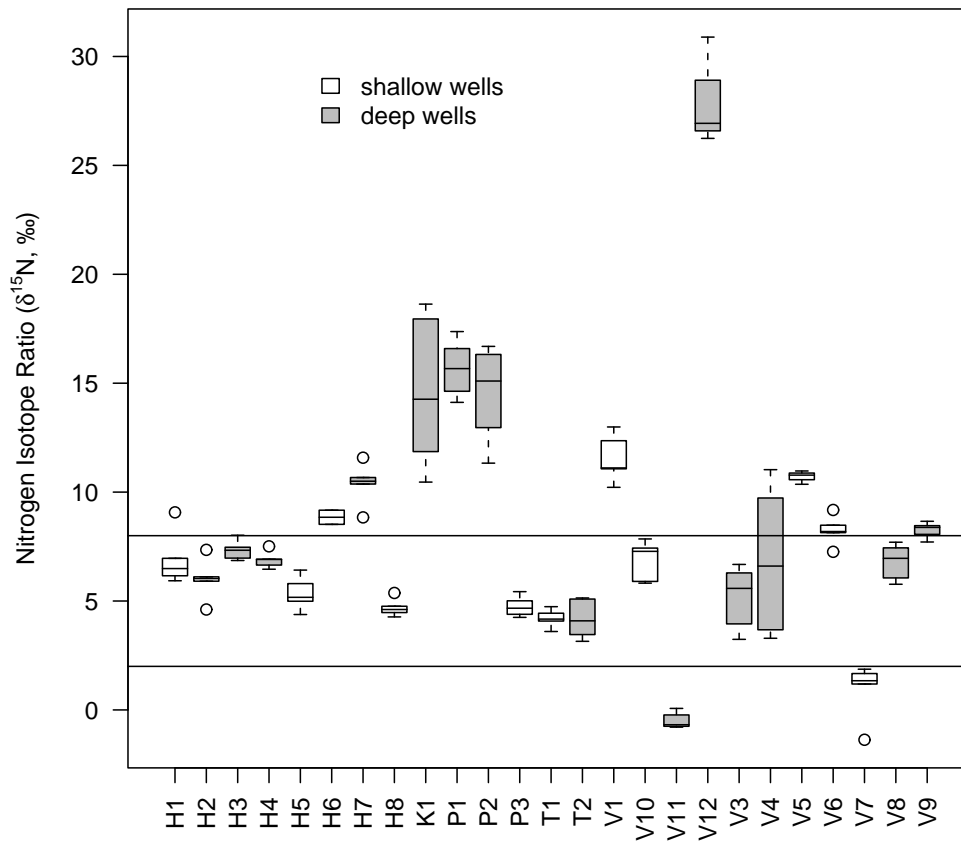


Figure 14: Boxplot of nitrogen isotope ratios ($\delta^{15}\text{N}$, ‰) for the well sites. A guideline for nitrate source identification presented by Wassenaar (1995) states that $\delta^{15}\text{N}$ values from +8 ‰ to +16 ‰ indicate an animal waste source, values between +2 and +8 ‰ (the lines shown here) indicate a mixture of inorganic fertilizer and animal waste sources, and values between -1 and +2 ‰ indicate inorganic fertilizer sources. Shading indicates a shallow vs. deep well classification (see text for classification criteria).

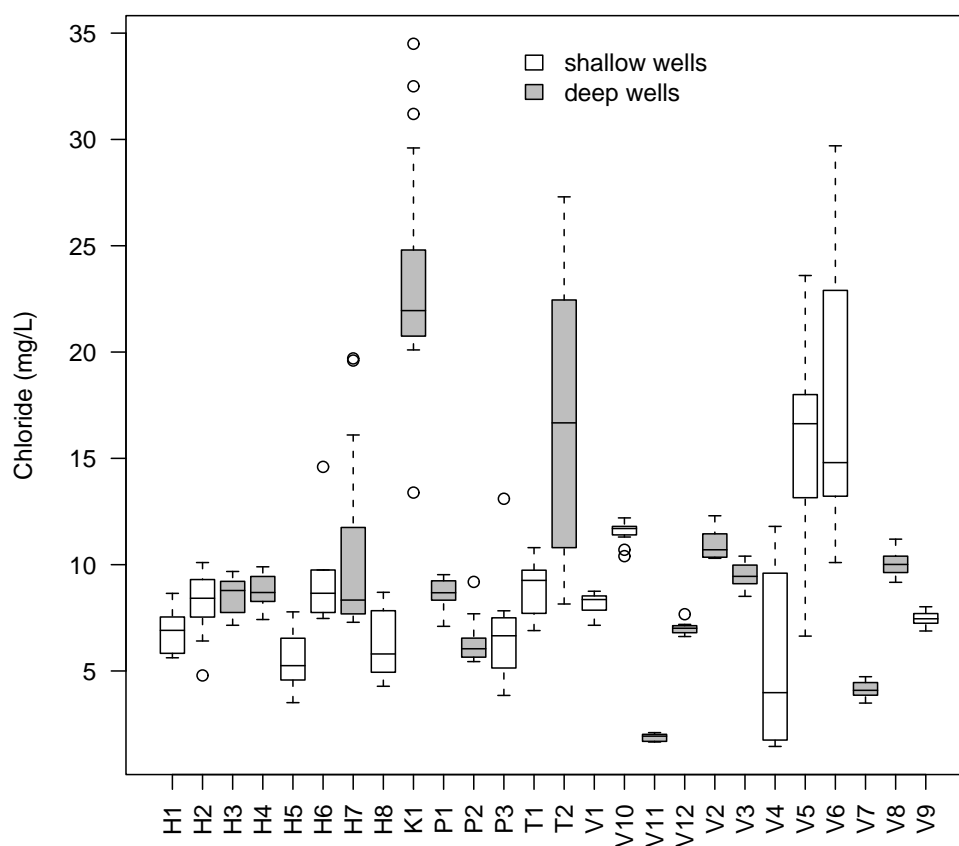


Figure 15: Boxplot of chloride concentrations for the well sites. Shading indicates a shallow vs. deep well classification (see text for classification criteria).

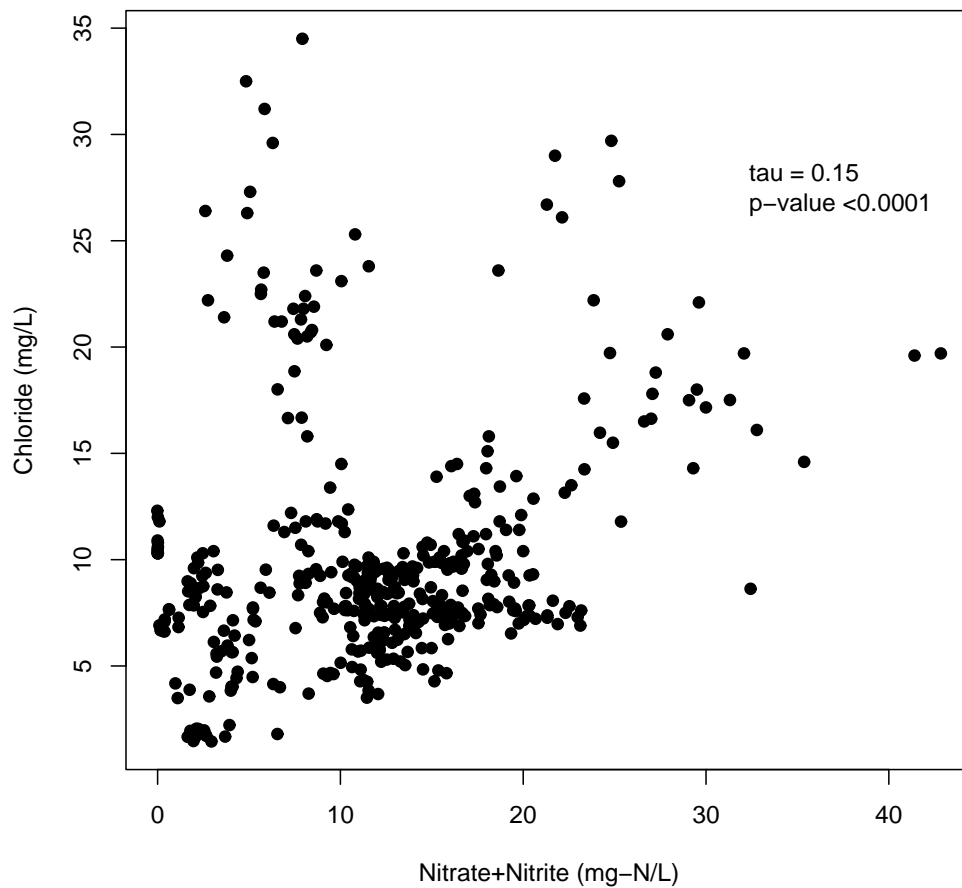


Figure 16: Scatterplot of nitrate+nitrite vs. chloride concentrations for the well sites. Kendall's tau correlation results (shown here) indicate a significant correlation.

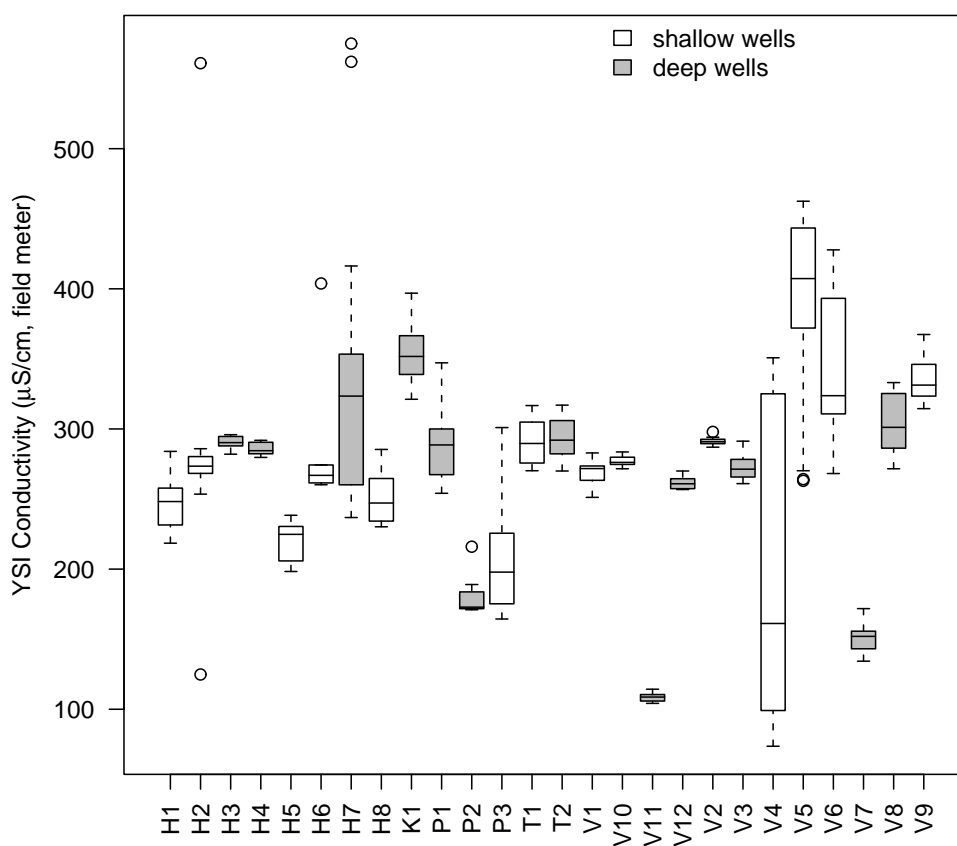


Figure 17: Boxplot of conductivity values for the well sites. Measurements were made in the field using a YSI model 85 analyzer and a flow-through cell. Shading indicates a shallow vs. deep well classification (see text for classification criteria).

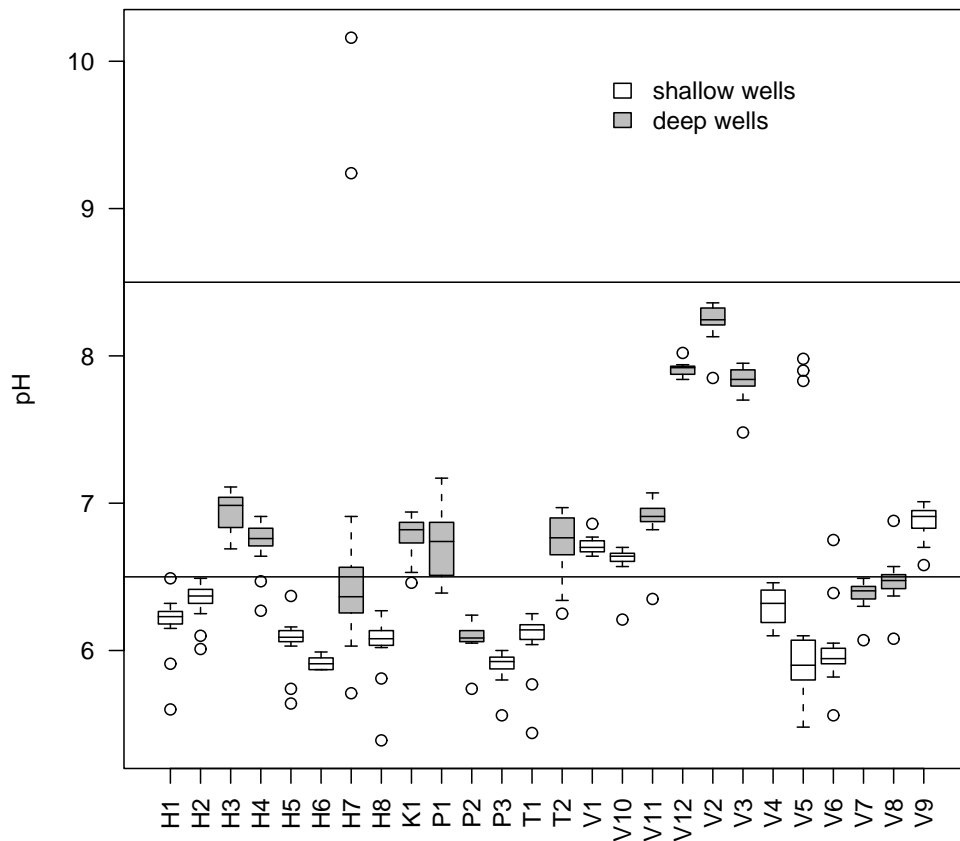


Figure 18: Boxplot of pH values for the well sites. The line represents the lower criterion for pH in ground water (6.5); the upper criterion (8.5) is off scale. Shading indicates a shallow vs. deep well classification (see text for classification criteria).

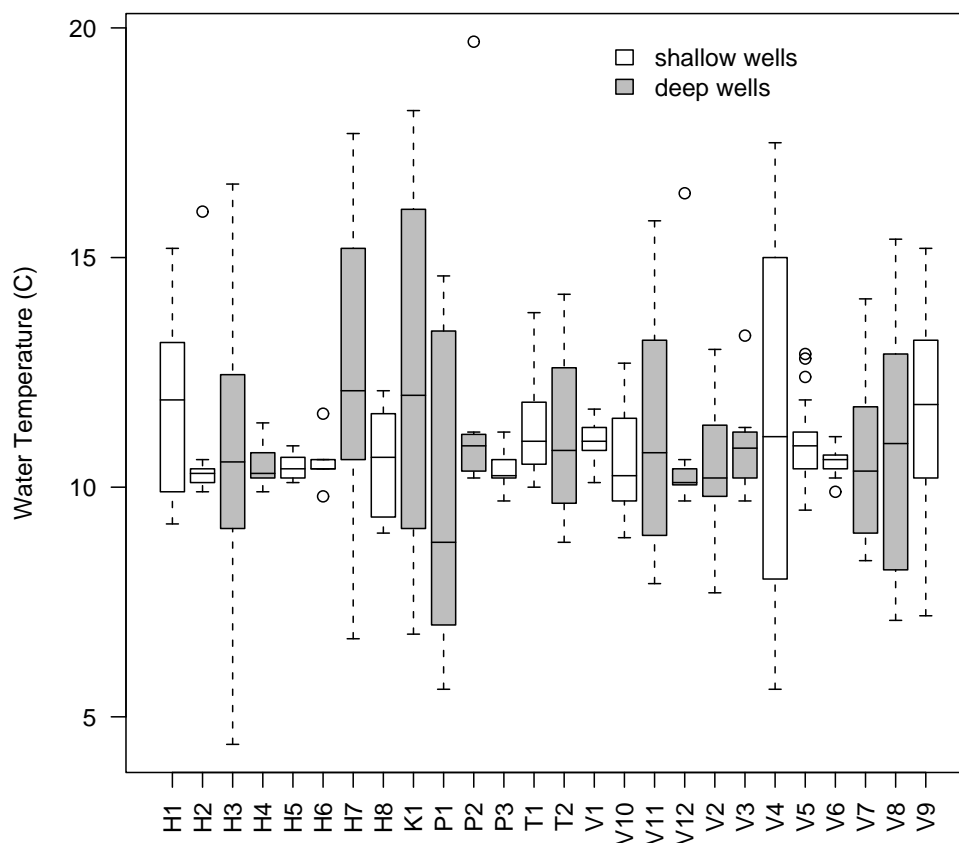


Figure 19: Boxplot of ground water temperature values for the well sites. Measurements were made in the field using a YSI model 85 analyzer and a flow-through cell. Shading indicates a shallow vs. deep well classification (see text for classification criteria).

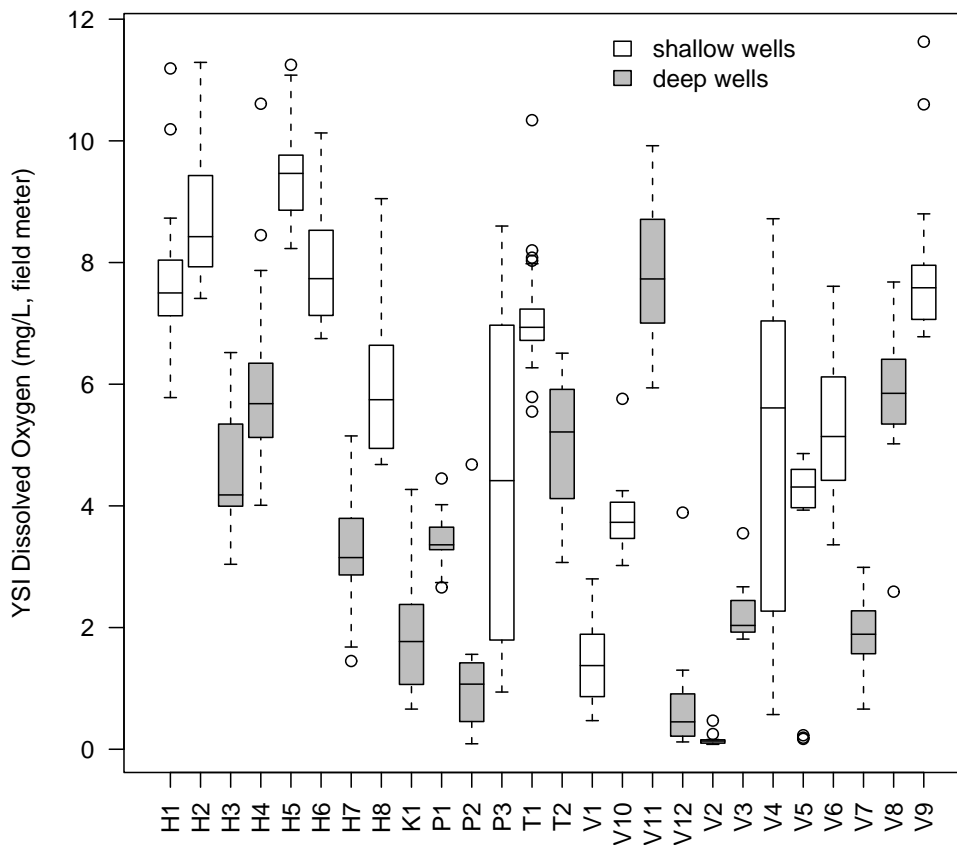


Figure 20: Boxplot of dissolved oxygen concentrations for the well sites. Measurements were made in the field using a YSI model 85 analyzer and a flow-through cell. Shading indicates a shallow vs. deep well classification (see text for classification criteria).

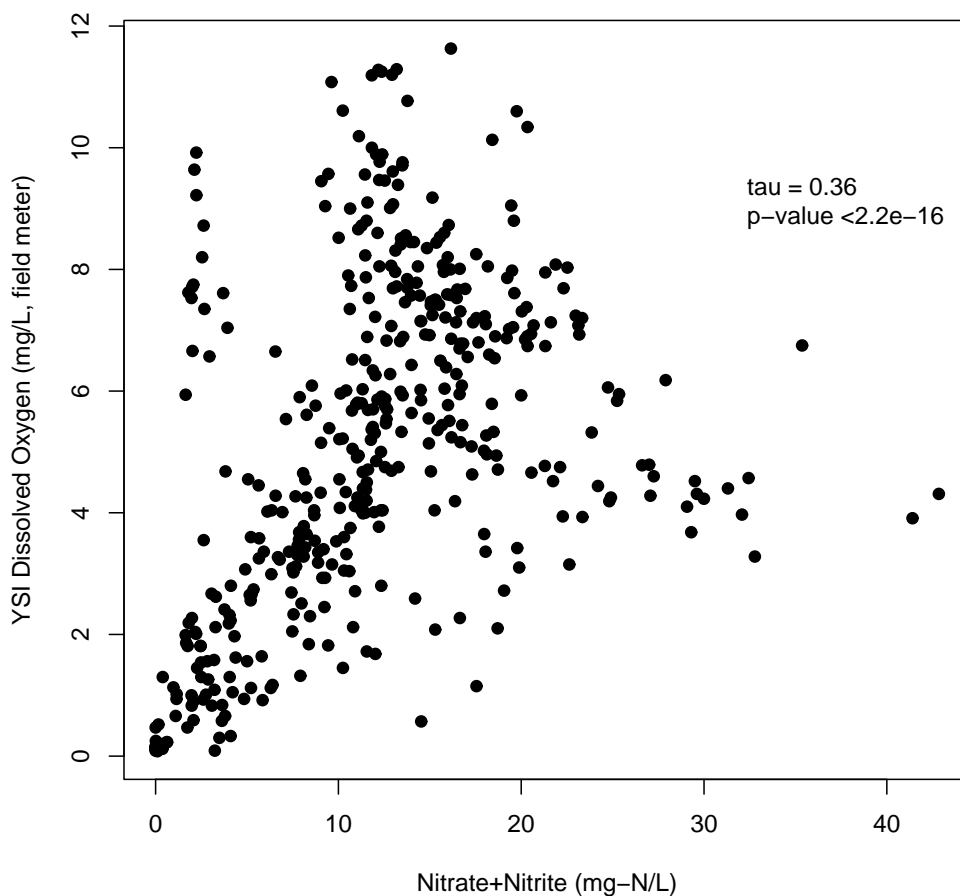


Figure 21: Scatterplot of nitrate+nitrite vs. dissolved oxygen concentrations for the well sites. Measurements were made in the field using a YSI model 85 analyzer. Kendall's tau correlation results (shown here) indicate a significant correlation.

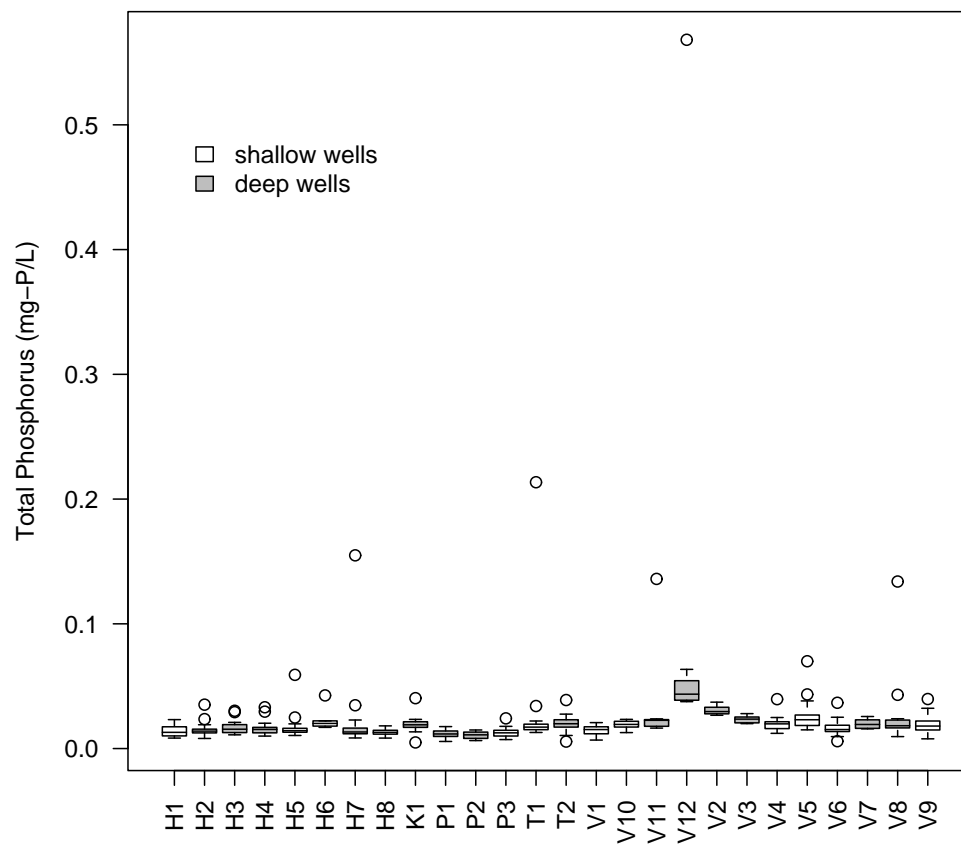


Figure 22: Boxplot of total phosphorus concentrations for the well sites. Shading indicates a shallow vs. deep well classification (see text for classification criteria).

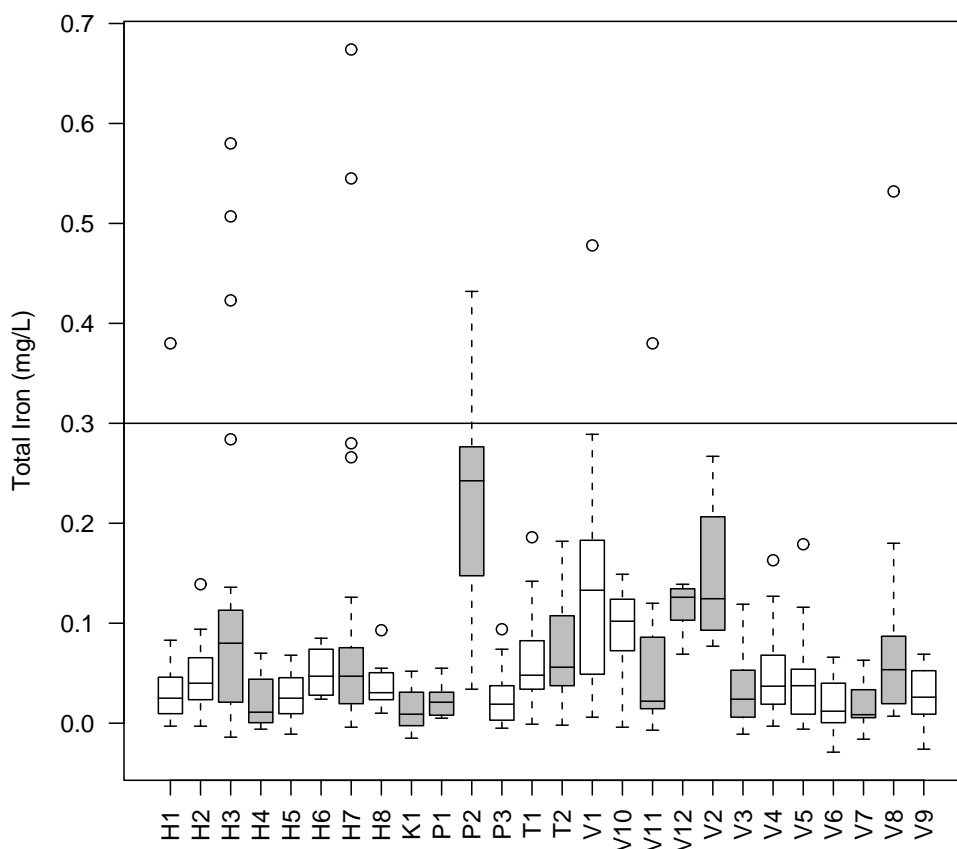


Figure 23: Boxplot of total iron concentrations for the well sites with one outlier from well V5 removed (outlier had a total iron concentration of 2.37 mg/L measured on March 18, 2004). The line indicates the ground water quality criterion for total iron of 0.3 mg/L. Shading indicates a shallow vs. deep well classification (see text for classification criteria).

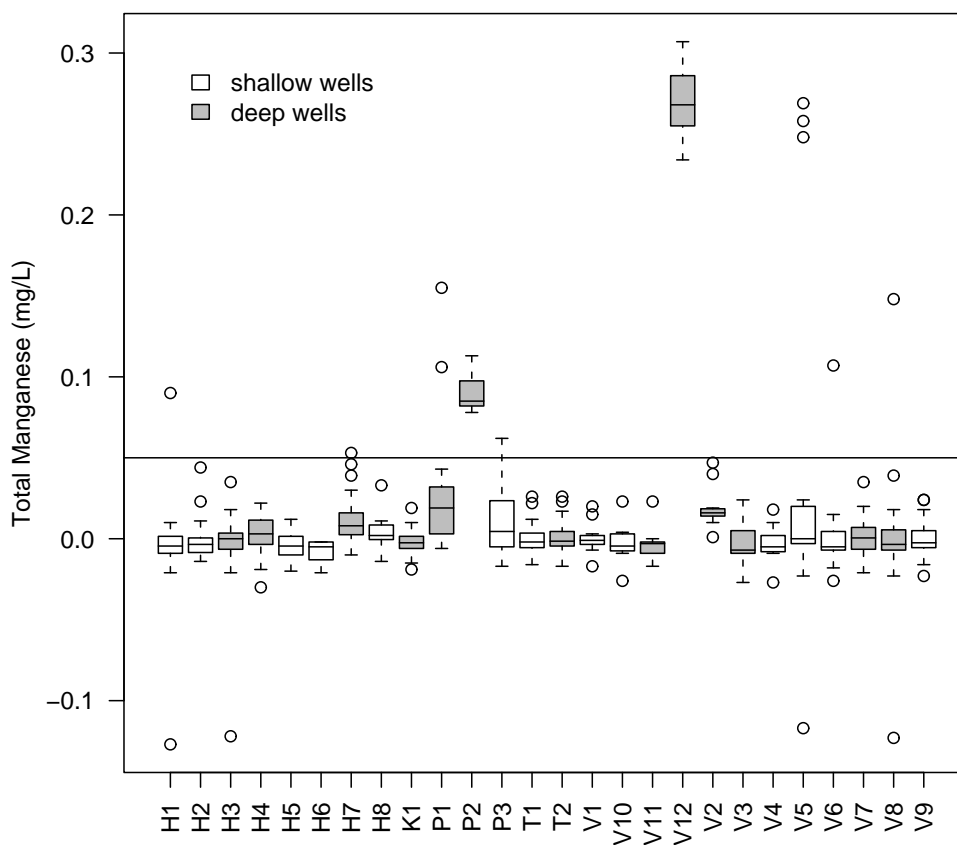


Figure 24: Boxplot of total manganese concentrations for the well sites. The line indicates the ground water quality criterion for total manganese of 0.05 mg/L. Shading indicates a shallow vs. deep well classification (see text for classification criteria).

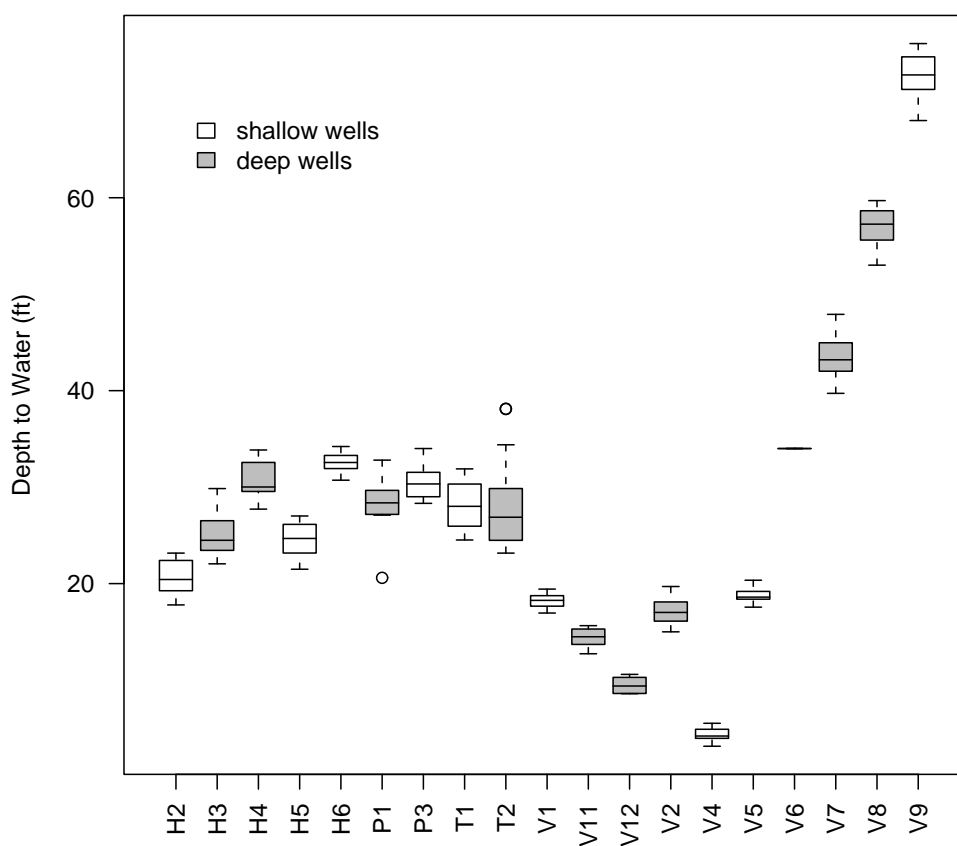


Figure 25: Boxplot of depth to water for some well sites. Note that the water level was not measured at all well sites due to some access restrictions. Shading indicates a shallow vs. deep well classification (see text for classification criteria).

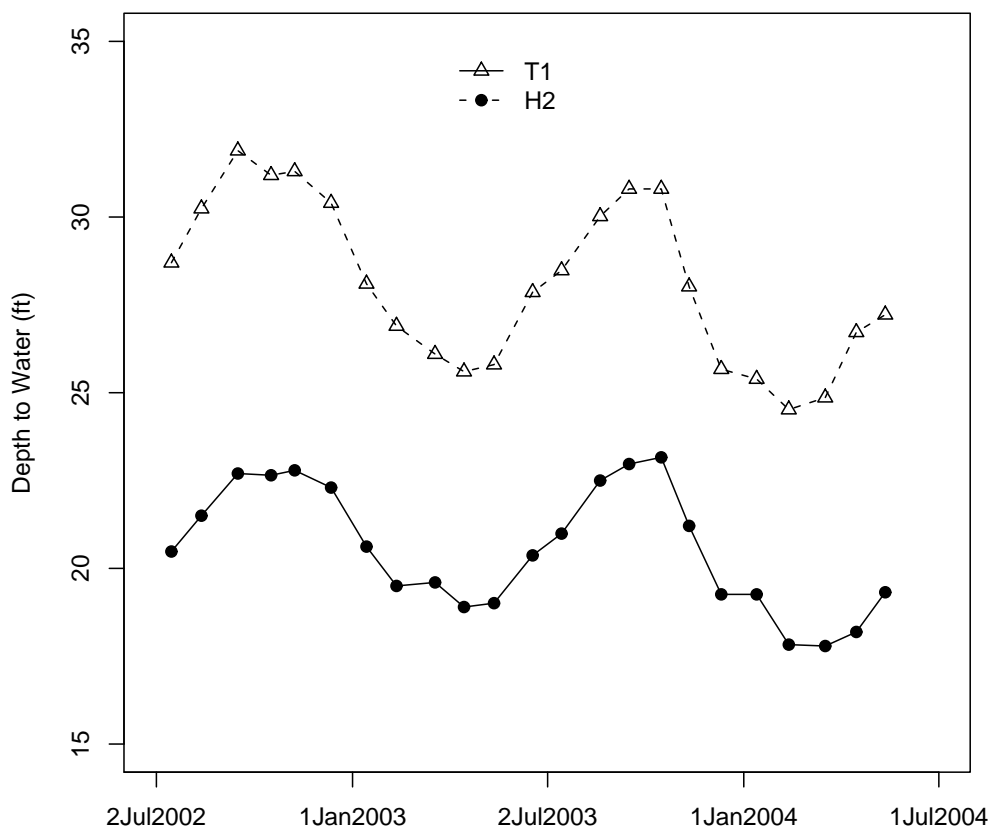


Figure 26: Scatterplot of depth to water vs. date for well sites T1 and H2.

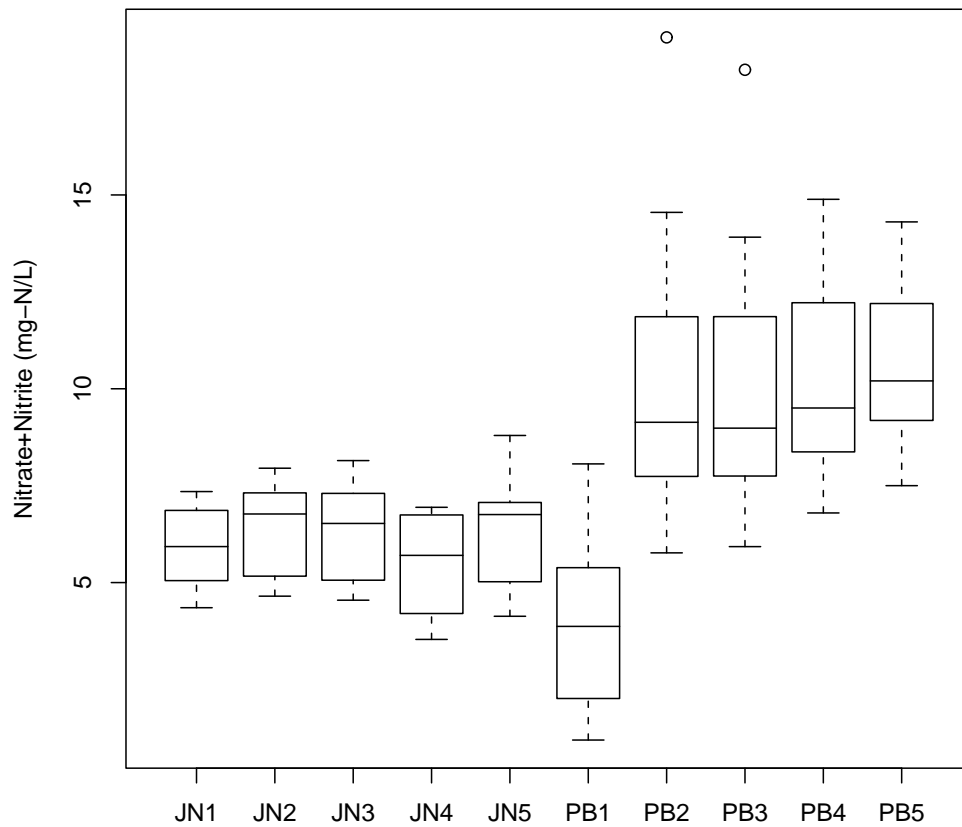


Figure 27: Boxplot of nitrate+nitrite concentrations for the stream sites.

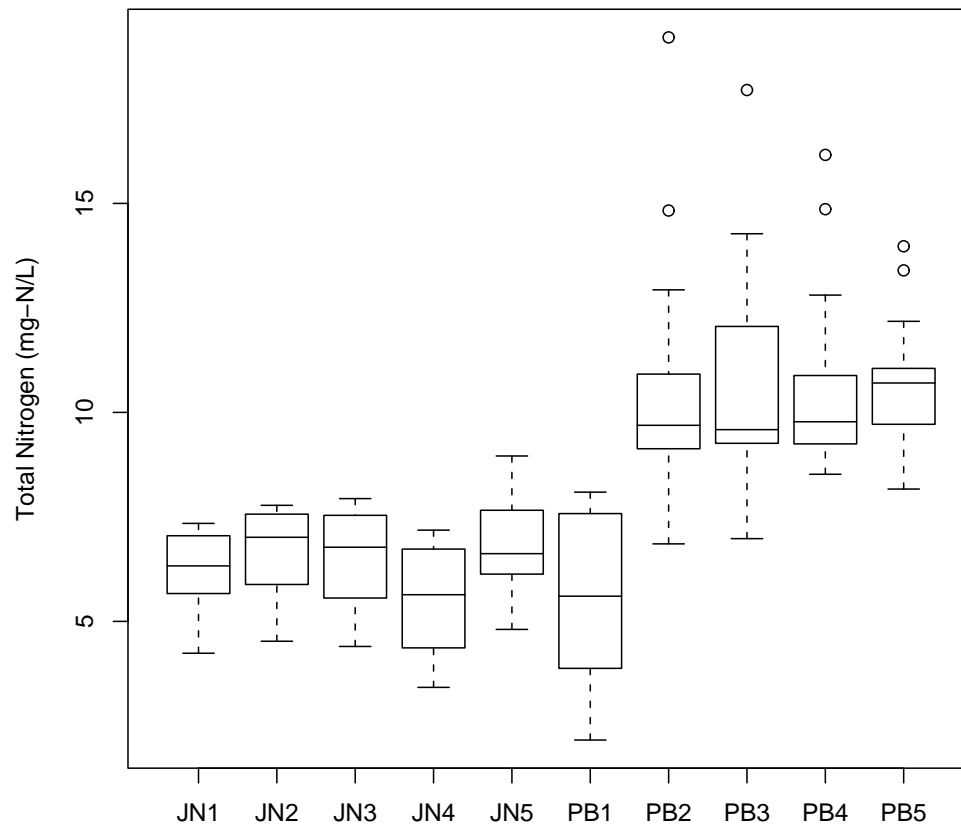


Figure 28: Boxplot of total nitrogen concentrations for the stream sites.

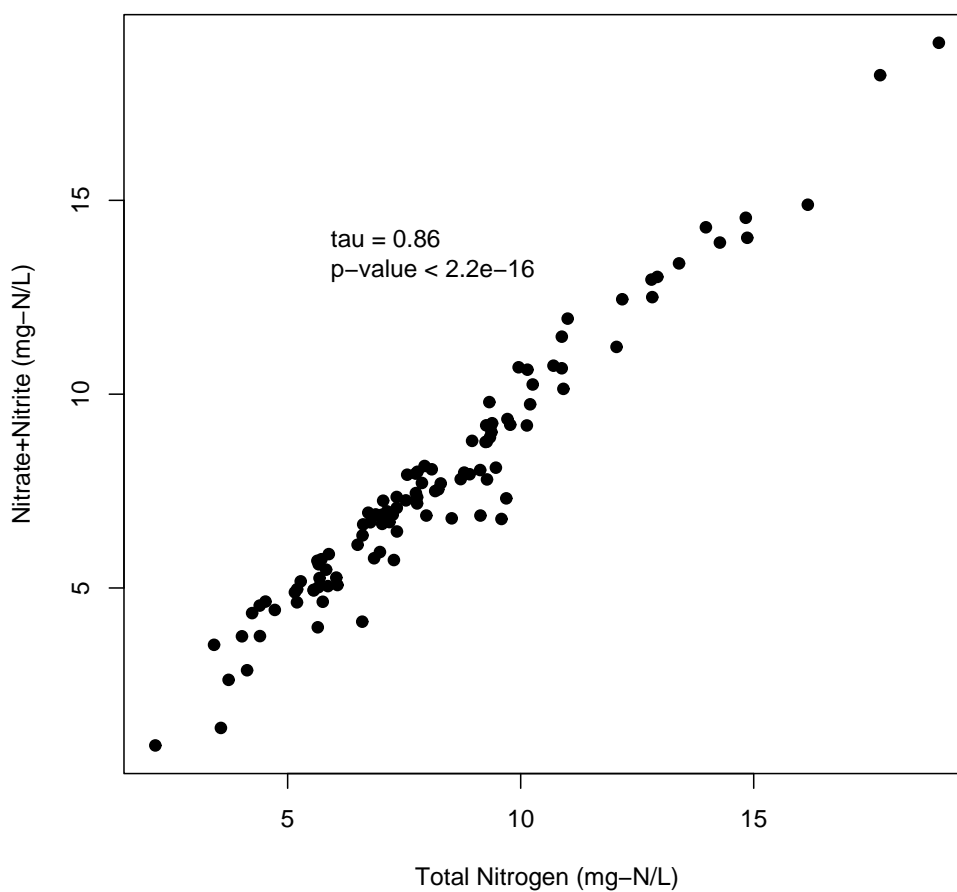


Figure 29: Scatterplot of nitrate+nitrite vs. total nitrogen concentrations for the stream sites. Kendall's tau correlation results (shown here) indicate a significant correlation.

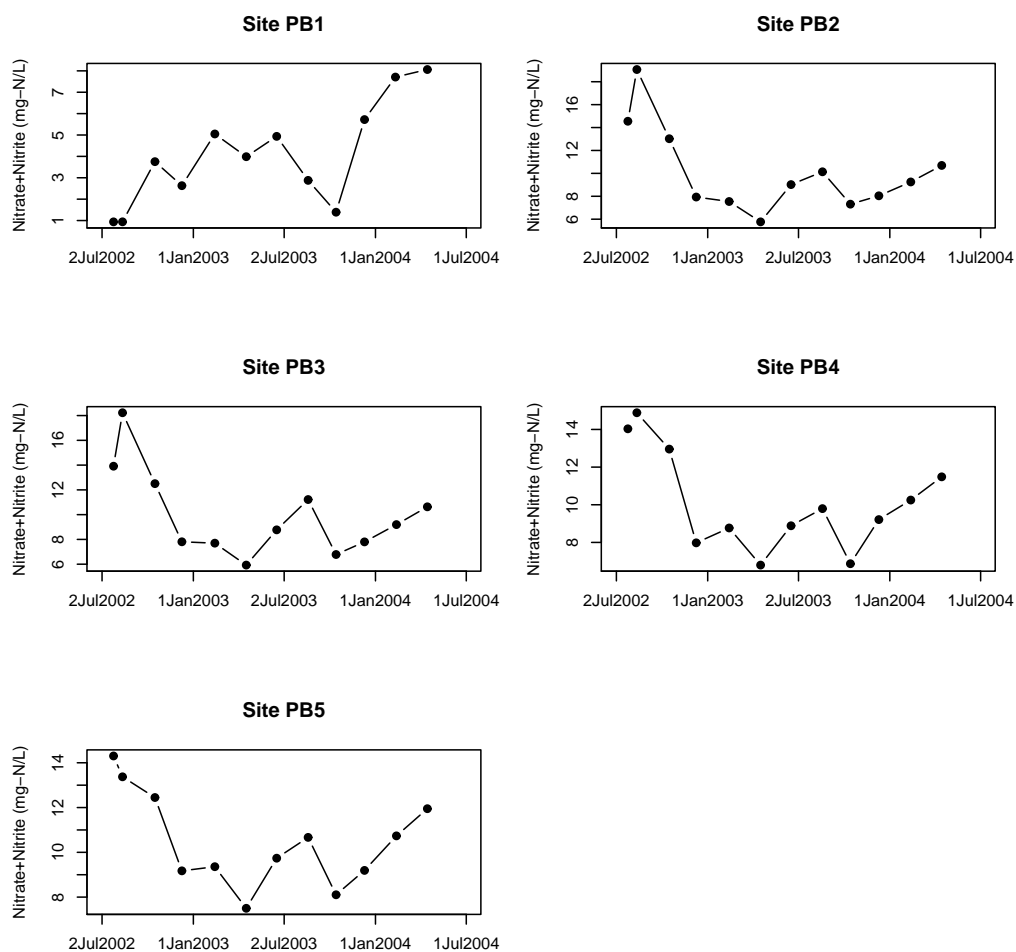


Figure 30: Multiple scatterplots of nitrate+nitrite concentration vs. date for the Pangborn Creek stream sites. Note that each plot has a different y-axis scale, but the same date range.

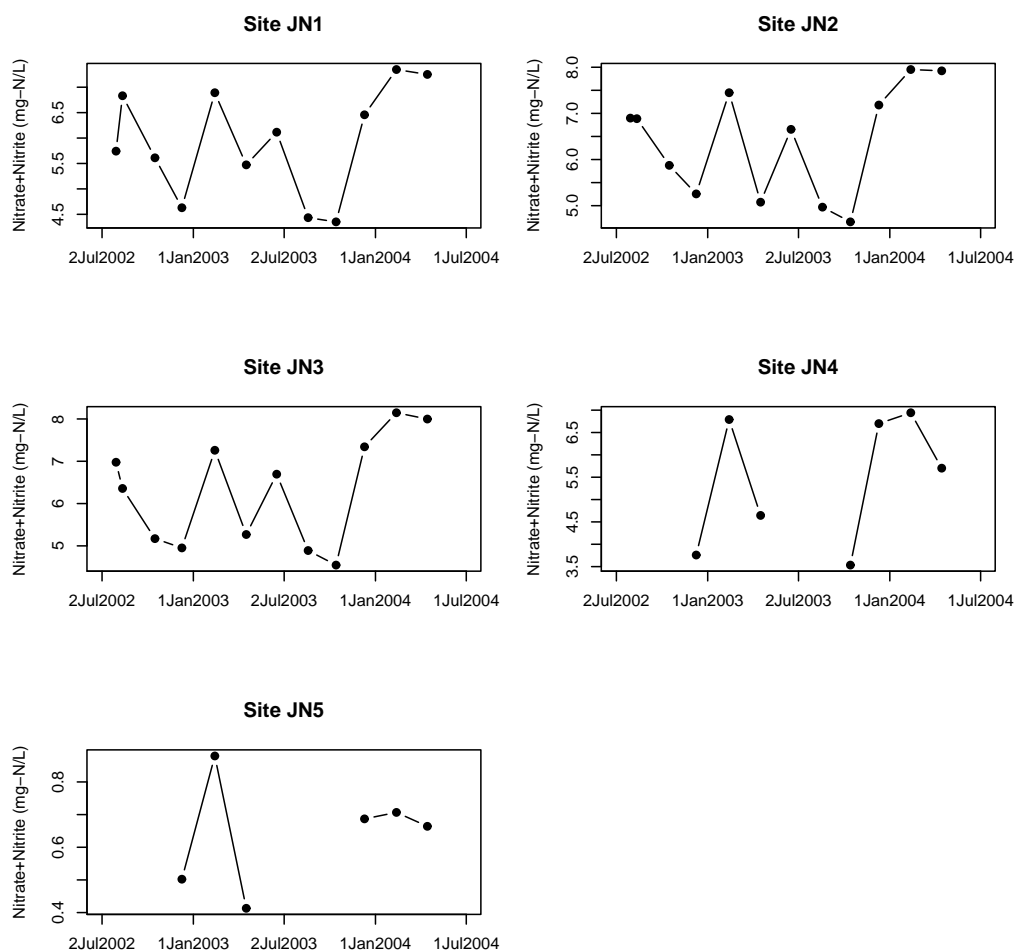


Figure 31: Multiple scatterplots of nitrate+nitrite concentration vs. date for the Johnson Creek stream sites. Note that each plot has a different y-axis scale, but the same date range.

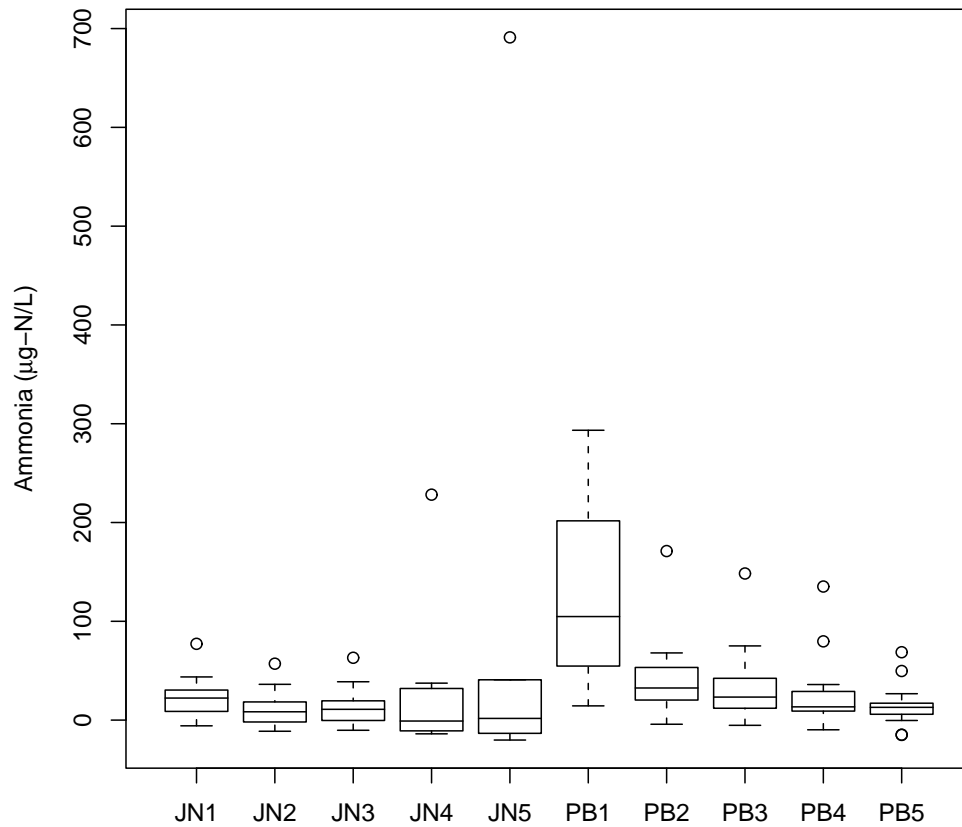


Figure 32: Boxplot of ammonia concentrations for the stream sites.

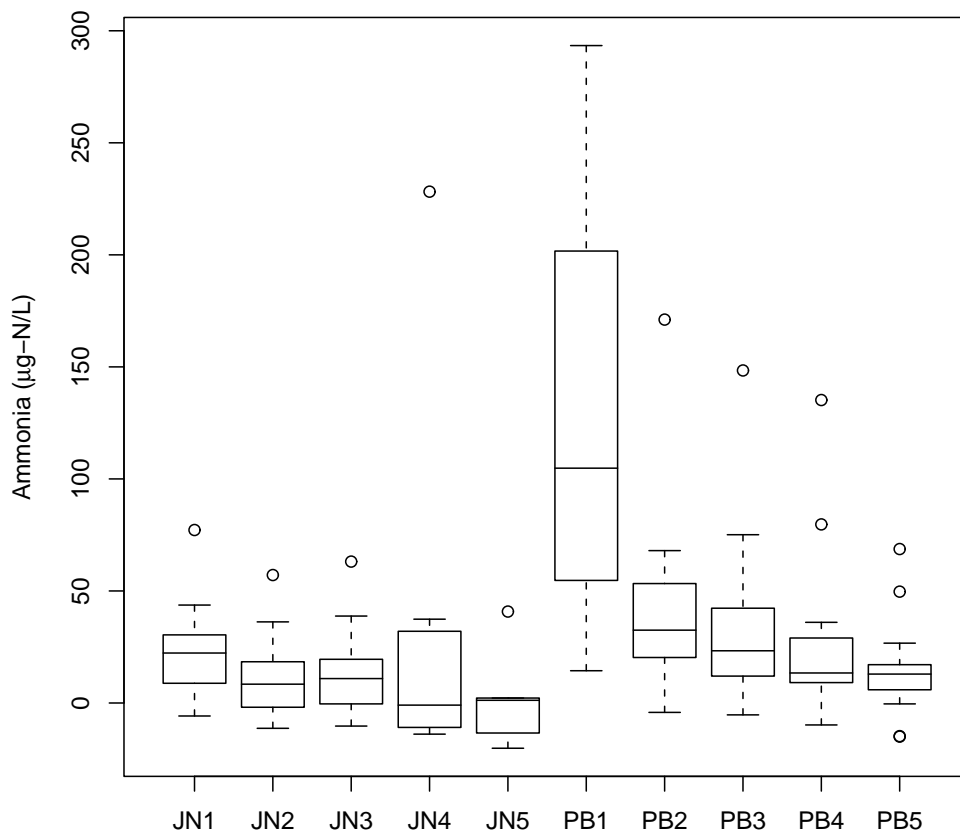


Figure 33: Boxplot of ammonia concentrations for the stream sites with one outlier for site JN5 removed (outlier had an ammonia concentration of $691.14 \mu\text{g-N/L}$ measured on April 17, 2003).

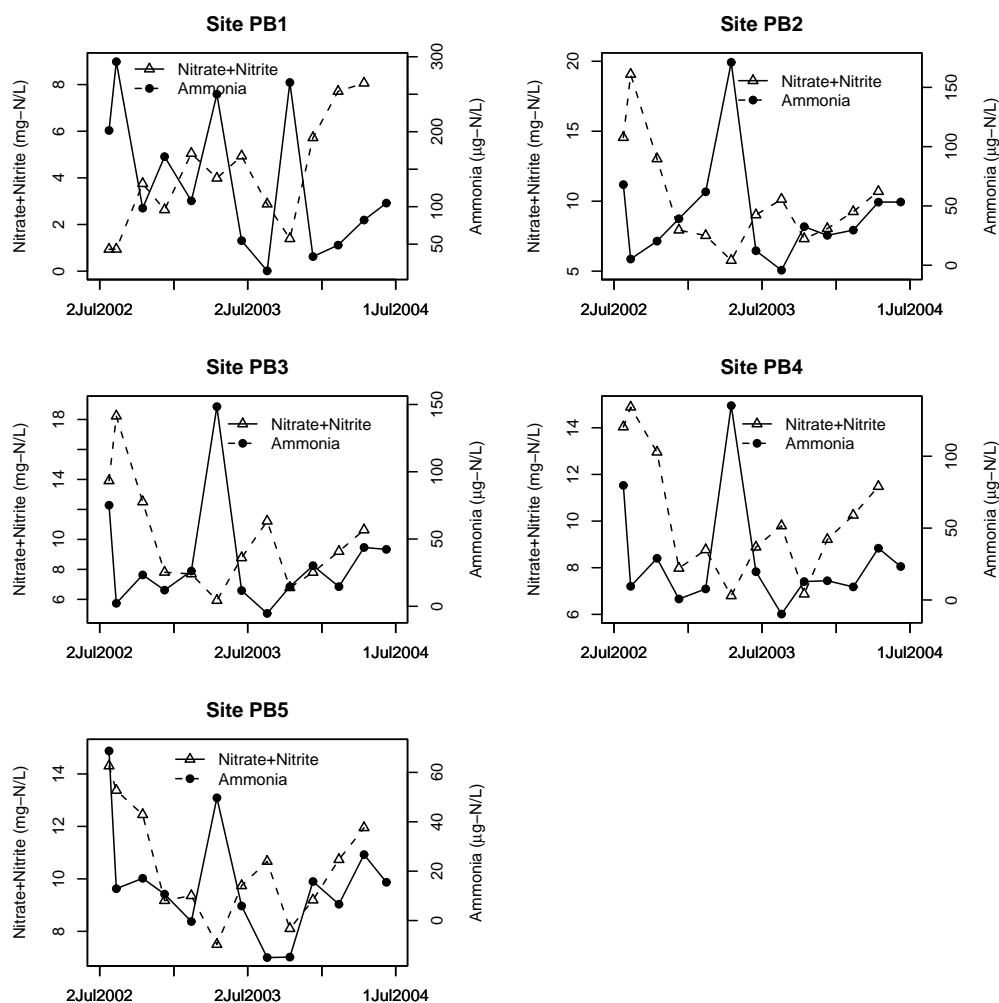


Figure 34: Multiple scatterplots of nitrate+nitrite and ammonia concentrations vs. date for the Pangborn Creek stream sites. Note that the scales and units for nitrate+nitrite and ammonia differ within each plot and the y-axis scales for both variables differ from plot to plot, but the date range remains constant.

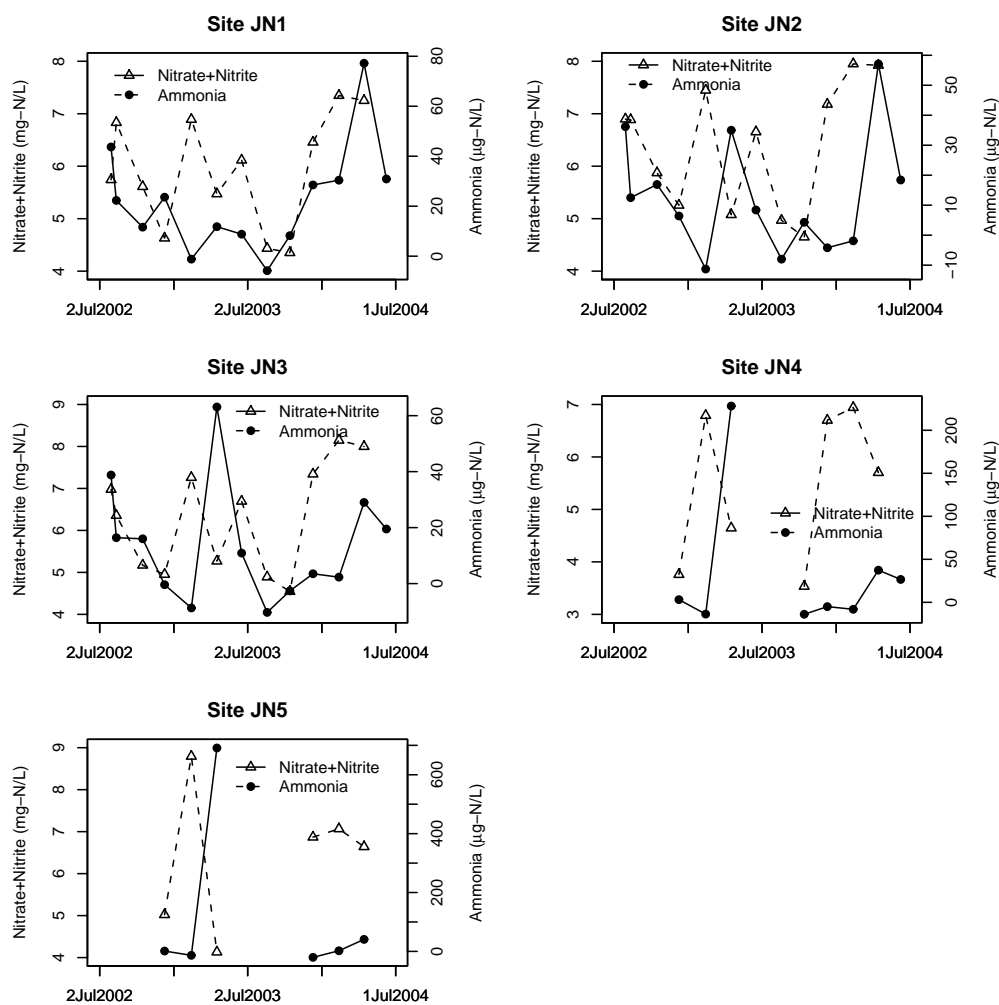


Figure 35: Multiple scatterplots of nitrate+nitrite and ammonia concentrations vs. date for the Johnson Creek stream sites. Note that the scales and units for nitrate+nitrite and ammonia differ within each plot and the y-axis scales for both variables differ from plot to plot, but the date range remains constant.

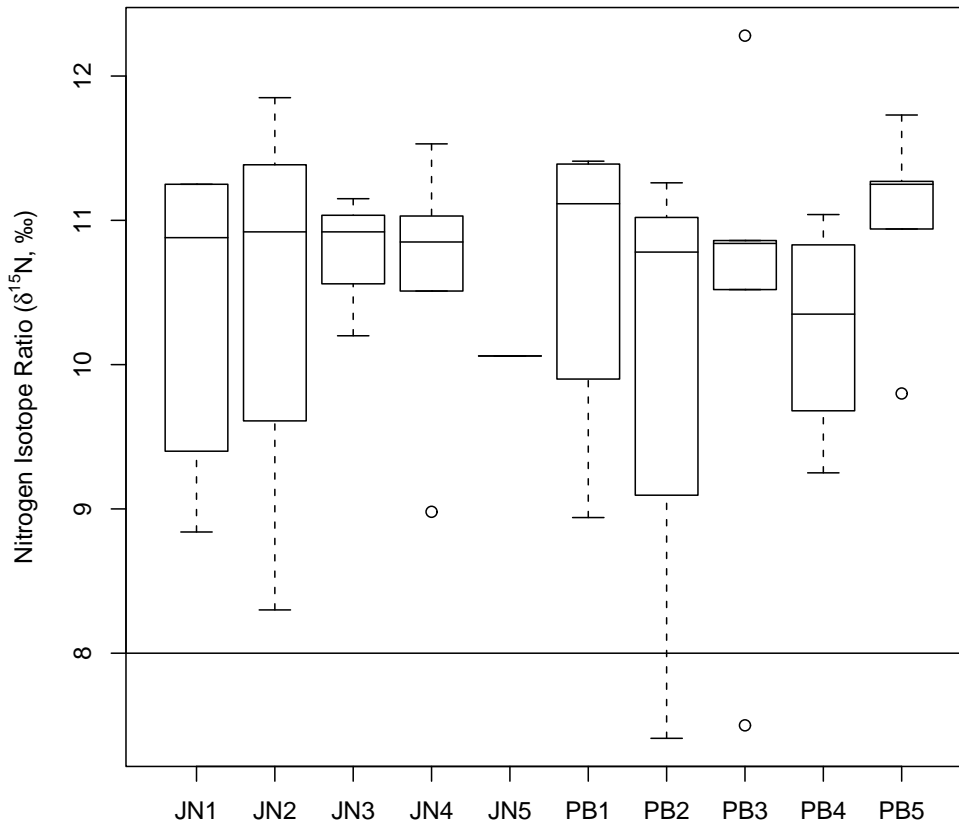


Figure 36: Boxplot of nitrogen isotope ratios ($\delta^{15}\text{N}$, ‰) for the stream sites. A guideline for nitrate source identification presented by Wassenaar (1995) states that $\delta^{15}\text{N}$ values from +8 ‰ (the line shown here) to +16 ‰ indicate an animal waste source, values between +2 and +8 ‰ indicate a mixture of inorganic fertilizer and animal waste sources, and values between -1 and +2 ‰ indicate inorganic fertilizer sources.

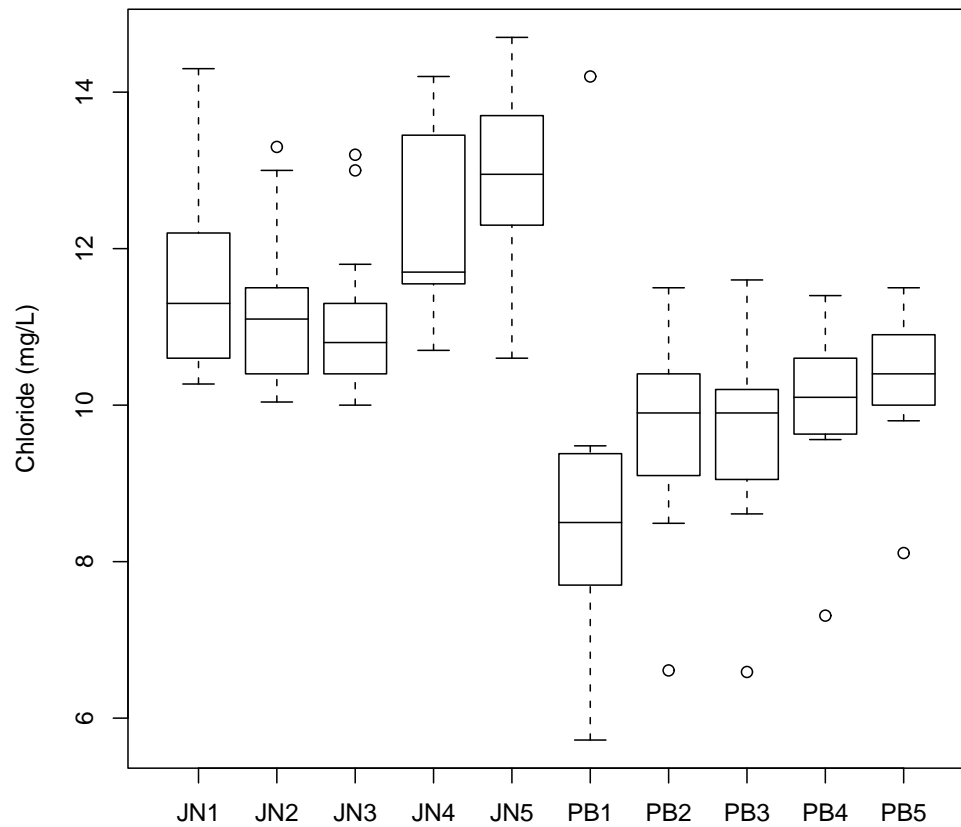


Figure 37: Boxplot of chloride concentrations for the stream sites.

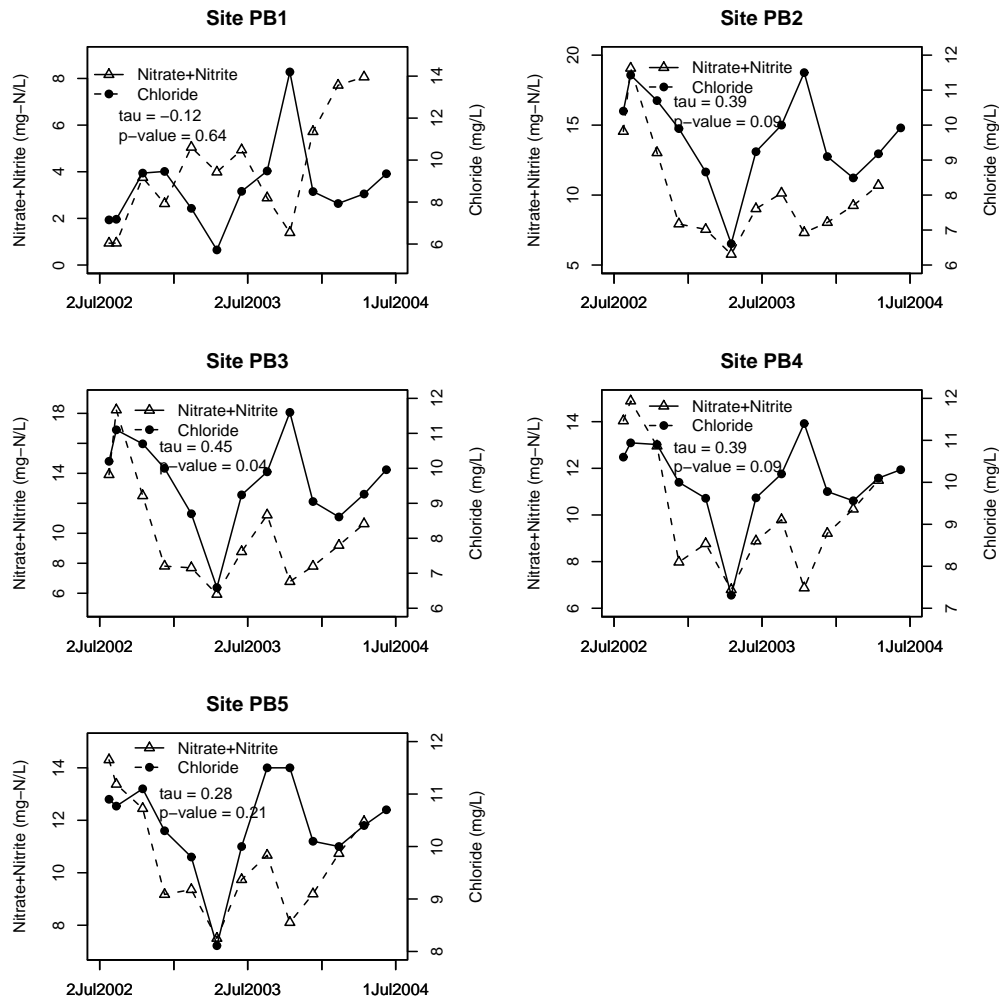


Figure 38: Multiple scatterplots of nitrate+nitrite and chloride concentrations vs. date for the Pangborn Creek stream sites. Note that the scales for nitrate+nitrite and chloride differ within each plot and the y-axis scales for both variables differ from plot to plot, but the date range remains constant. Kendall's tau correlation results for each individual site (shown here) indicate that nitrate+nitrite and chloride are significantly correlated at site PB3 only. An exact p-value could not be computed for site PB5 due to ties; therefore, the test statistic is the estimate scaled to zero mean and unit variance, and is approximately normally distributed.

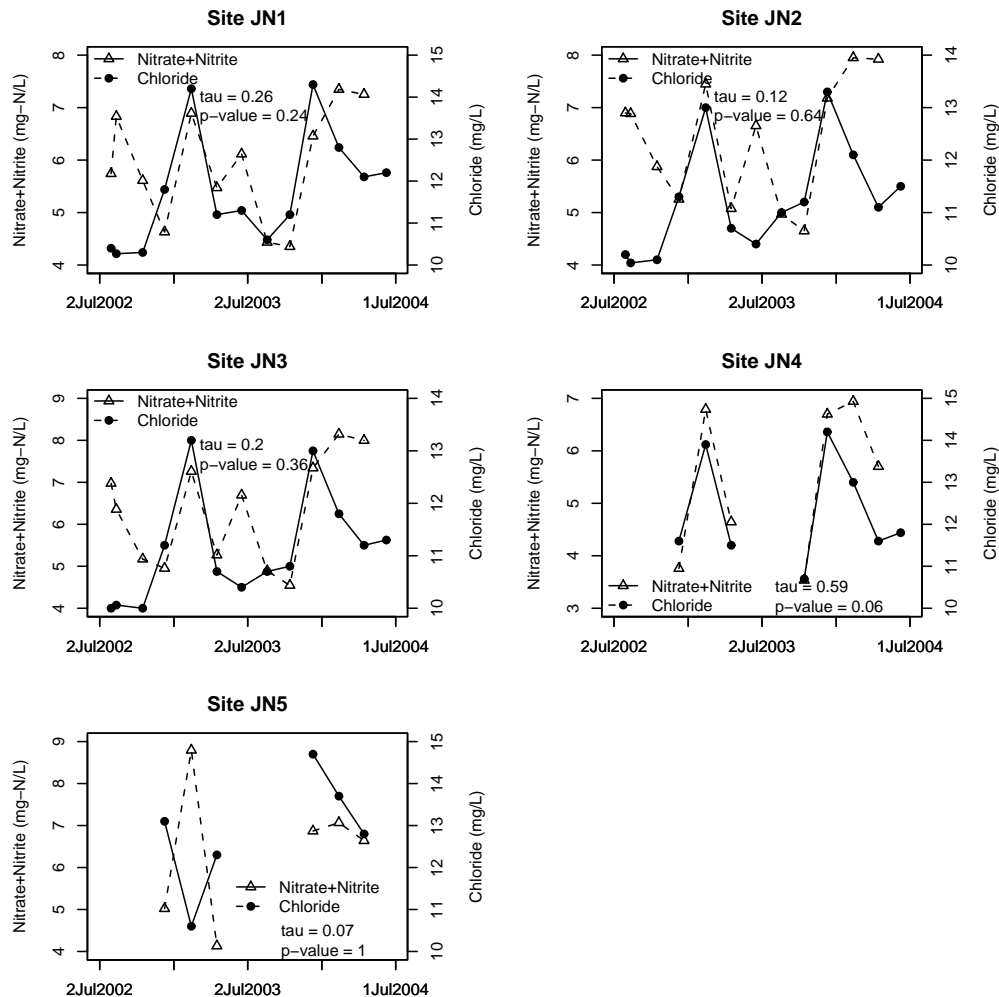


Figure 39: Multiple scatterplots of nitrate+nitrite and chloride concentrations vs. date for the Johnson Creek stream sites. Note that the scales for nitrate+nitrite and chloride differ within each plot and the y-axis scales for both variables differ from plot to plot, but the date range remains constant. Kendall's tau correlation results (shown here) indicate that nitrate+nitrite and chloride are not significantly correlated on a site by site basis. Exact p-values could not be computed for sites JN1, JN3, and JN5 due to ties; therefore, the test statistic is the estimate scaled to zero mean and unit variance, and is approximately normally distributed.

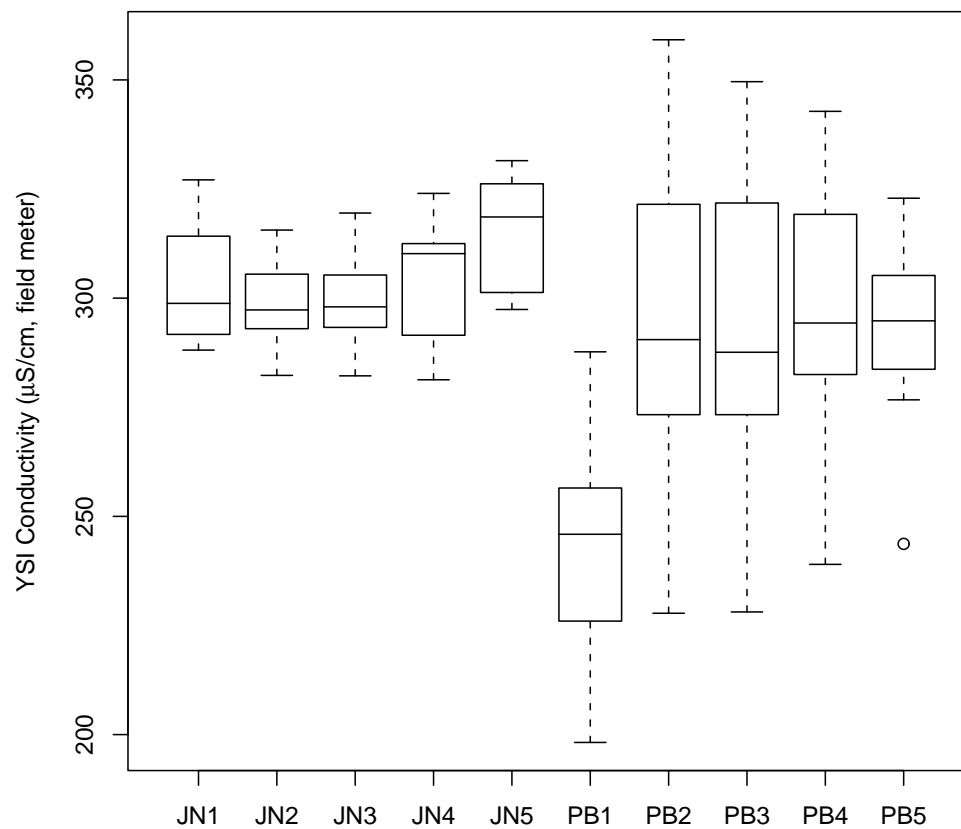


Figure 40: Boxplot of conductivity values for the stream sites. Measurements were made in the field using a YSI model 85 analyzer.

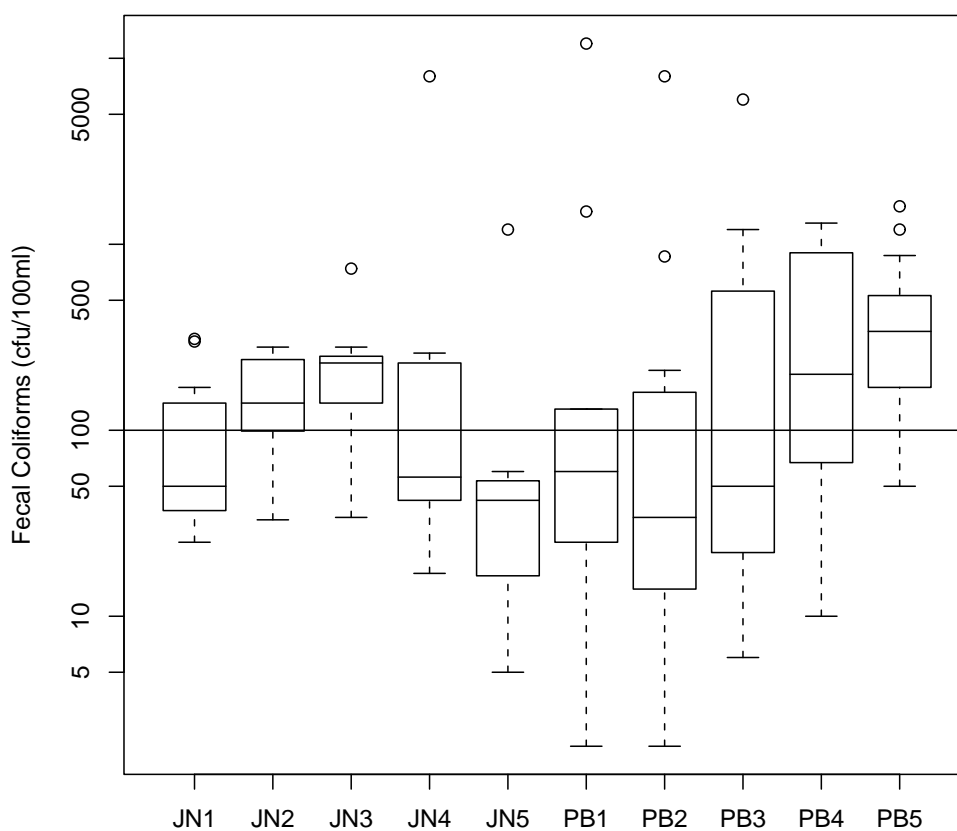


Figure 41: Boxplot of fecal coliform concentrations for the stream sites. The line represents the maximum fecal coliform concentration of 100 cfu/100 mL for Class A streams. Note that concentrations are shown in log scale.

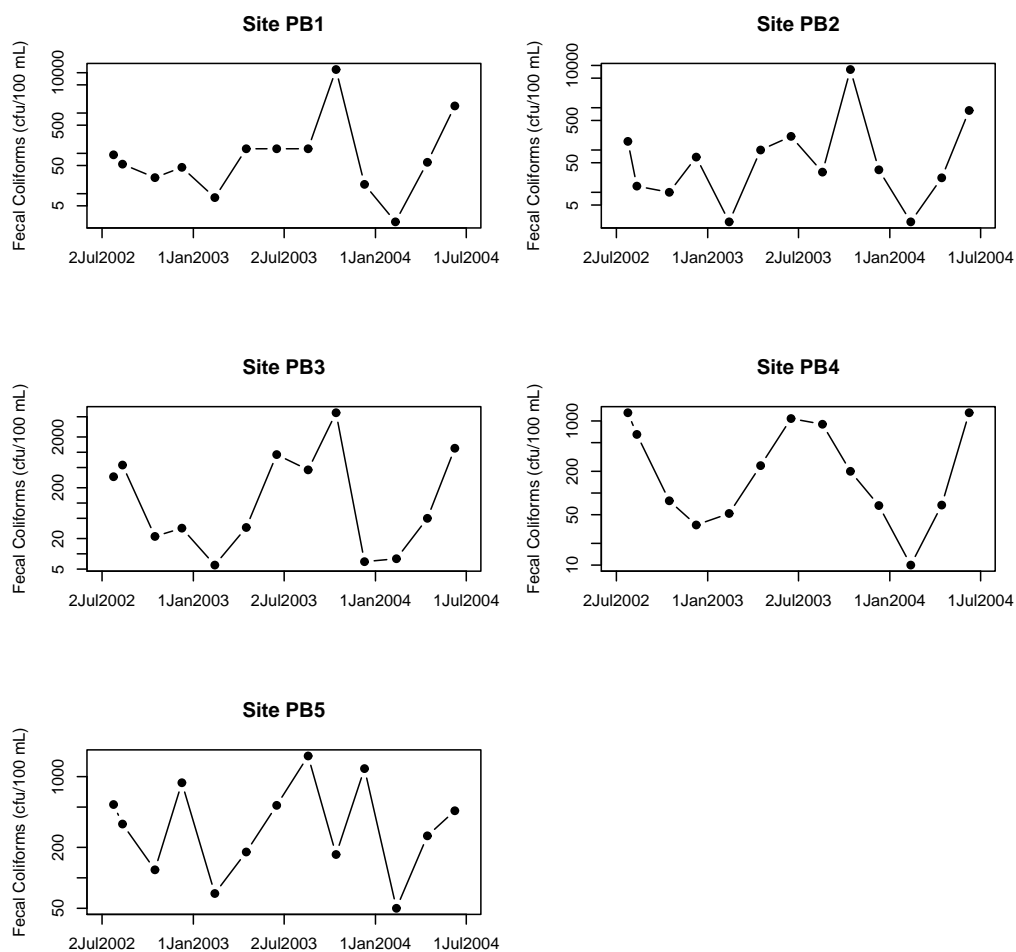


Figure 42: Multiple scatterplots of fecal coliform concentration vs. date for the Pangborn Creek stream sites. Note that concentrations are shown in log scale and that each plot has a different y-axis scale, but the same date range.

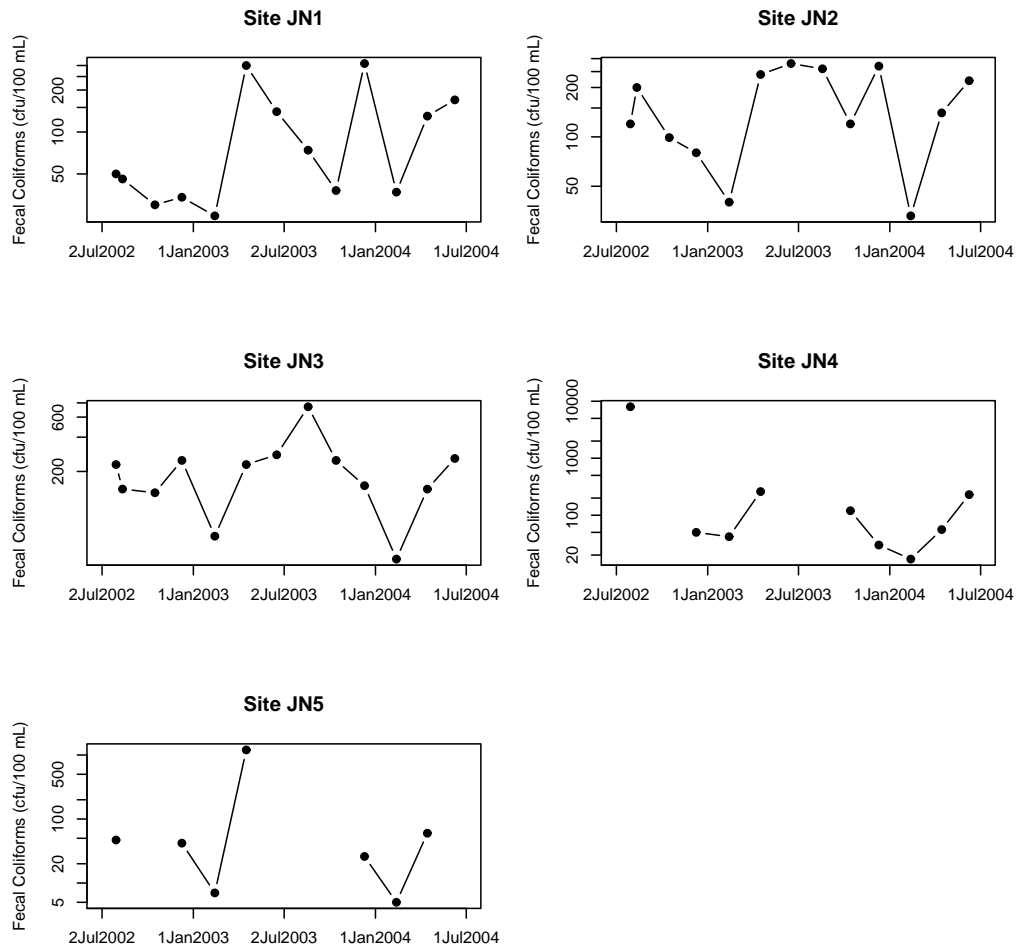


Figure 43: Multiple scatterplots of fecal coliform concentration vs. date for the Johnson Creek stream sites. Note that concentrations are shown in log scale and that each plot has a different y-axis scale, but the same date range.

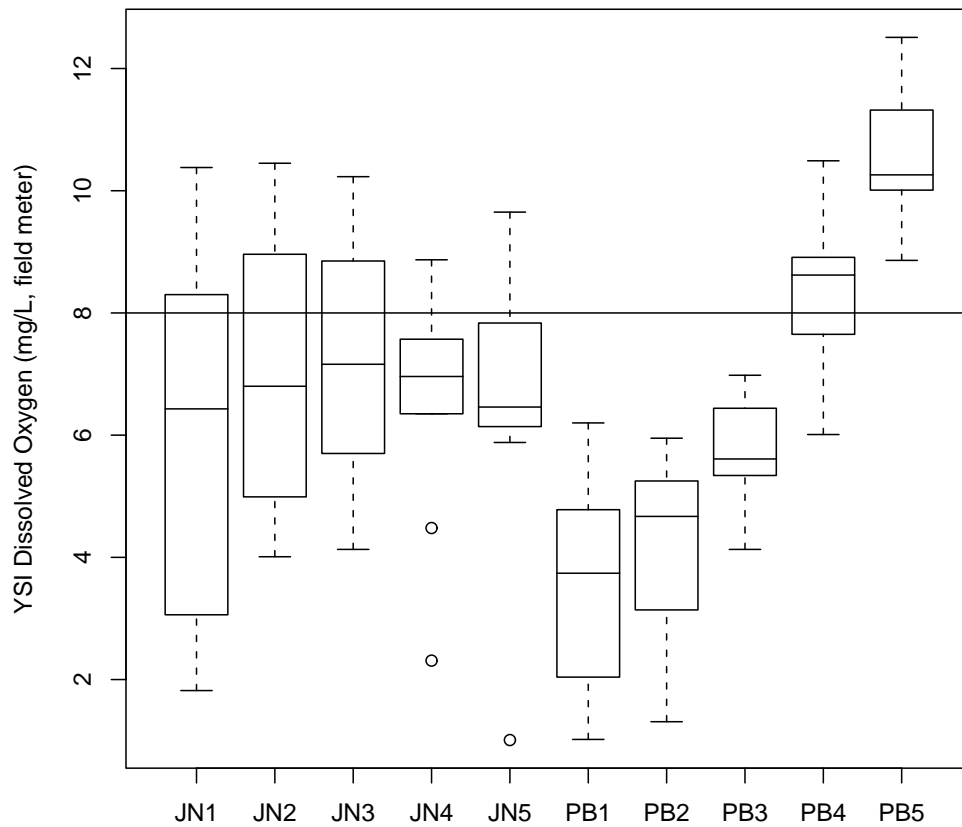


Figure 44: Boxplot of dissolved oxygen concentrations for the stream sites. The line represents the dissolved oxygen limit of 8.0 mg/L that must be exceeded for Class A streams. Measurements were made in the field using a YSI model 85 analyzer.

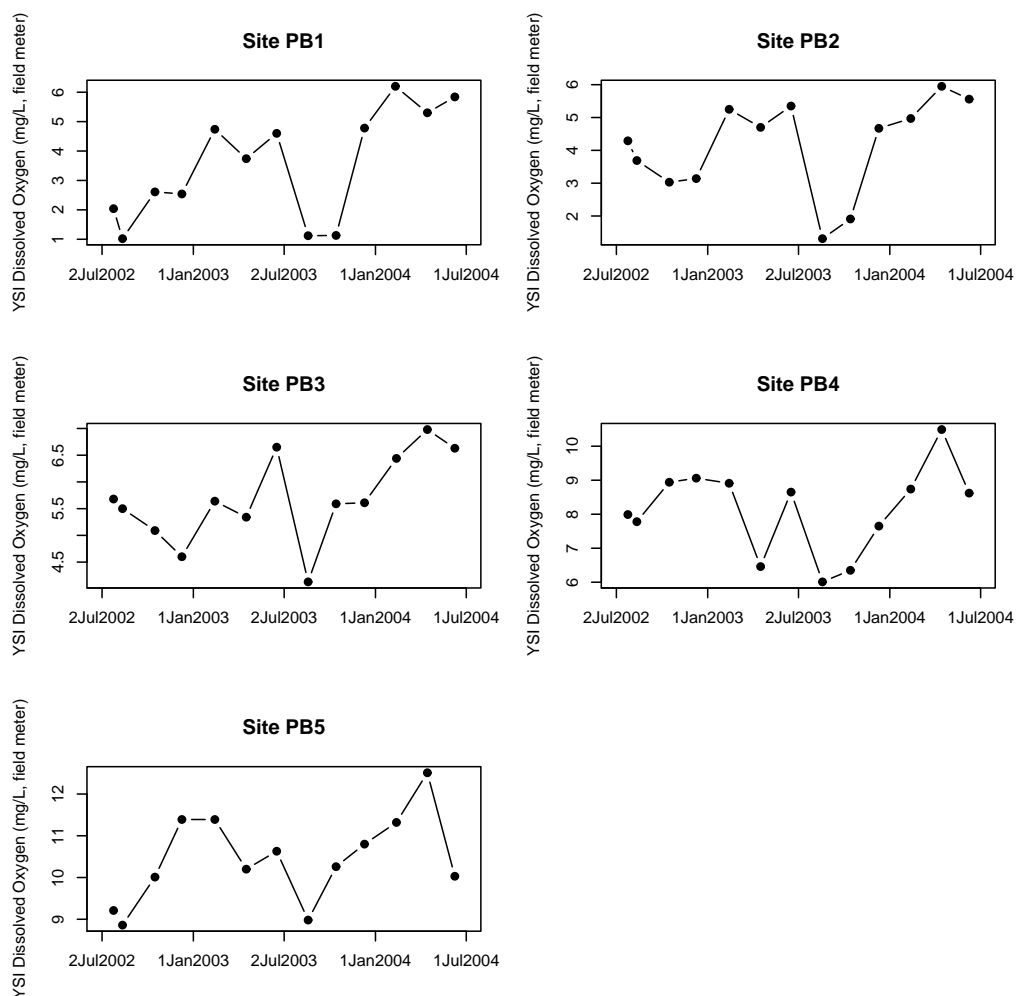


Figure 45: Multiple scatterplots of dissolved oxygen concentration vs. date for the Pangborn Creek stream sites. Measurements were made in the field using a YSI model 85 analyzer. Note that each plot has a different y-axis scale, but the same date range.

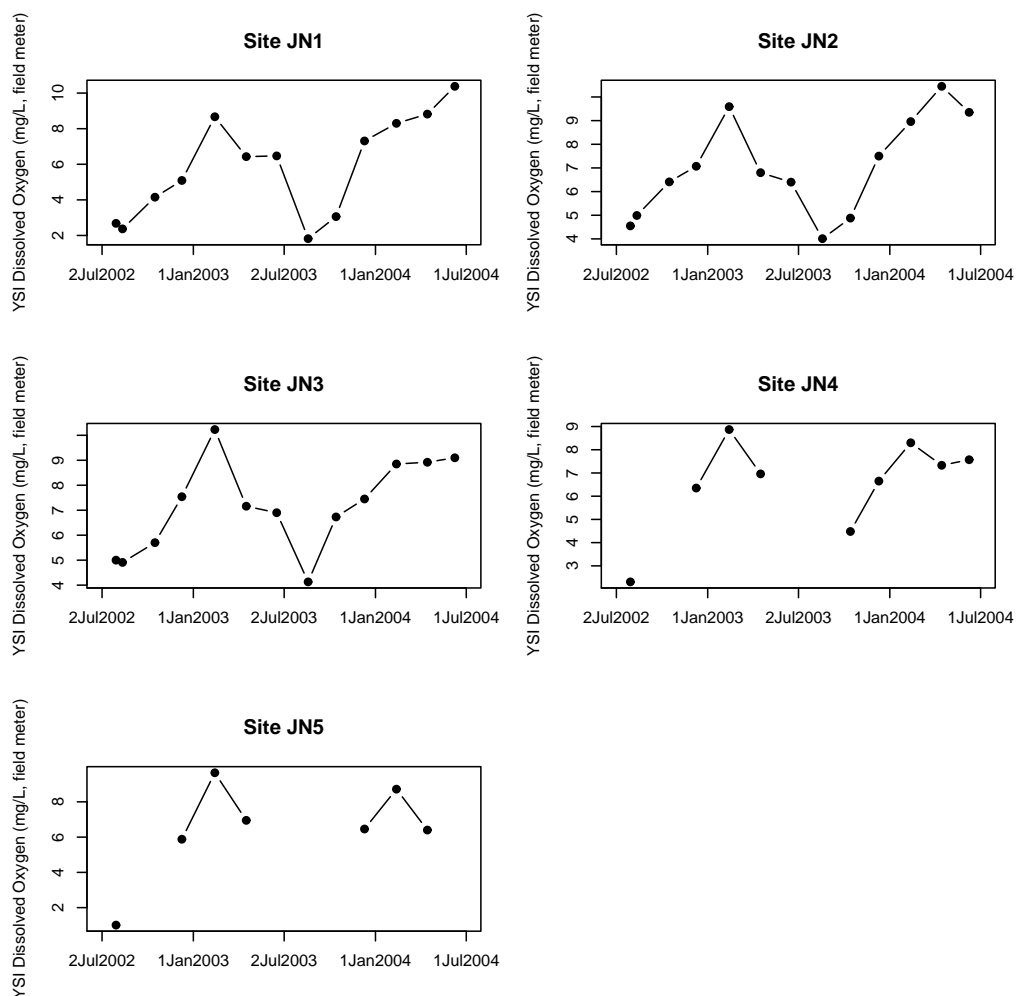


Figure 46: Multiple scatterplots of dissolved oxygen concentration vs. date for the Johnson Creek stream sites. Measurements were made in the field using a YSI model 85 analyzer. Note that each plot has a different y-axis scale, but the same date range.

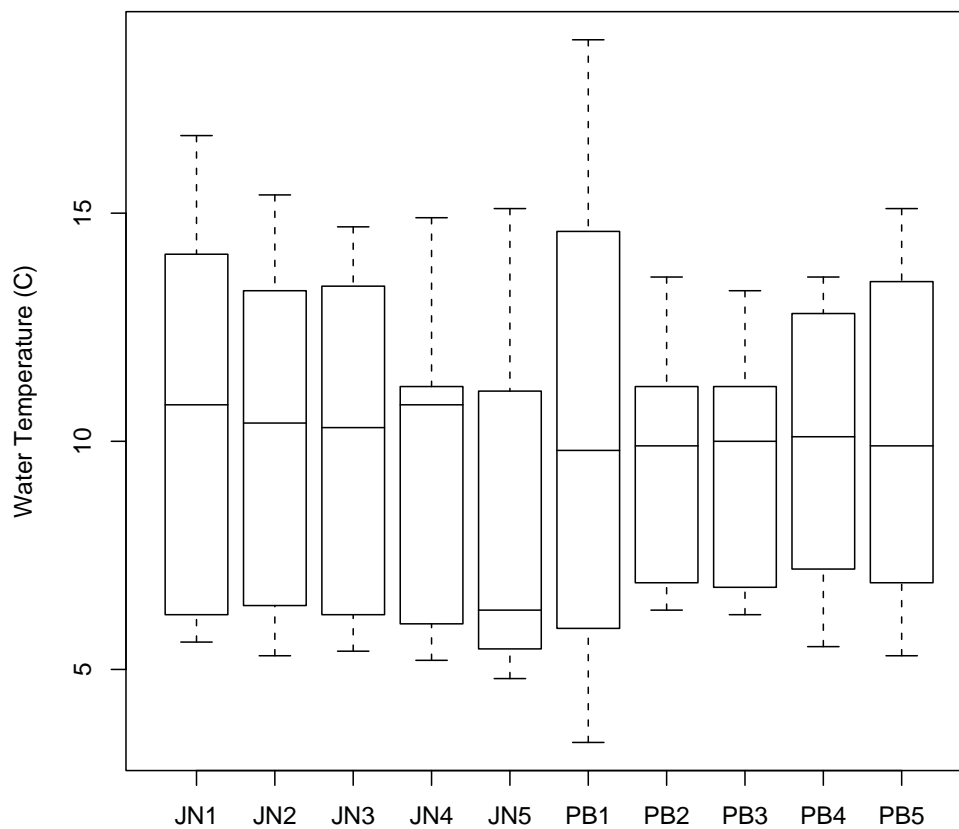


Figure 47: Boxplot of water temperature for the stream sites. Measurements were made in the field using a YSI model 85 analyzer.

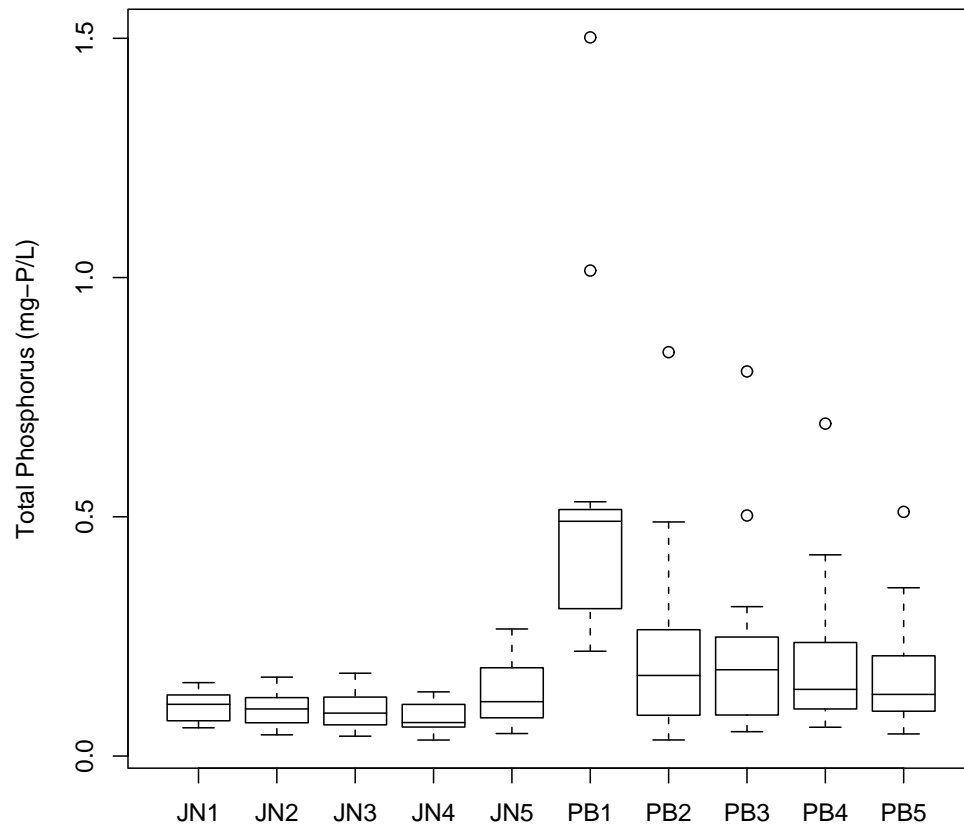


Figure 48: Boxplot of total phosphorus concentrations for the stream sites.

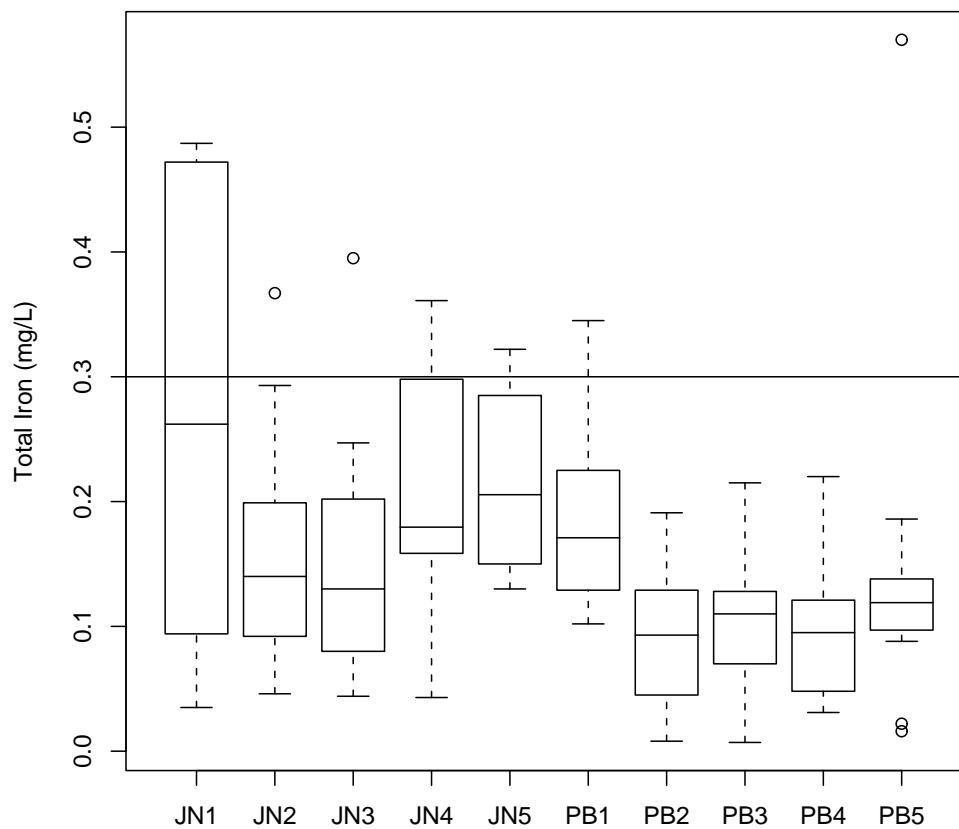


Figure 49: Boxplot of total iron concentrations for the stream sites. The line indicates the ground water quality criterion for total iron of 0.3 mg/L.

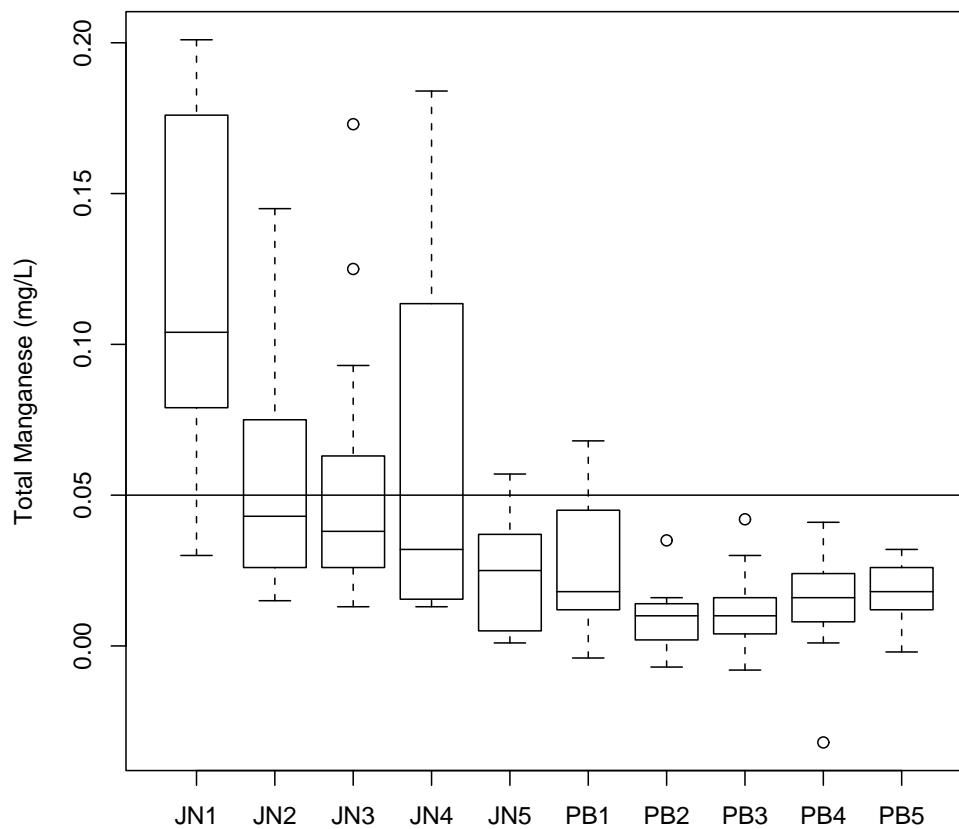


Figure 50: Boxplot of total manganese concentrations for the stream sites. The line indicates the ground water quality criterion for total manganese of 0.05 mg/L.

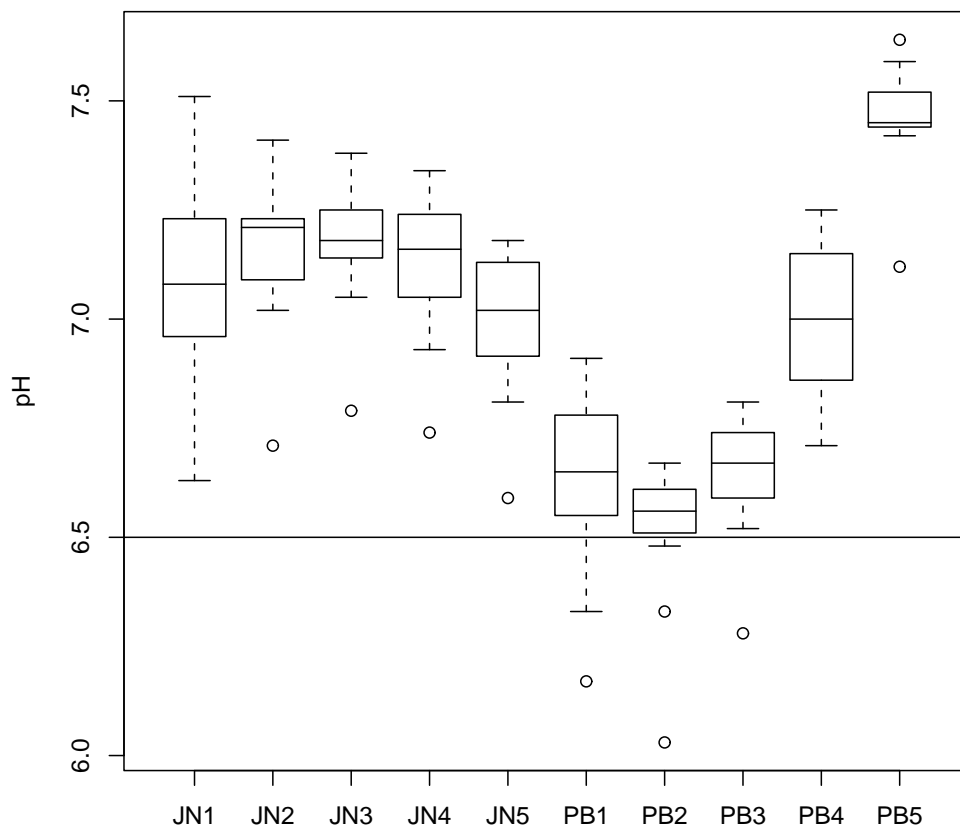


Figure 51: Boxplot of pH values for the stream sites. The lines represent the upper and lower criteria for pH in ground water of 8.5 and 6.5, respectively.

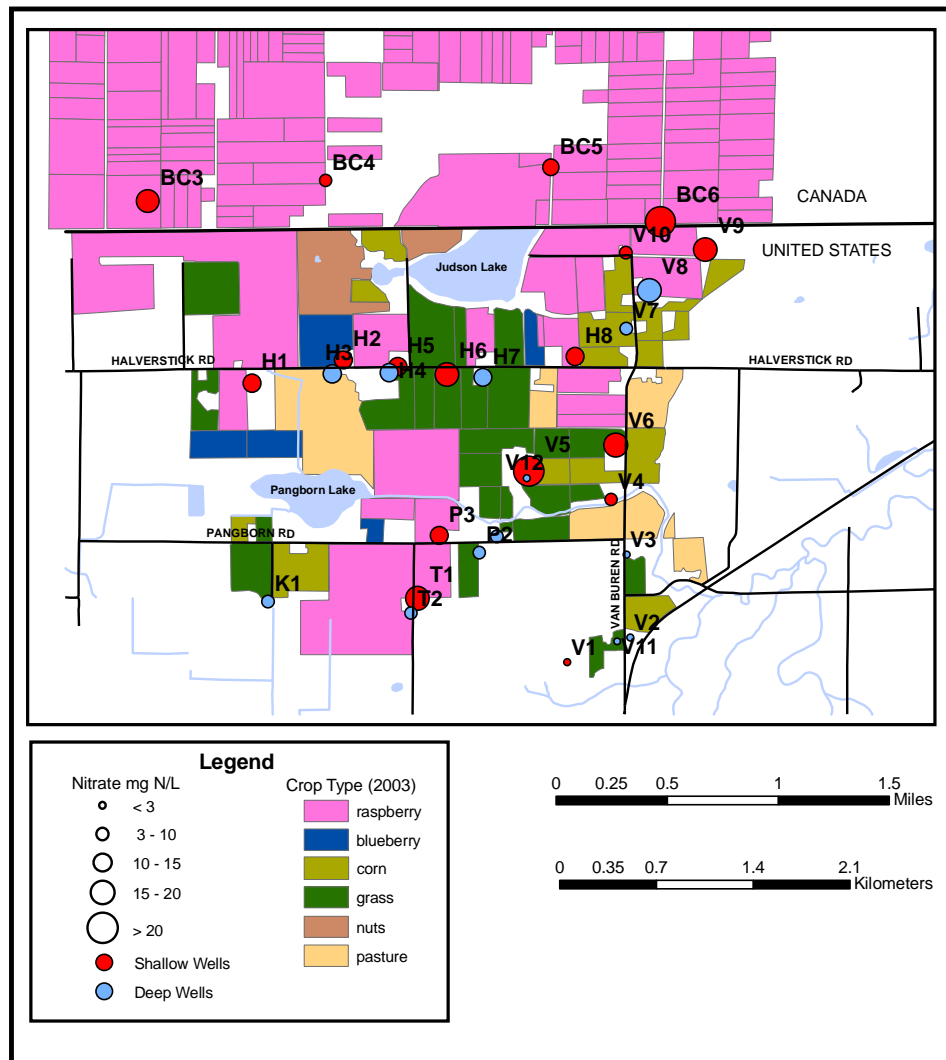


Figure 52: Median groundwater nitrate values and 2003 land use in the Western Washington University study site.

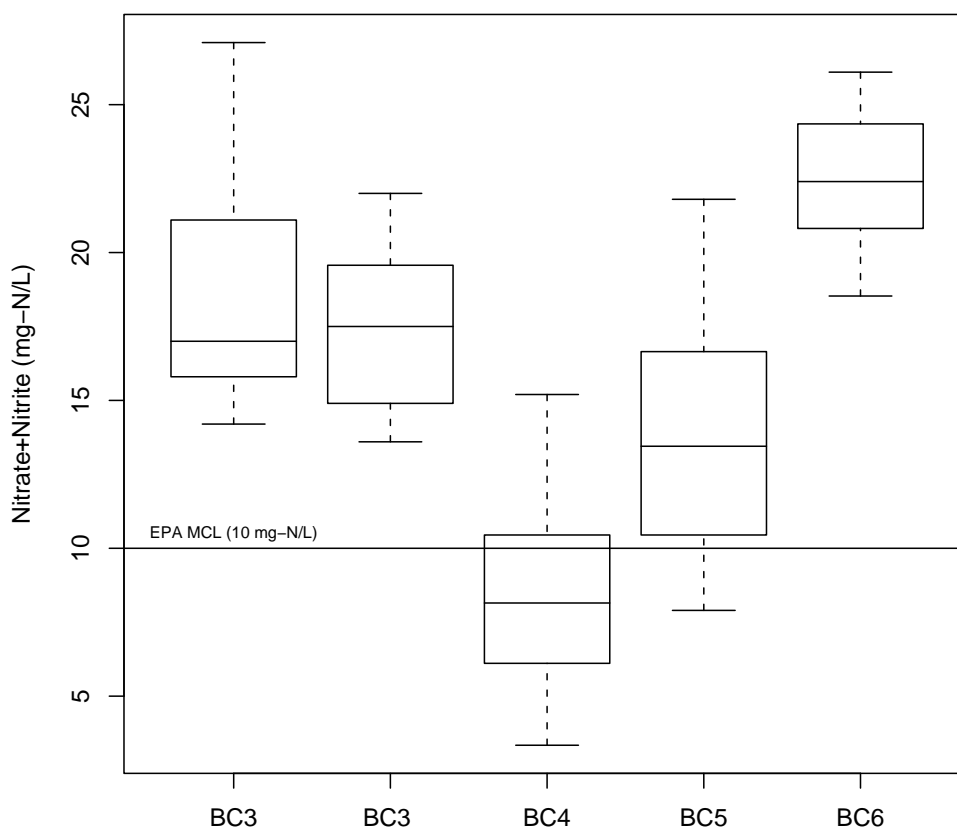


Figure 53: Boxplot of nitrate+nitrite concentrations for the British Columbia (BC) wells located north of the study area across the international boundary. The EPA's Maximum Contaminant Limit (MCL) of 10 mg-N/L for nitrate in drinking water is shown as a line.

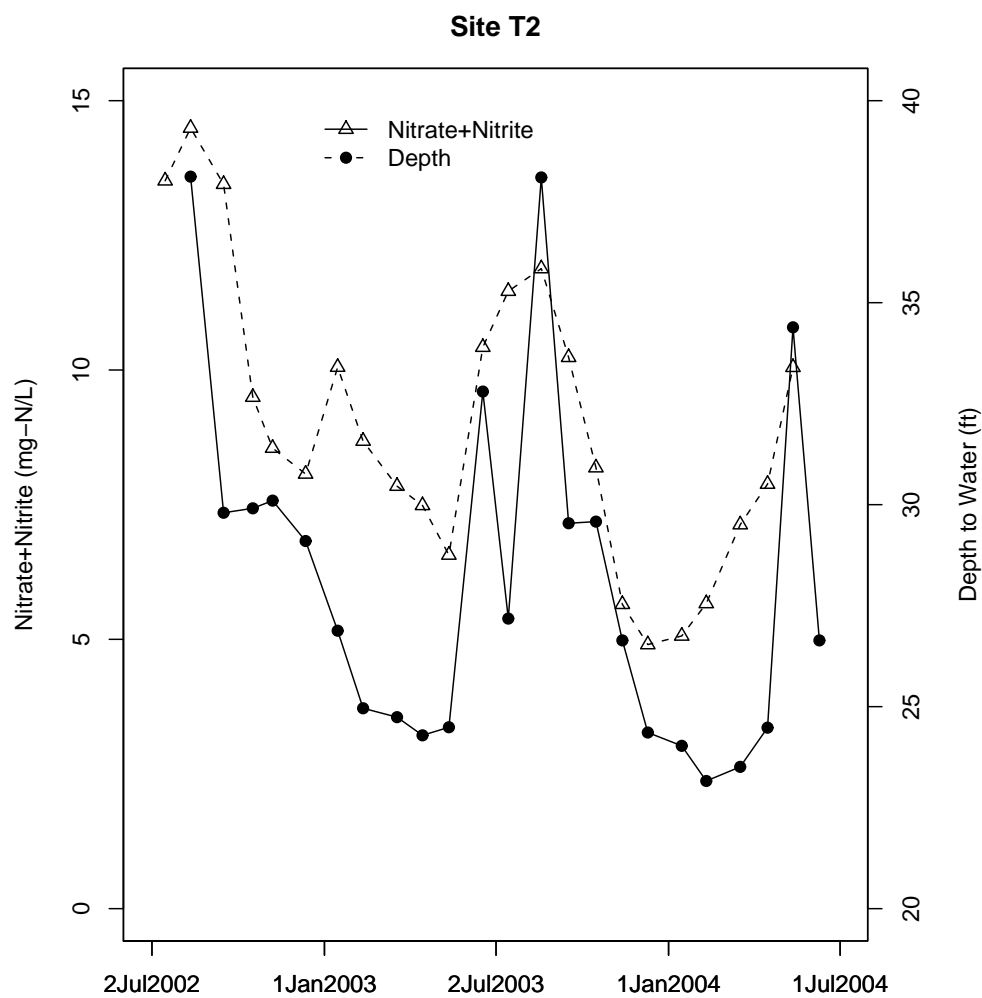


Figure 54: Scatterplot of nitrate+nitrite concentration and depth to water vs. date for well site T2 (an irrigation well). Note that the y-axis scales for nitrate+nitrite and depth to water are different.

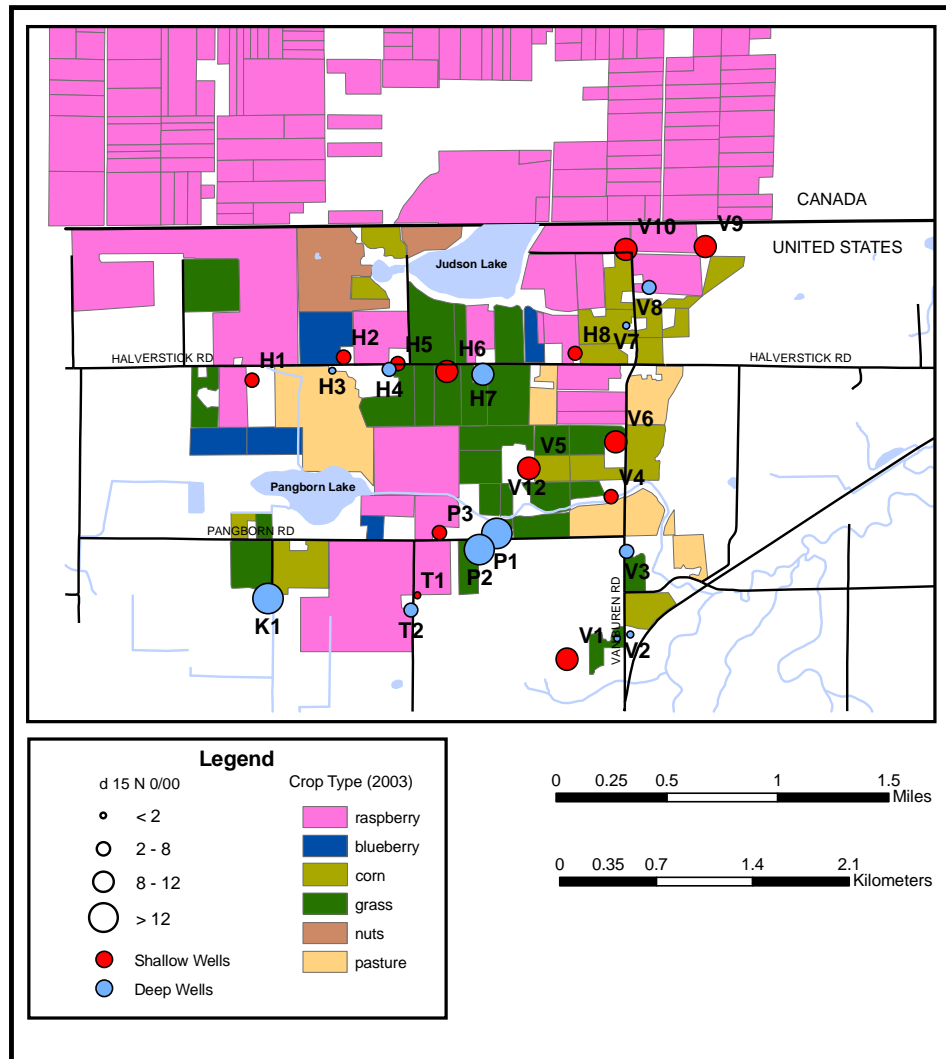


Figure 55: Median groundwater nitrogen isotope values and 2003 land use in the Western Washington University study site.

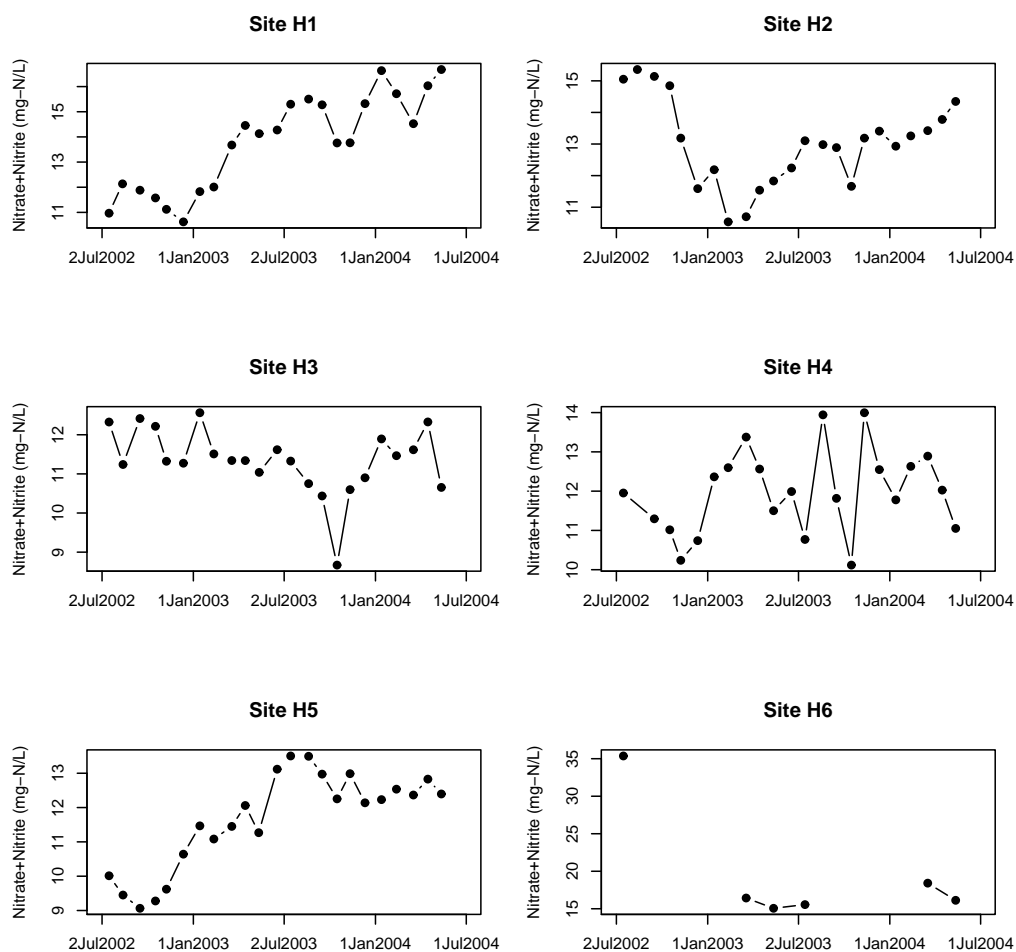


Figure 56: Multiple scatterplots of nitrate+nitrite concentration vs. date for some of the well sites. Note that each plot has a different y-axis scale, but the same date range.

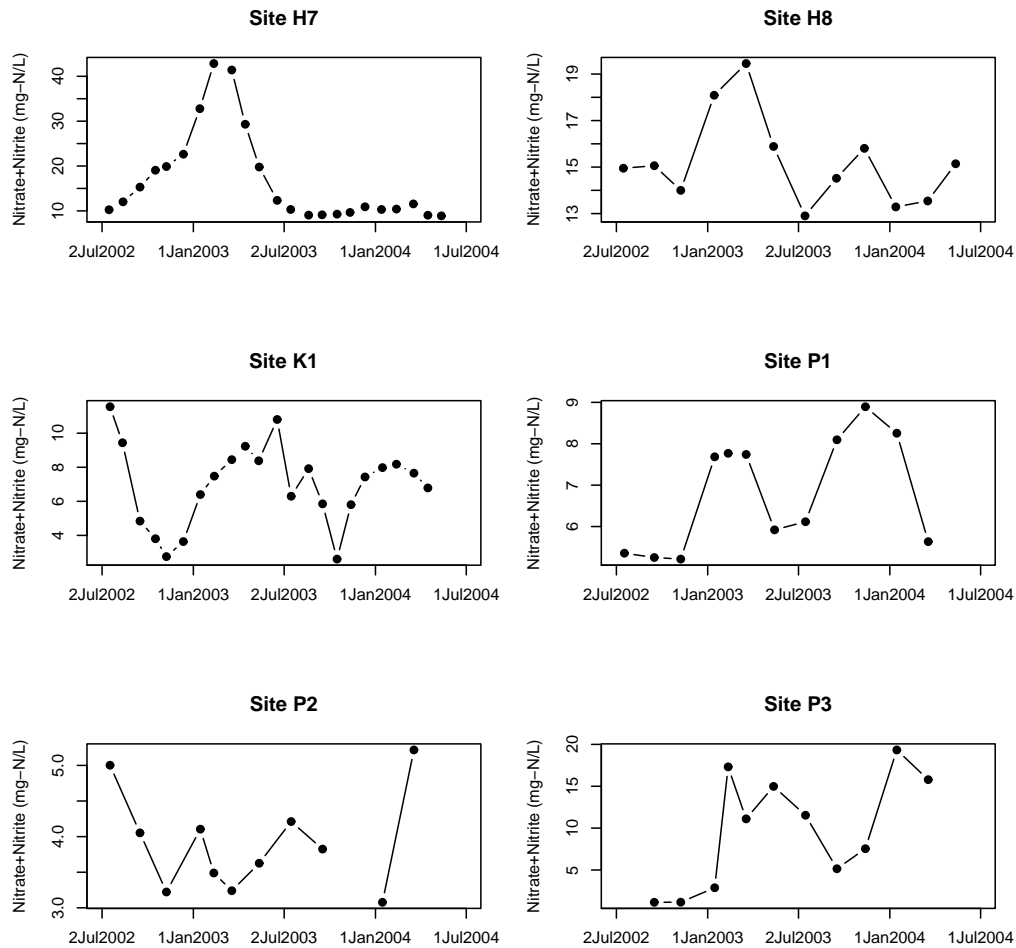


Figure 57: Multiple scatterplots of nitrate+nitrite concentration vs. date for some of the well sites. Note that each plot has a different y-axis scale, but the same date range.

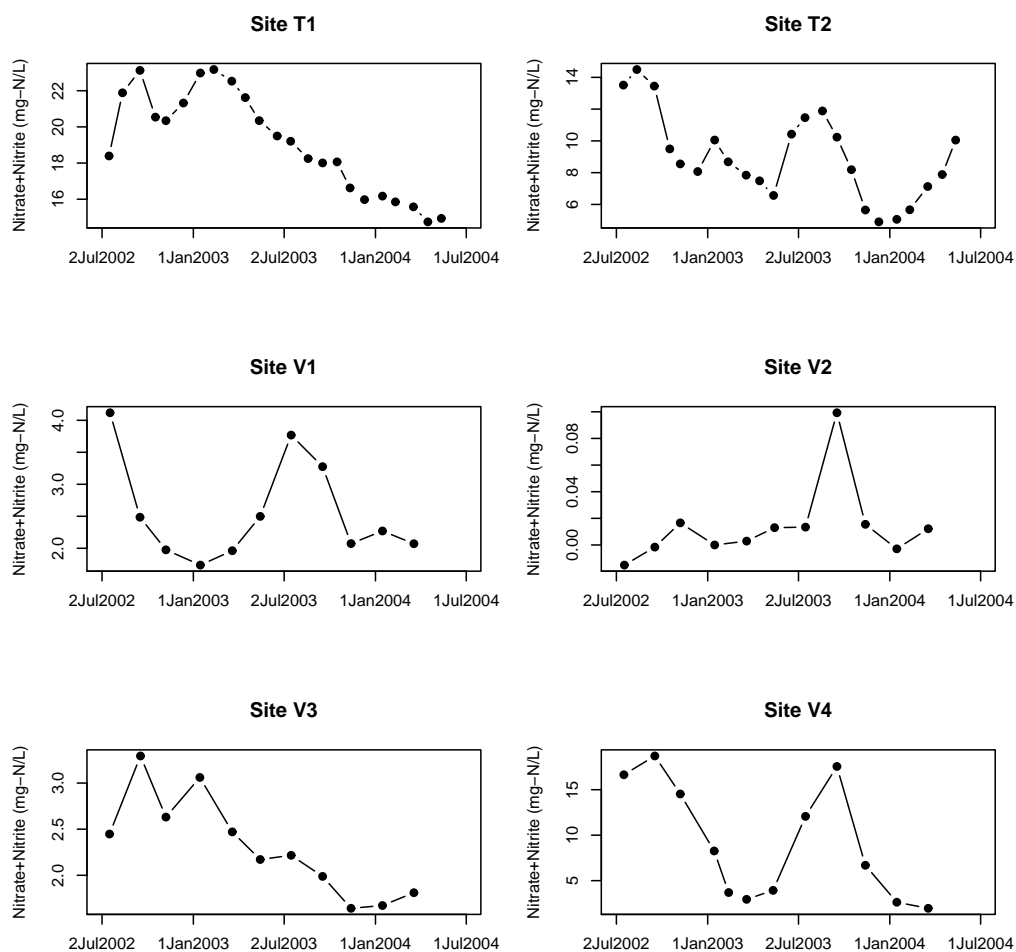


Figure 58: Multiple scatterplots of nitrate+nitrite concentration vs. date for some of the well sites. Note that each plot has a different y-axis scale, but the same date range.

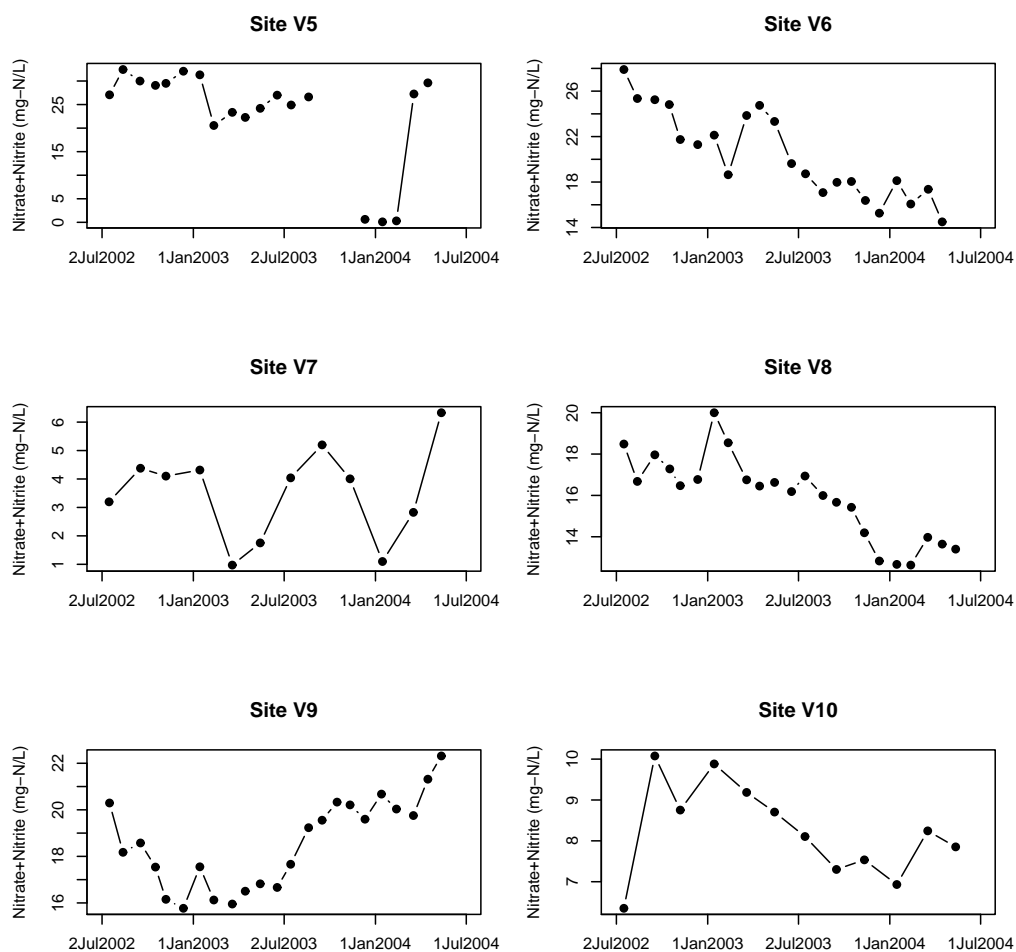


Figure 59: Multiple scatterplots of nitrate+nitrite concentration vs. date for some of the well sites. Note that each plot has a different y-axis scale, but the same date range.

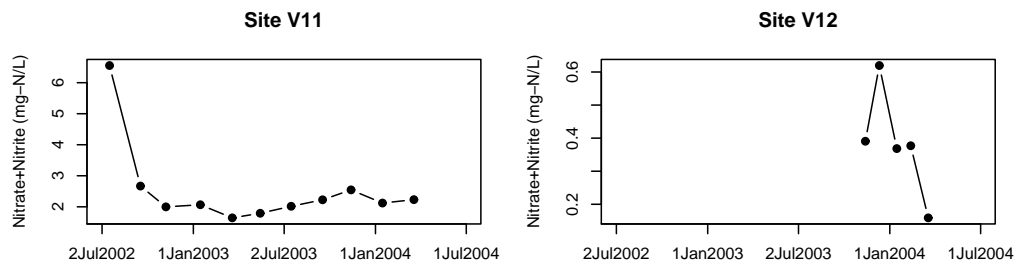


Figure 60: Multiple scatterplots of nitrate+nitrite concentration vs. date for some of the well sites. Note that each plot has a different y-axis scale, but the same date range.

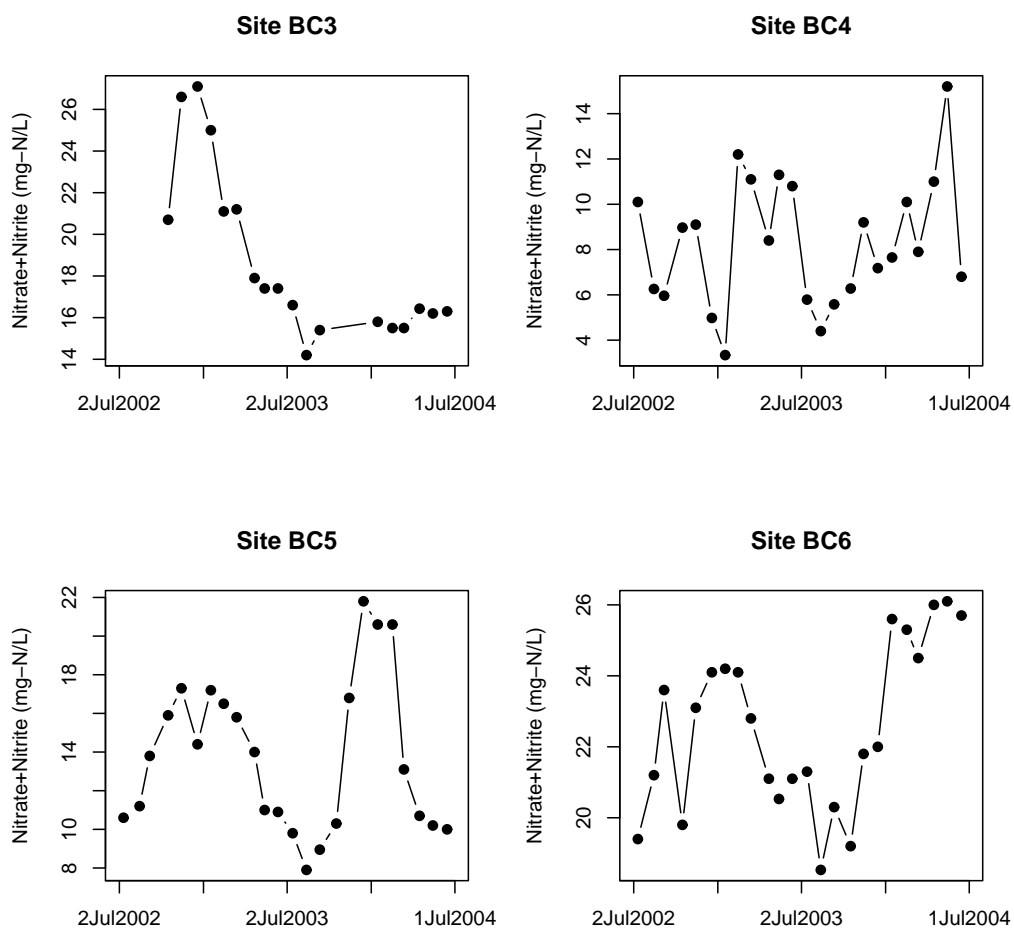


Figure 61: Multiple scatterplots of nitrate+nitrite concentration vs. date by site for the British Columbia (BC) wells located north of the study area across the international boundary. Note that each plot has a different y-axis scale, but the same date range.

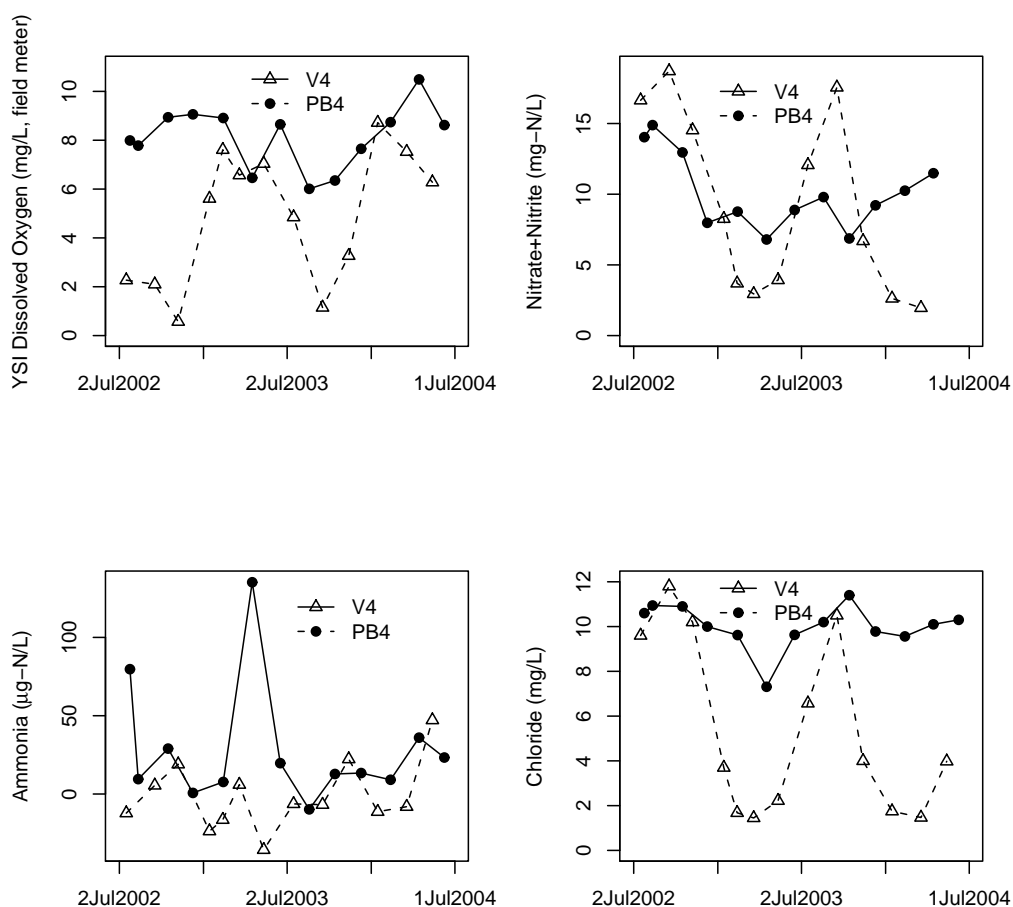


Figure 62: Scatterplots of dissolved oxygen, nitrate+nitrite, ammonia, and chloride concentrations vs. date for well site V4 and stream site PB4. Note that each plot has a different y-axis scale, but the same date range. Dissolved oxygen measurements were made in the field using a YSI model 85 analyzer and, at the well site, a flow-through cell.

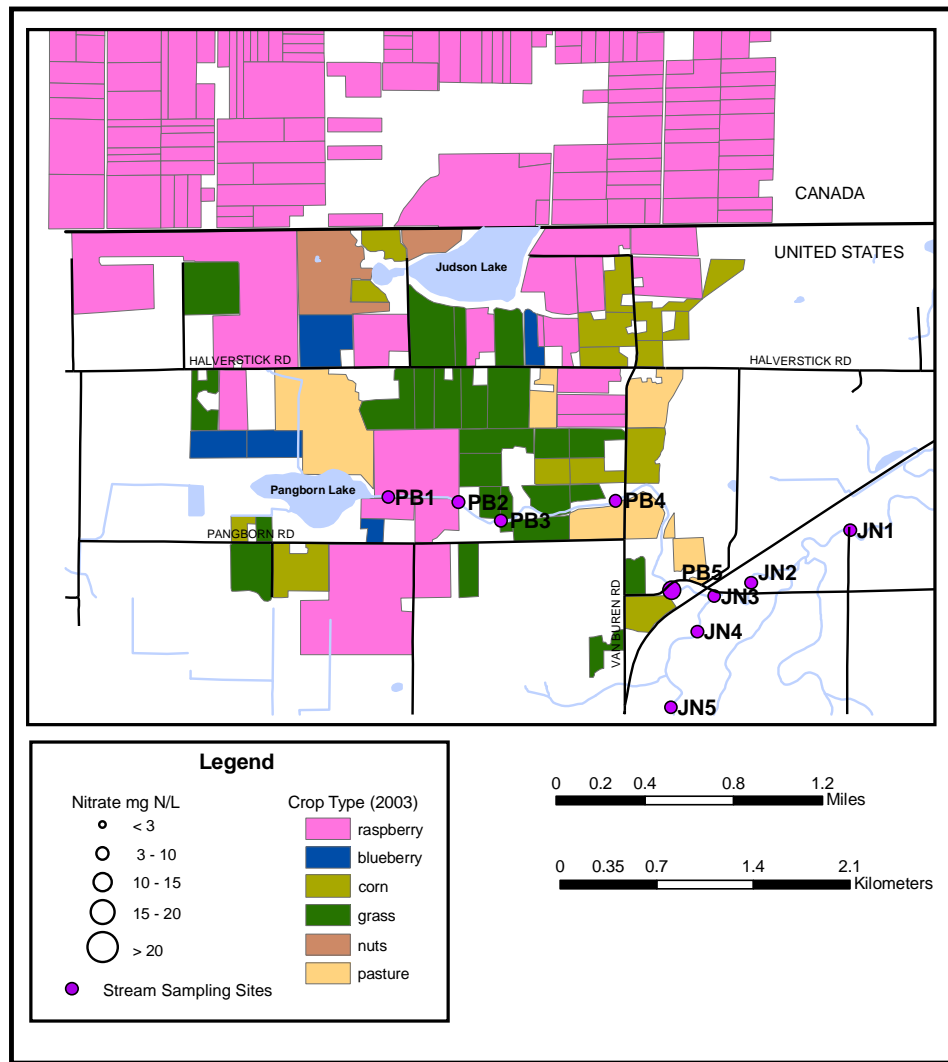


Figure 63: Median surface water nitrate values and 2003 land use in the Western Washington University study site.

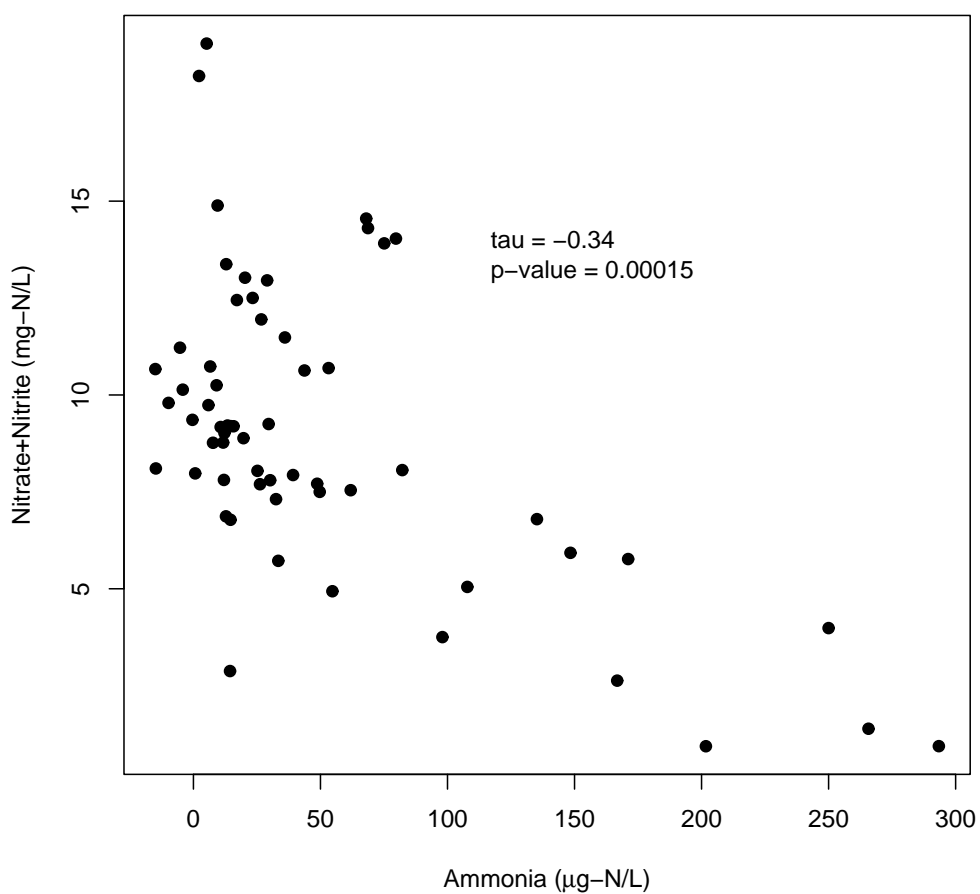


Figure 64: Scatterplot of nitrate+nitrite vs. ammonia concentrations for the Pangborn Creek stream sites only. Kendall's tau correlation results (shown here) indicate a significant correlation.

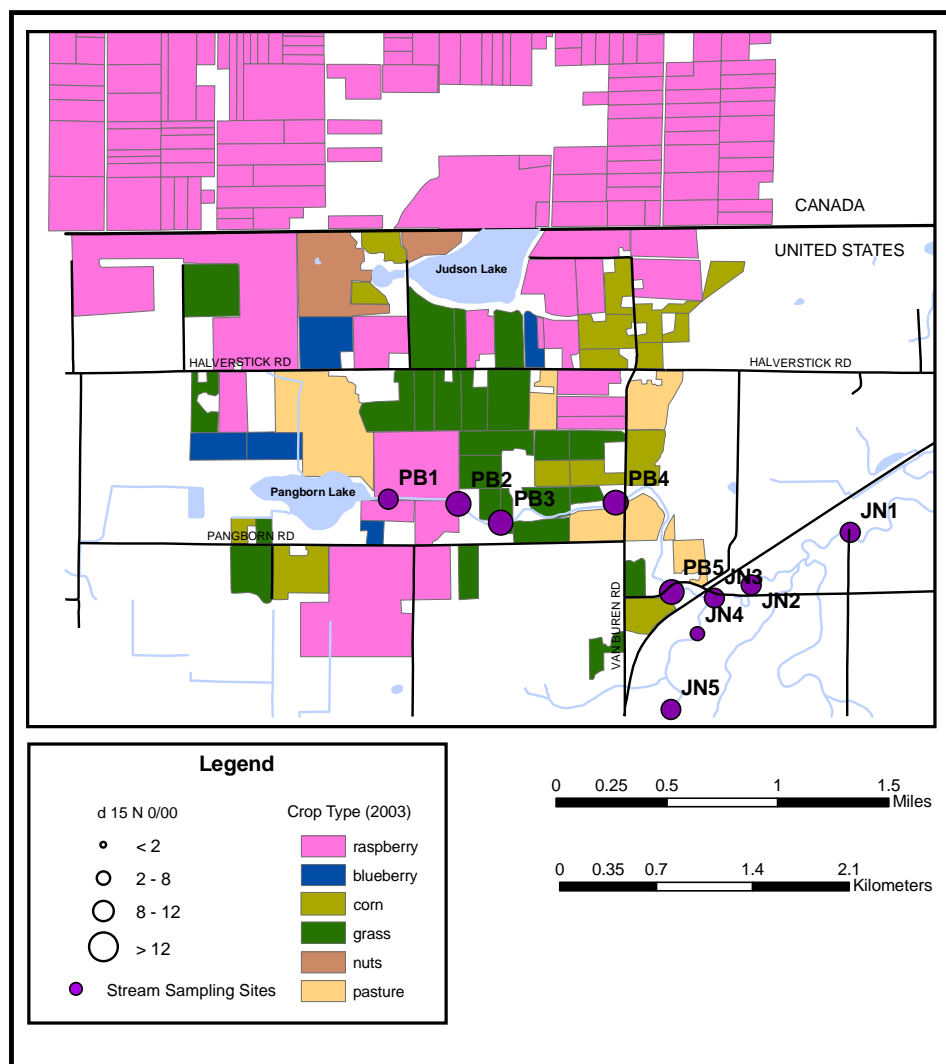


Figure 65: Median surface water nitrogen isotope values and 2003 land use in the Western Washington University study site.

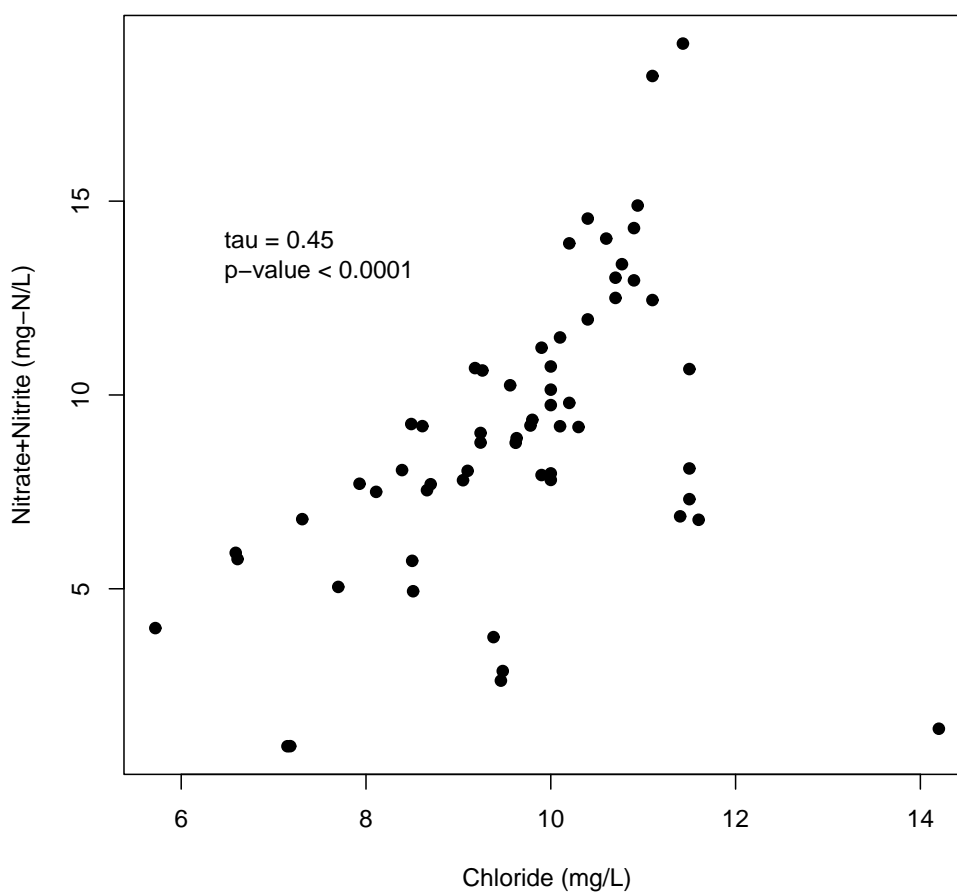


Figure 66: Scatterplot of nitrate+nitrite vs. chloride concentrations for the Pang-born Creek stream sites only. Kendall's tau correlation results (shown here) indicate a significant correlation.

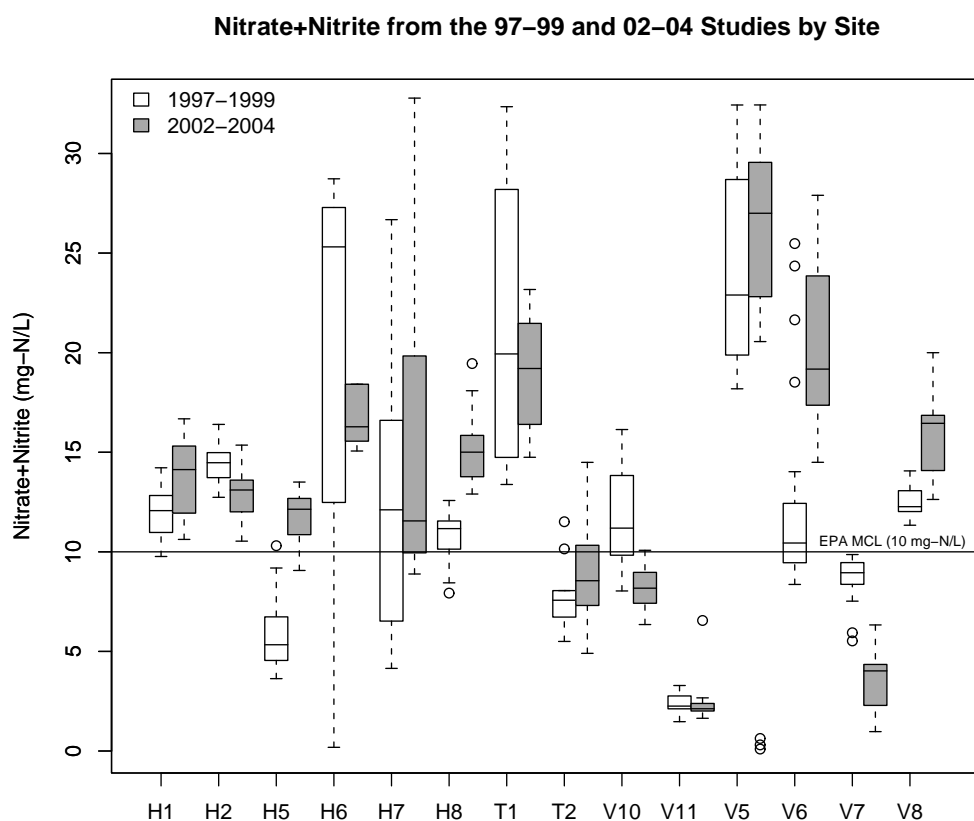


Figure 67: Boxplot of nitrate concentrations from the 1997–1999 and 2002–2004 studies (only the wells that were sampled during both studies are shown). The EPA's Maximum Contaminant Limit (MCL) of 10 mg-N/L for nitrate in drinking water is shown as a line. Note that the shading in this boxplot corresponds to the study years. Refer to the text to determine which wells are shallow vs. deep.

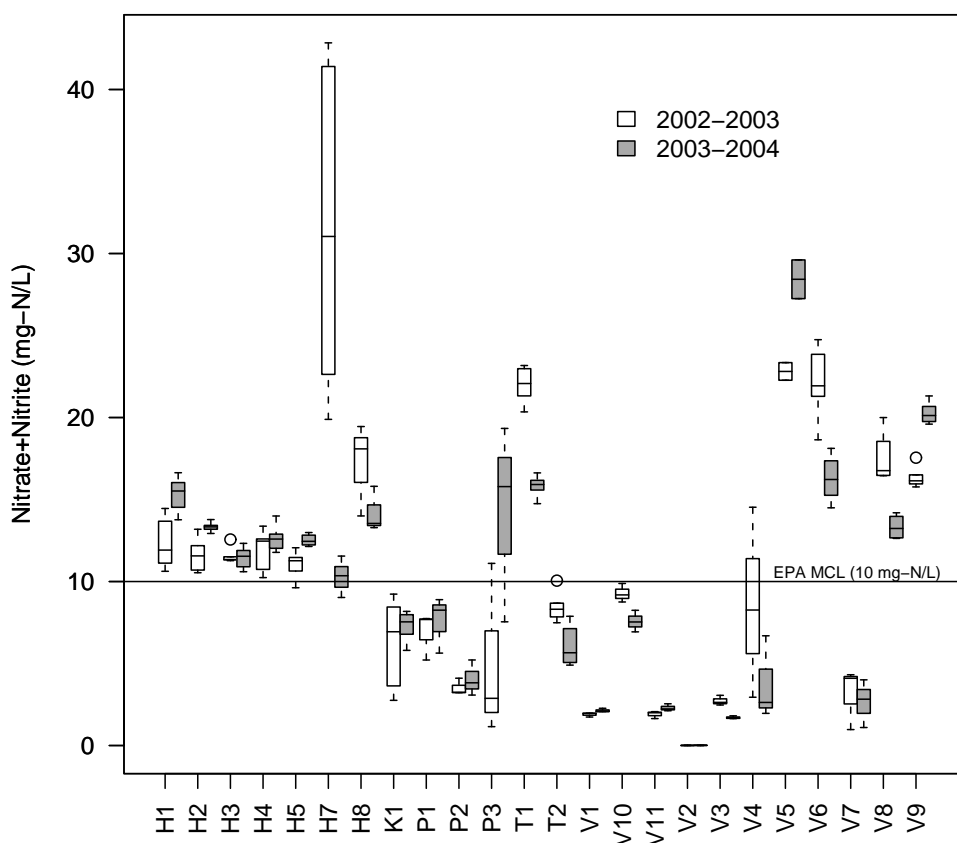
Nitrate+Nitrite from the 02–03 and 03–04 Winter Months by Site

Figure 68: Boxplot of nitrate concentrations from the winter months (Nov–Apr) of 2002–2003 and 2003–2004. The EPA’s Maximum Contaminant Limit (MCL) of 10 mg-N/L for nitrate in drinking water is shown as a line. Note that the shading corresponds to winter years. Refer to the text to determine which wells are shallow vs. deep.

A Quality Control

Data were entered from field sheets and laboratory notebooks into Excel spreadsheets. Each month, the laboratory manager reviewed field and

laboratory log books and spreadsheets for this project. At the end of the project, the laboratory manager verified all printed and electronic versions of the raw data to correct any data transcription or entry mistakes. The data used to calculate statistics were not screened to remove values below detection limits. These data may include negative numbers because the location assigned as zero for the calibration curve has some degree of associated error. Detection limits and holding times for each parameter are listed in Tables 5–6 (pages 113–114).

A.1 Laboratory duplicates

Laboratory duplicates were analyzed for at least 10% of all water quality parameters except the YSI and isotope samples (Figures 69–75, pages 116–122). Laboratory duplicates were used to create control charts that track analytical precision over time. Upper and lower acceptance limits (± 2 standard deviations from the mean pair difference) and upper and lower warning limits (± 3 standard deviations from the mean pair difference) were developed using training data from July 2002 through June 2003. These limits were used to track the continuing performance of the results from July 2003 through June 2004. The control charts indicated that there were no significant changes in analytical precision over time.

A.2 Field duplicates

Field duplicates are used to estimate the variability associated with duplicate samples collected from the same site at the same time. During each sampling period, 10% of the field samples were duplicated. To evaluate the accuracy of the YSI field meter, water samples were collected at 10% of the sampling locations and analyzed in the laboratory following the methods listed in Table 5.

The field duplicate figures show the original and duplicate values and the absolute mean difference between duplicates (Figures 76–79, pages 123–126). Because the data have not been censored, some analytes (e.g., ammonia) had negative or below detection values. For these analytes, the analytical detection limit is indicated on the figure.

The absolute mean differences for ammonia, chloride, pH, nitrate+nitrite, total iron, total manganese, total nitrogen, and YSI conductivity duplicates were similar to expected analytical precision estimates (see Table 5). The mean differences for total phosphorus and YSI dissolved oxygen were higher than the analytical precision estimates in Table 5, but this is not unusual because field duplicates include both sampling and analytical variability. For total phosphorus, most of the variation was caused by three outliers, all with concentrations $\geq 250 \mu\text{g P/L}$. When a 5% trimmed mean was used, the field precision improved considerably (Figure 86).

The YSI dissolved oxygen results indicated a consistent bias, with the YSI field meter reporting higher oxygen concentrations than the laboratory Winkler results. It is not unusual for field dissolved oxygen meters to be less accurate than the Winkler method. Some of the variability may have been introduced during sample collection. The YSI well data were measured inside a sealed flow cell, but came from a flushed water line for the Winkler results. This could have introduced oxygen into the Winkler samples, particularly since duplicates from the streams had a lower mean difference (0.62 mg/L) than duplicates from wells (1.03 mg/L).

A.3 Field blanks

Field blanks were collected on each sampling day by filling an extra sample bottle with deionized water from the IWS laboratory. the field blank was treated identically as if it were a regular sample and was analyzed for ammonia, chloride, nitrate+nitrite, total iron, total manganese, total nitrogen, and total phosphorus. Chloride and total manganese concentrations were below detection in all field blanks (see Table 5 for detection limits). Total phosphorus and total iron was detectable in 4% of the blanks; total nitrogen and nitrate+nitrite concentrations were detectable in 8 and 10% of the blanks, respectively; and ammonia was detectable in 23% of the blanks. The mean and median analyte concentration for all blanks was below detection.

A.4 Laboratory check standards

Two external check standards (20% and 80% of the calibration standard) were analyzed with each analytical run for ammonia, chloride, nitrate+nitrite, total iron, total manganese, total nitrogen, and total phosphorus (Figures 87–93, pages 134–140). Training data from July 2002 through June 2003 were used to generate control charts to verify that analytical precision and accuracy was acceptable. If results were unacceptable, a cause was identified, the sample was re-analyzed if necessary, or the associated sample results were not reported.

A.5 Laboratory spike recoveries

Laboratory spikes were used to measure analyte recovery for ammonia, nitrate+nitrite, total nitrogen, and total phosphorus (Figures 94–97 (pages 141–144). Training data from July 2002 through June 2003 were used to generate control charts to verify that analyte recovery was acceptable. Although no specific recovery objectives were stated in the original sampling plan, a common objective is to maintain at least 80% recovery rates. The upper and lower acceptance limits generated from the training data were all within 80–120% Recovery, and most of the test data fell within the acceptance limits.

A.6 Annual performance evaluations

The IWS laboratory is accredited by the Washington State Department of Ecology. The process of accreditation includes performance and system audits, and IWS participates in regular proficiency testing through an independent contractor. For the Abbotsford-Sumas project, ammonia, laboratory conductivity, nitrate, pH, and total phosphorus are certified analytes; all other analyses were conducted using the same quality-control procedures that have been defined for certified analyses. The Performance Evaluation results for 2002–2004 are included in Table 7 (page 115). All reported values were within acceptance limits for the analysis.

Parameter	Method [†]	Description	Precision	Expected Range	Detection Limit
Ammonia ($\mu\text{g N/L}$)	4500-NH ₃ G	Automated phenate	± 20.9	15–2000	8.4
Chloride (mg/L)	4110	Ion chromatography	± 0.2	0–20	1.0
Conductance ($\mu\text{S/cm}$)	2510	YSI 85	± 2.2	70–400	2
Dissolved oxygen (mg/L)	4500-O G	Membrane electrode YSI 85	± 0.1	0–10	0.1
Dissolved oxygen (mg/L)	4500-O C	Winkler method	± 0.1	0–10	0.1
Fecal coliforms (cfu/100 mL)	9222	Membrane fi lter	na	<1–2000	1
Nitrate+nitrite ($\mu\text{g N/L}$)	4500-NO ₃ I	Automated Cd reduction	± 31.2	10–15000	16.1
pH	4500-H	Orion EA940 meter	± 0.1	5–9	na
Temperature	2550	Thermometric	± 0.1	4–18	na
Total iron (mg/L)	3500-Fe	AA, direct aspiration	± 0.09	0–2	0.06
Total manganese (mg/L)	3500-Mn	AA, direct aspiration	± 0.05	0–2	0.03
Total nitrogen ($\mu\text{g N/L}$)	4500-N C	Persulfate oxidation/auto Cd reduction	± 16.2	10–15000	15.9
Total phosphorus ($\mu\text{g N/L}$)	4500-P H	Persulfate oxidation/auto colorimetry	± 2.3	10–500	8.3

[†]APHA (1998)

Table 5: Analytical methods, analytical precision, expected range, and detection limits for the 2002–2004 Abbotsford-Sumas Monitoring Project.

Parameter	APHA Storage/ Max Holding Time	Regulatory Holding Time	Container	IWS Storage/ Holding Time
Iron and manganese	Filter on site, 6 mo ultra-pure HNO ₃ to pH<2 (mercury, 28 days)	6 mo	Nalgene	Filter within 8 hr, 6 mo ultra-pure HNO ₃ to pH<2 (mercury, 28 days)
Temperature and d. oxygen (probe)	Analyzed immediately	na	na	Analyze immediately
Chloride	28 days	28 days	Nalgene	28 days
Conductivity	Cool, 4° C, 28 days	28 days	Nalgene	Cool, 4° C, 28 days
pH	None required, analyze immediately	15 min	Nalgene, no	Cool, 4° C 8 hr for natural waters
D. oxygen (Winkler)	Fix on site, store dark, 8 hrs	8 hrs	BOD glass bottle	Fix on site, store 4° C, dark 3 days with water seal
Ammonia	Filer, cool, 4° C, H ₂ SO ₄ to pH <2, 7 days	28 days	Nalgene	Filter, freeze [†] , analyze within 28 days
Nitrate+nitrite	Filer, cool, 4° C, H ₂ SO ₄ to pH <2, 1–2 days	28 days	Nalgene	Filter, freeze [†] , analyze within 60 days
Ammonia	Cool, 4° C, H ₂ SO ₄ to pH <2, 28 days	28 days	Nalgene	Freeze [†] , analyze within 28 days, or digest within 8 hrs and hold up to 28 days

[†]Freezing is in replacement of preservation with H₂SO₄

Table 6: Proper container, storage, and sample holding time for the 2002–2004 Abbotsford-Sumas Monitoring Project.

	PE Report Date	Reported Value [†]	True Value [†]	Acceptance Limits
Specific conductance ($\mu\text{S}/\text{cm}$ at 25°C)	May 22, 2002	952.0	934	856–1012
	Nov 15, 2002	1140	1100	1008–1192
	June 30, 2003	465.0	462	426–499
	June 1, 2004	1044.0	1020	935–1110
Ammonia-N, automated phenate (mg N/L)	May 22, 2002	14.1	14.5	11.3–17.6
	Nov 15, 2002	5.39	5.36	4.11–6.57
	June 30, 2003	1.35	1.30	0.926–1.69
	June 1, 2003	1.03	1.20	0.847–1.56
Nitrate-N, automated Cd reduction (mg N/L)	May 22, 2002	23.2	22.5	17.8–26.7
	Nov 15, 2002	9.46	9.39	7.42–11.2
	June 30, 2003	36.8	36.6	29.0–43.4
	June 1, 2003	6.59	6.58	5.19–7.84
Total phosphorus, persulfate oxidation (mg P/L)	May 22, 2002	4.65	4.98	3.79–5.84
	Nov 15, 2002	4.87	5.28	4.01–6.19
	June 30, 2003	4.47	4.30	3.27–5.05
	June 1, 2003	3.68	3.78	2.87–4.44
pH	May 22, 2002	5.16	5.20	5.10–5.31
	Nov 15, 2002	5.68	5.70	5.58–5.86
	June 30, 2003	9.38	9.20	8.92–9.48
	June 1, 2003	5.86	5.90	5.77–5.99

Table 7: Summary of 2002–2004 Performance Evaluation Reports WP-073, WP-077, WP-083, and WP-093.

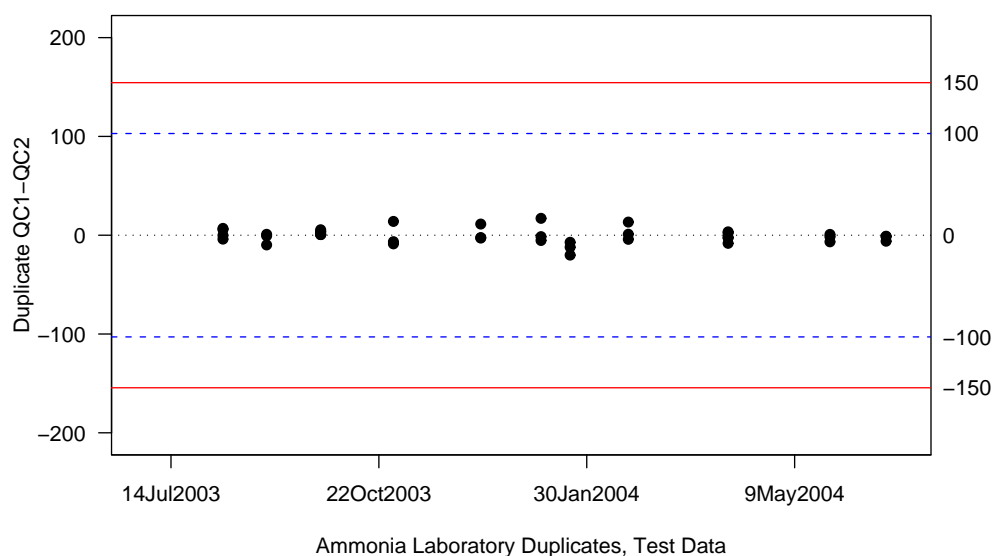
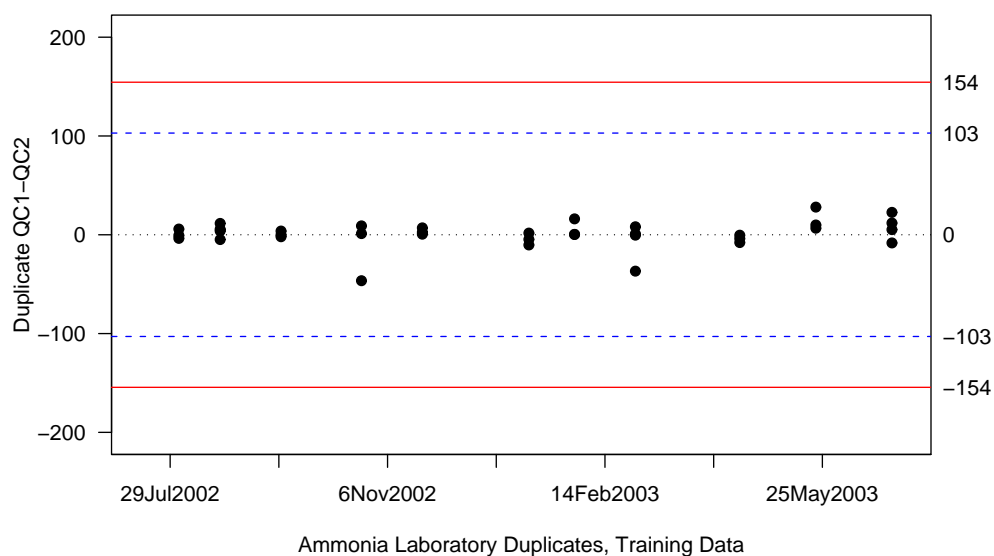


Figure 69: Ammonia laboratory duplicates for the 2002–2004 Abbotsford-Sumas Monitoring Project. Upper/lower acceptance limits (± 2 std. dev. from mean pair difference) and upper/lower warning limits (± 3 std. dev. from mean pair difference) were calculated using data from July 2002 through June 2003.

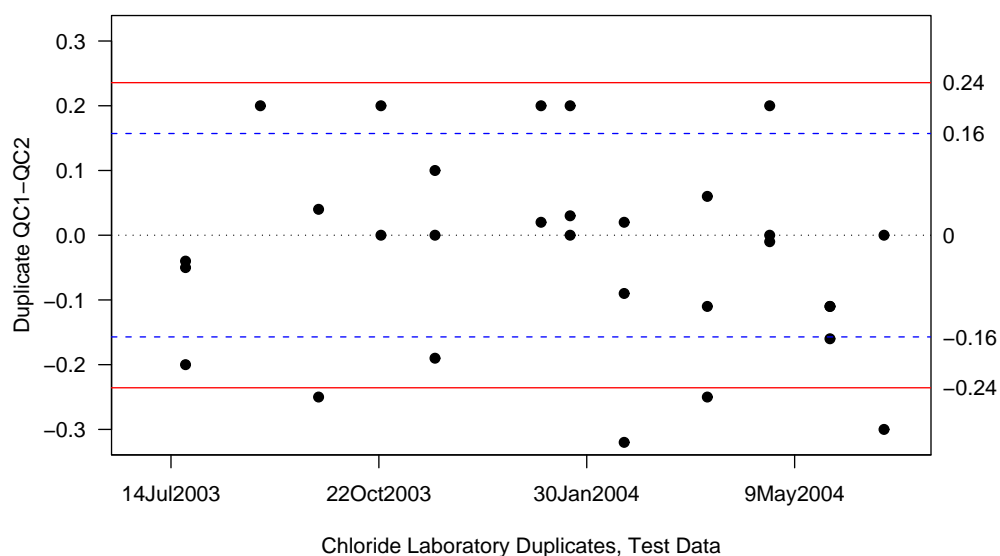
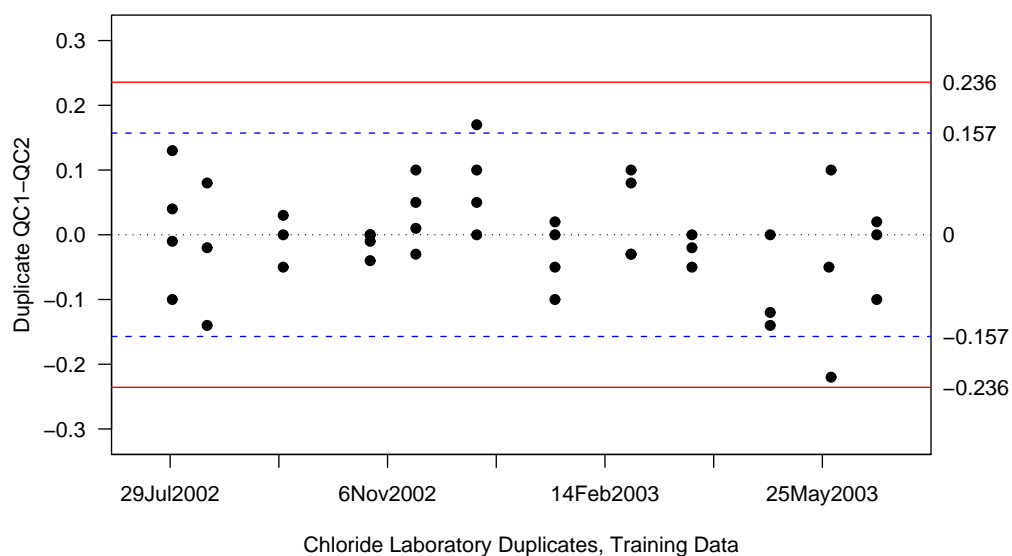


Figure 70: Chloride laboratory duplicates for the 2002–2004 Abbotsford-Sumas Monitoring Project. Upper/lower acceptance limits (± 2 std. dev. from mean pair difference) and upper/lower warning limits (± 3 std. dev. from mean pair difference) were calculated using data from July 2002 through June 2003.

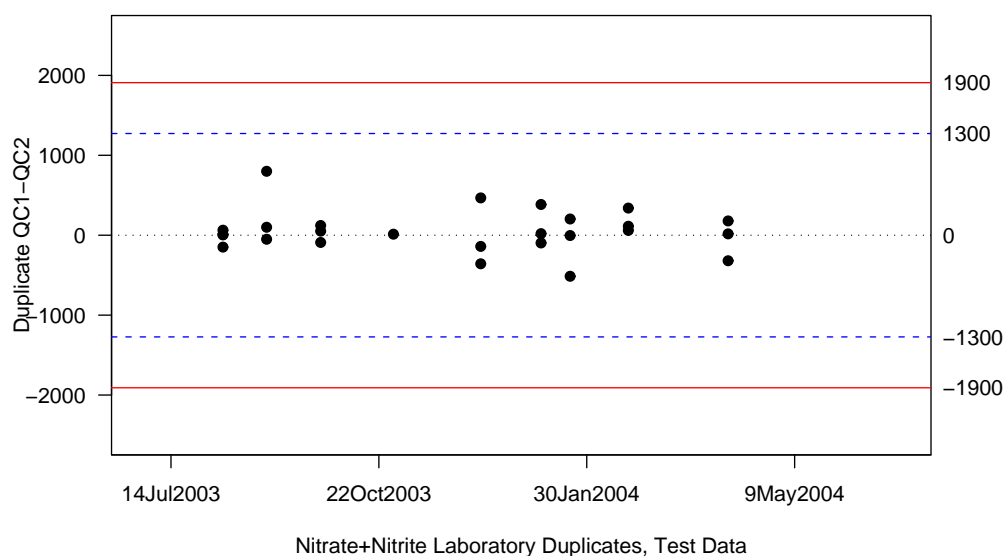
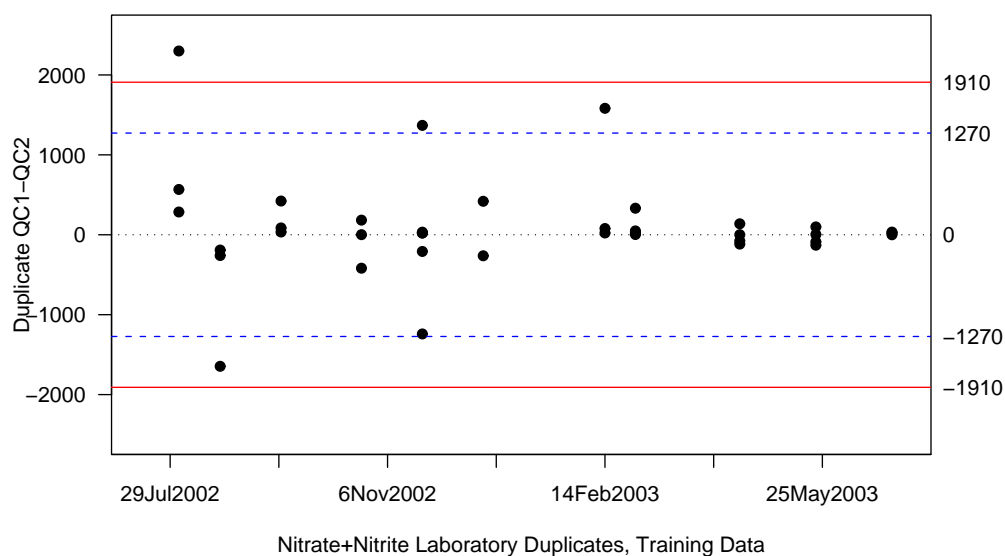


Figure 71: Nitrate+nitrite laboratory duplicates for the 2002–2004 Abbotsford-Sumas Monitoring Project. Upper/lower acceptance limits (± 2 std. dev. from mean pair difference) and upper/lower warning limits (± 3 std. dev. from mean pair difference) were calculated using data from July 2002 through June 2003.

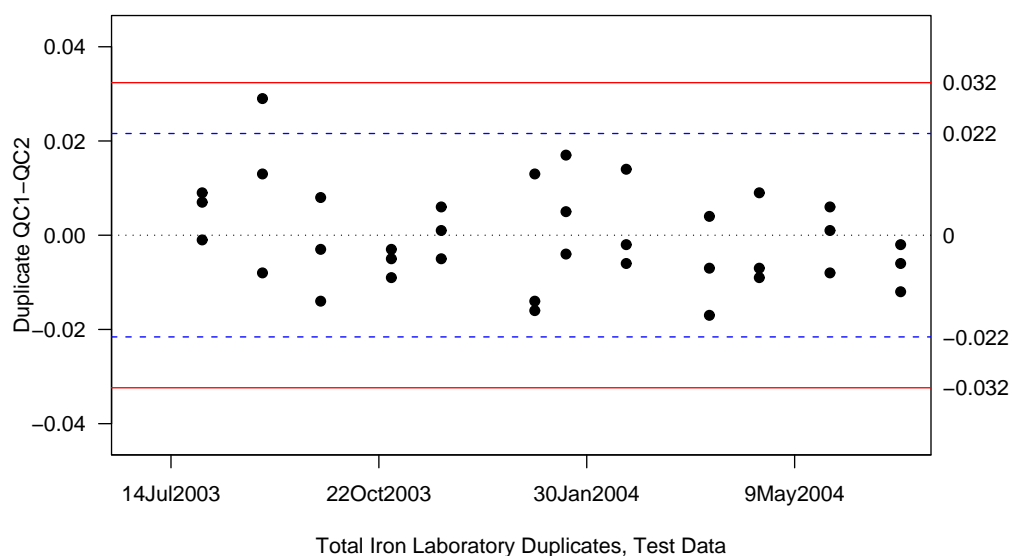
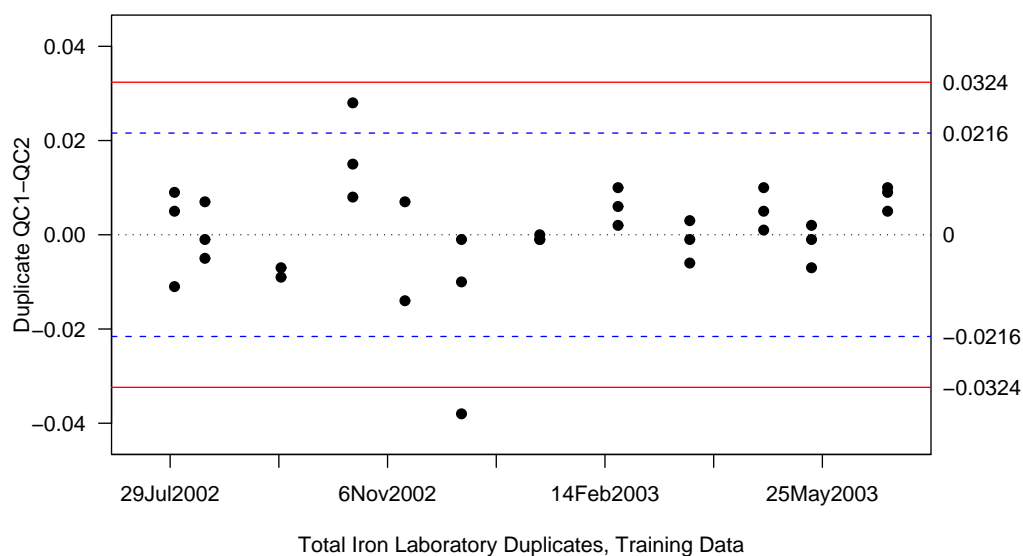


Figure 72: Total iron laboratory duplicates for the 2002–2004 Abbotsford-Sumas Monitoring Project. Upper/lower acceptance limits (± 2 std. dev. from mean pair difference) and upper/lower warning limits (± 3 std. dev. from mean pair difference) were calculated using data from July 2002 through June 2003.

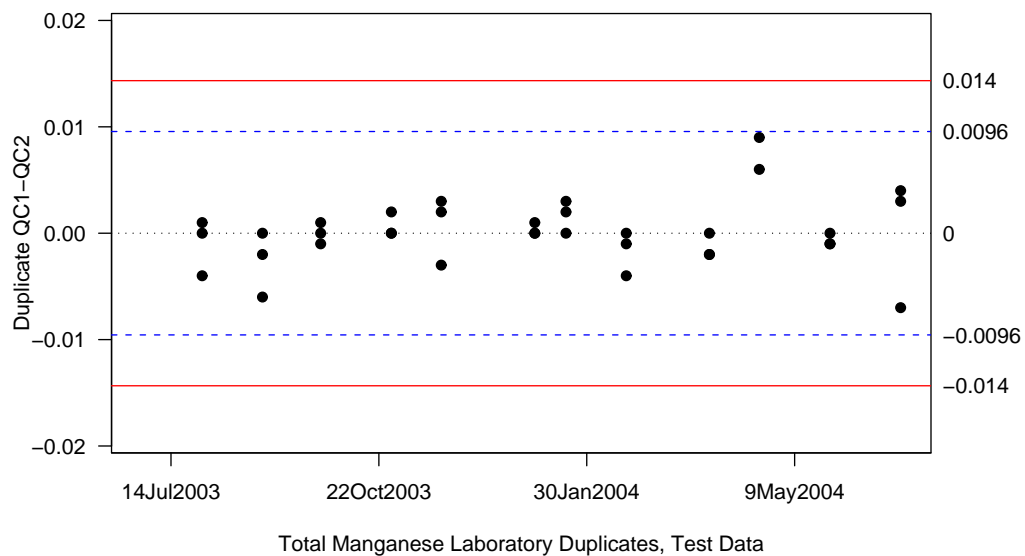
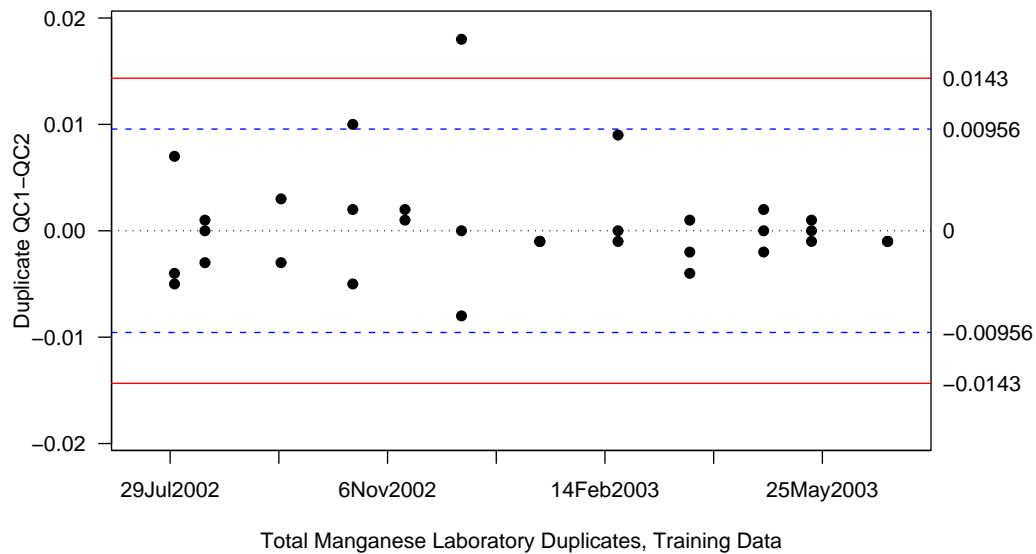


Figure 73: Total manganese laboratory duplicates for the 2002–2004 Abbotsford-Sumas Monitoring Project. Upper/lower acceptance limits (± 2 std. dev. from mean pair difference) and upper/lower warning limits (± 3 std. dev. from mean pair difference) were calculated using data from July 2002 through June 2003.

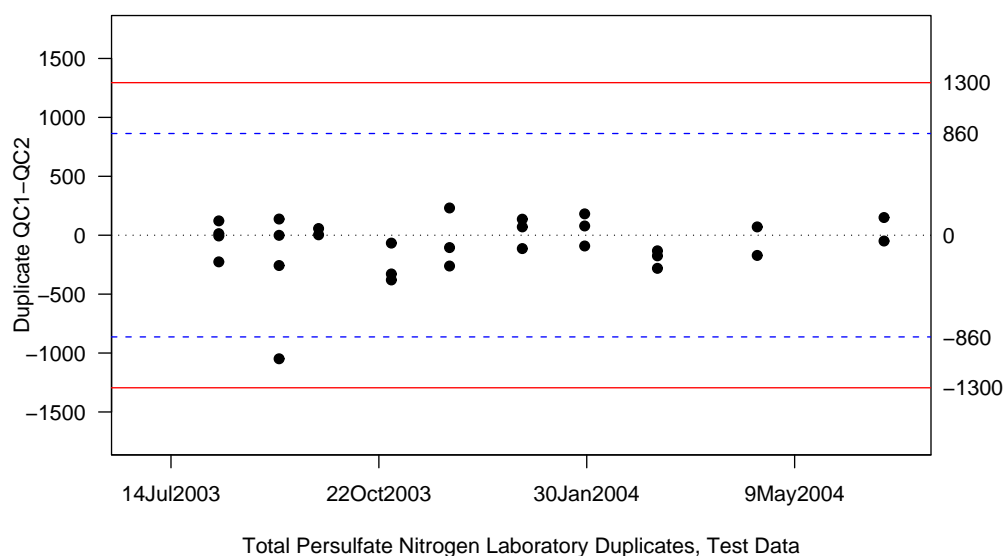
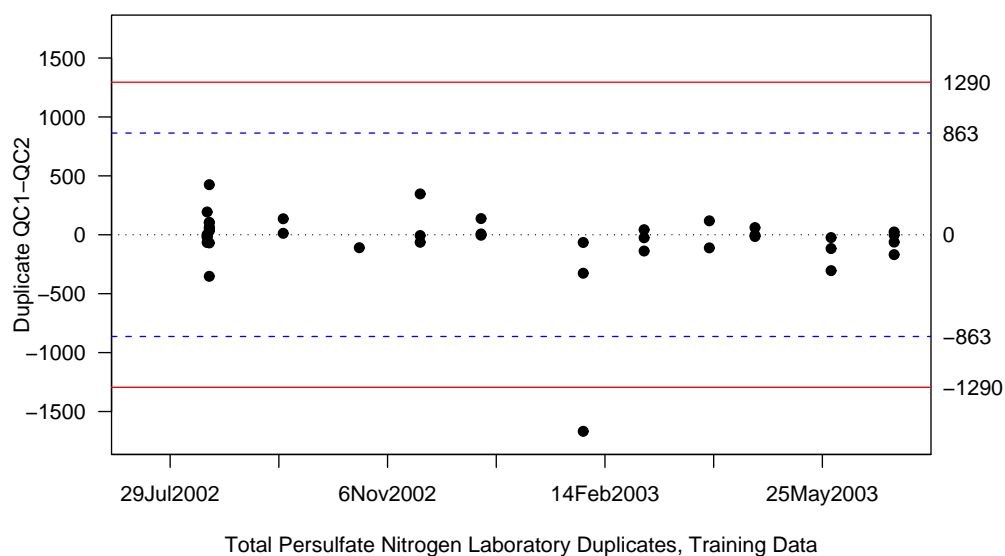


Figure 74: Total nitrogen laboratory duplicates for the 2002–2004 Abbotsford-Sumas Monitoring Project. Upper/lower acceptance limits (± 2 std. dev. from mean pair difference) and upper/lower warning limits (± 3 std. dev. from mean pair difference) were calculated using data from July 2002 through June 2003.

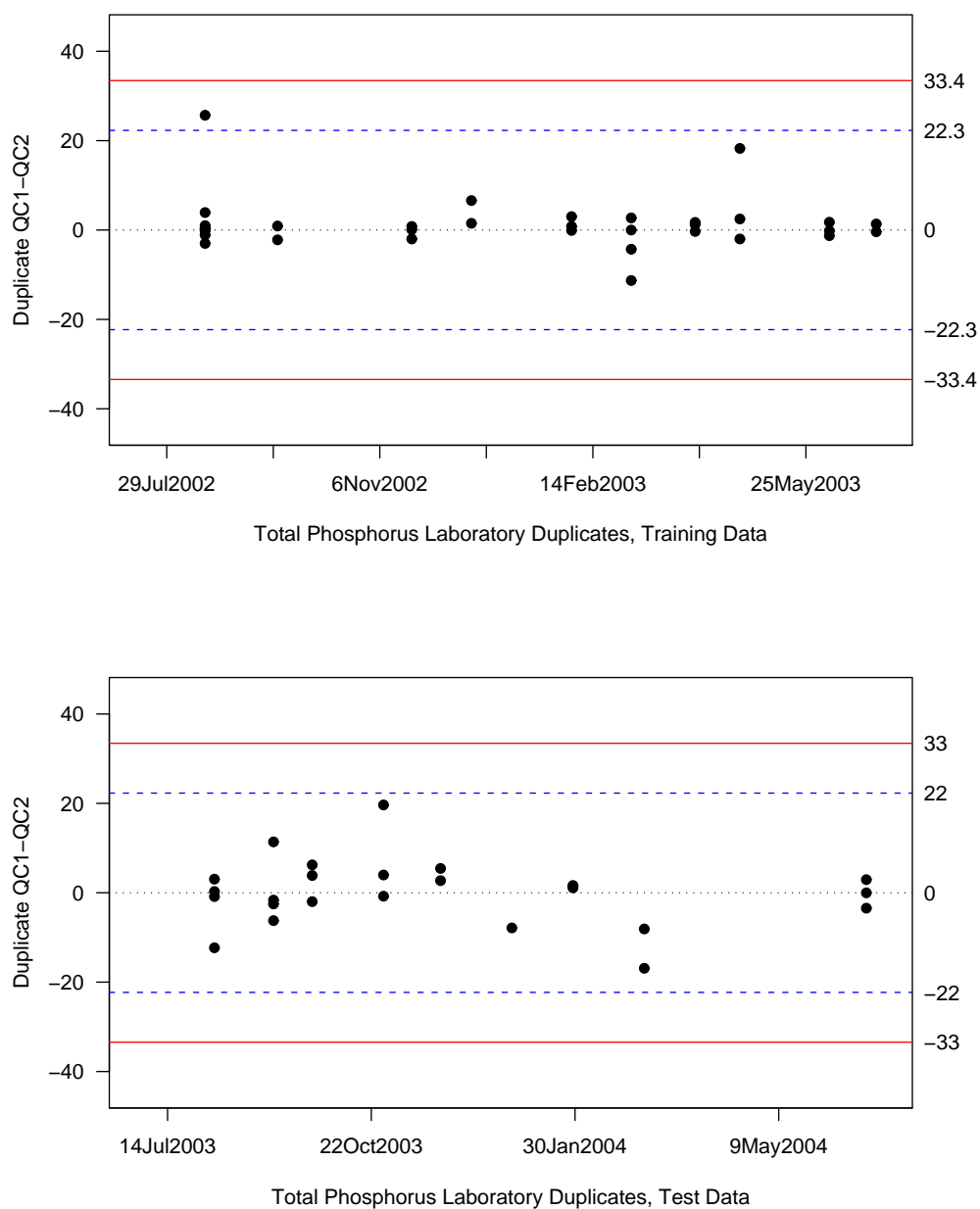


Figure 75: Total phosphorus laboratory duplicates for the 2002–2004 Abbotsford-Sumas Monitoring Project. Upper/lower acceptance limits (± 2 std. dev. from mean pair difference) and upper/lower warning limits (± 3 std. dev. from mean pair difference) were calculated using data from July 2002 through June 2003.

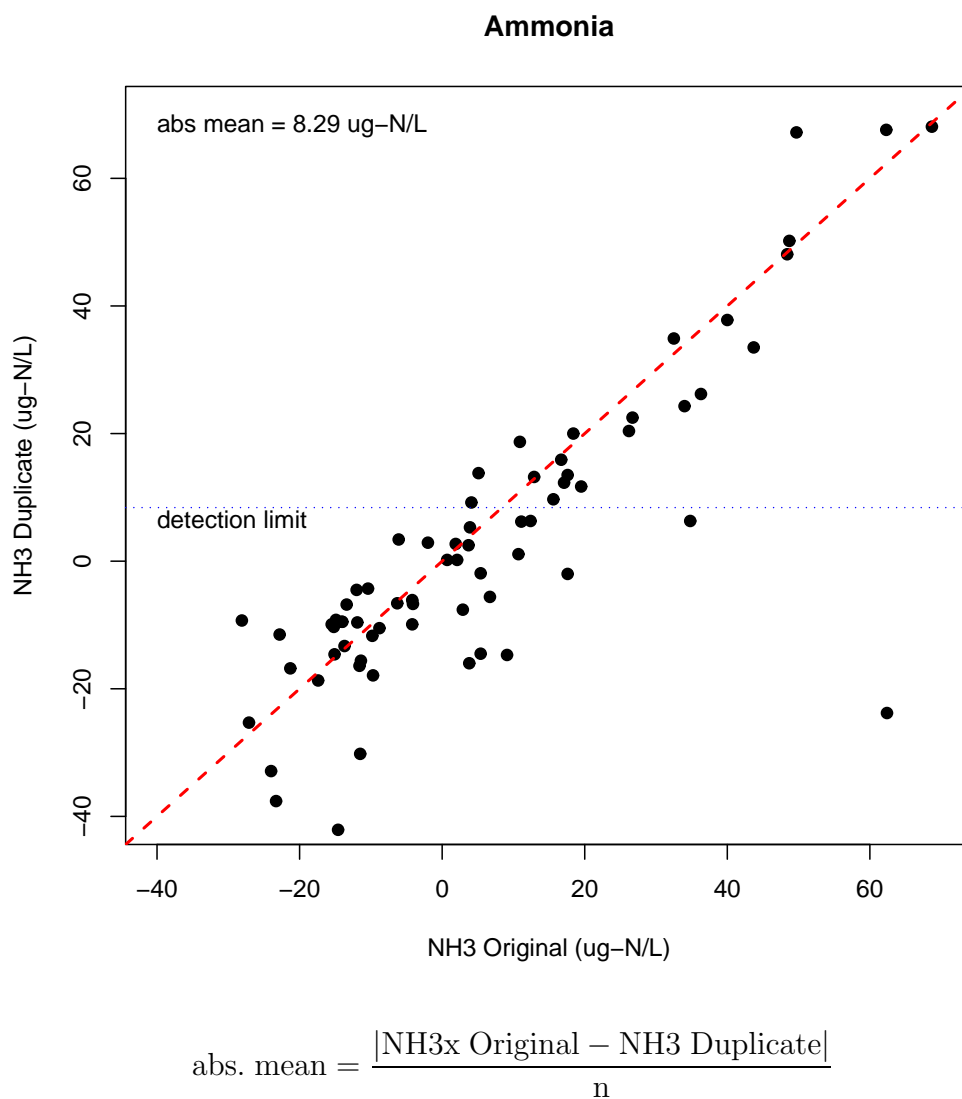


Figure 76: Ammonia field duplicates for the 2002–2004 Abbotsford-Sumas Monitoring Project. Diagonal reference line shows a 1:1 relationship.

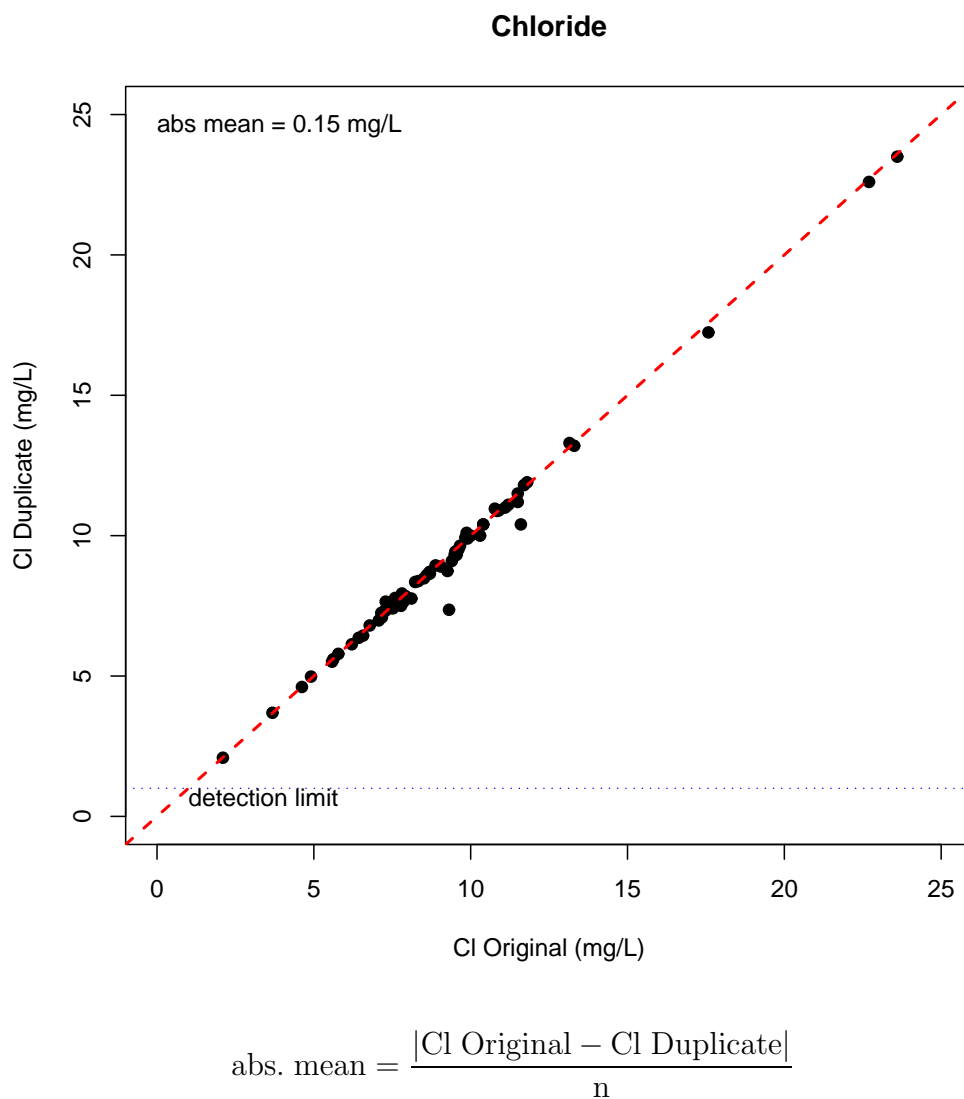


Figure 77: Chloride field duplicates for the 2002–2004 Abbotsford-Sumas Monitoring Project. Diagonal reference line shows a 1:1 relationship.

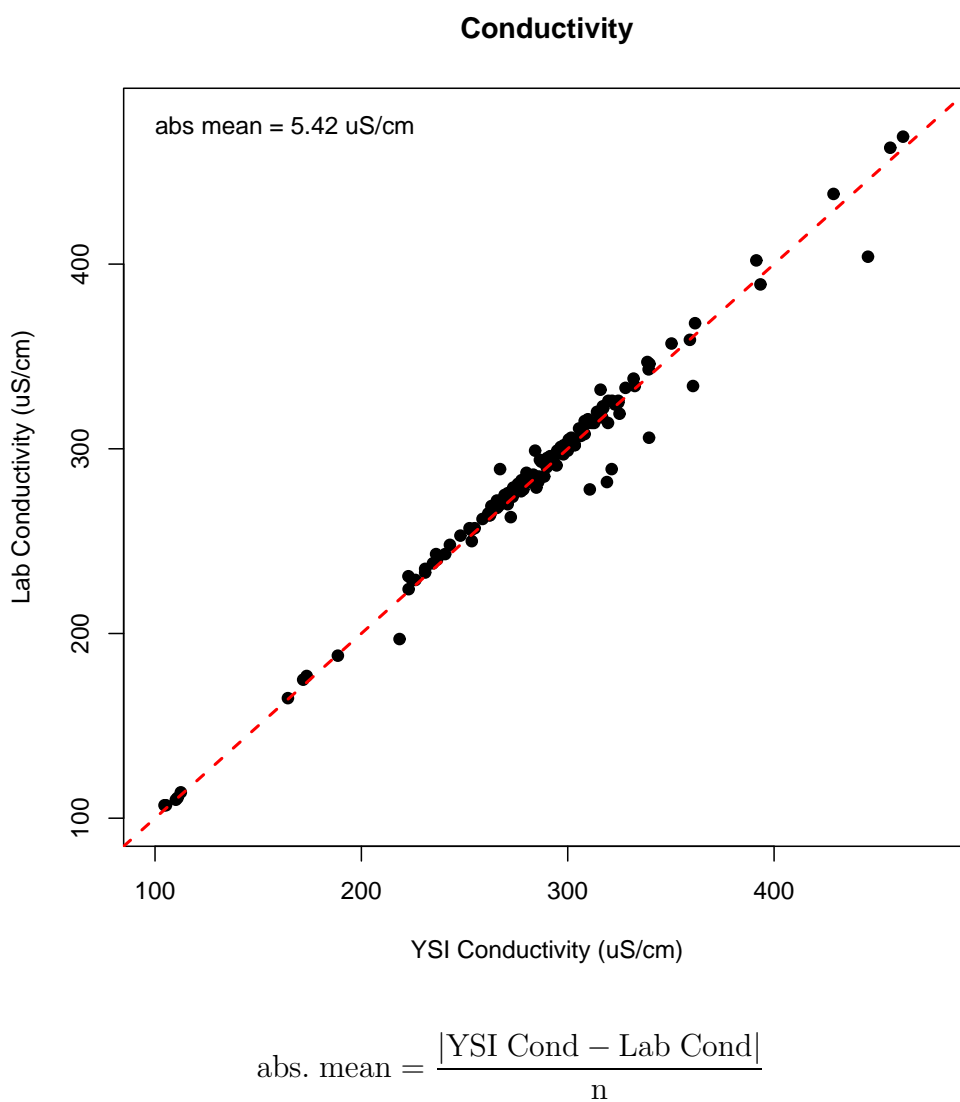


Figure 78: Comparison between field meter (YSI) and laboratory conductivity duplicate samples for the 2002–2004 Abbotsford-Sumas Monitoring Project. Diagonal reference line shows a 1:1 relationship.

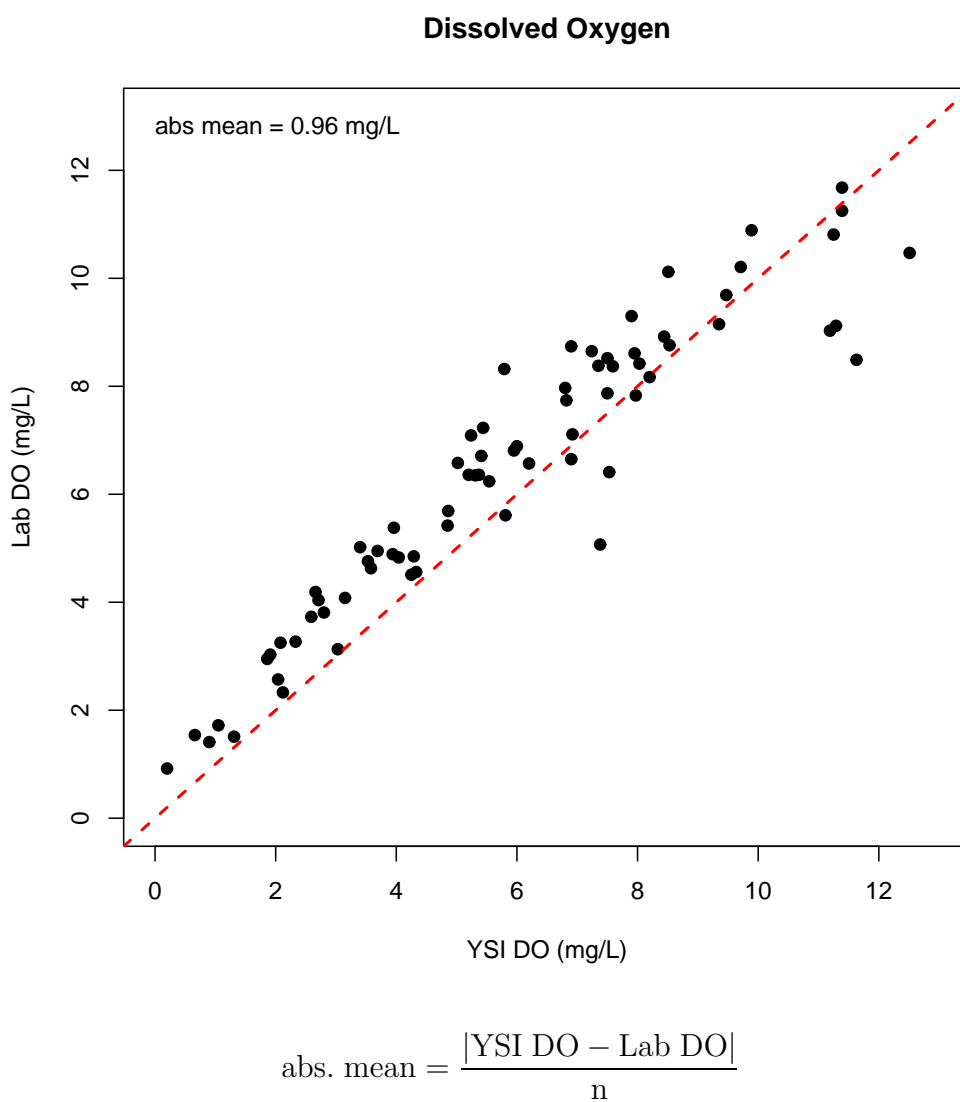


Figure 79: Comparison between field meter (YSI) and laboratory dissolved oxygen duplicate samples for the 2002–2004 Abbotsford-Sumas Monitoring Project. Diagonal reference line shows a 1:1 relationship.

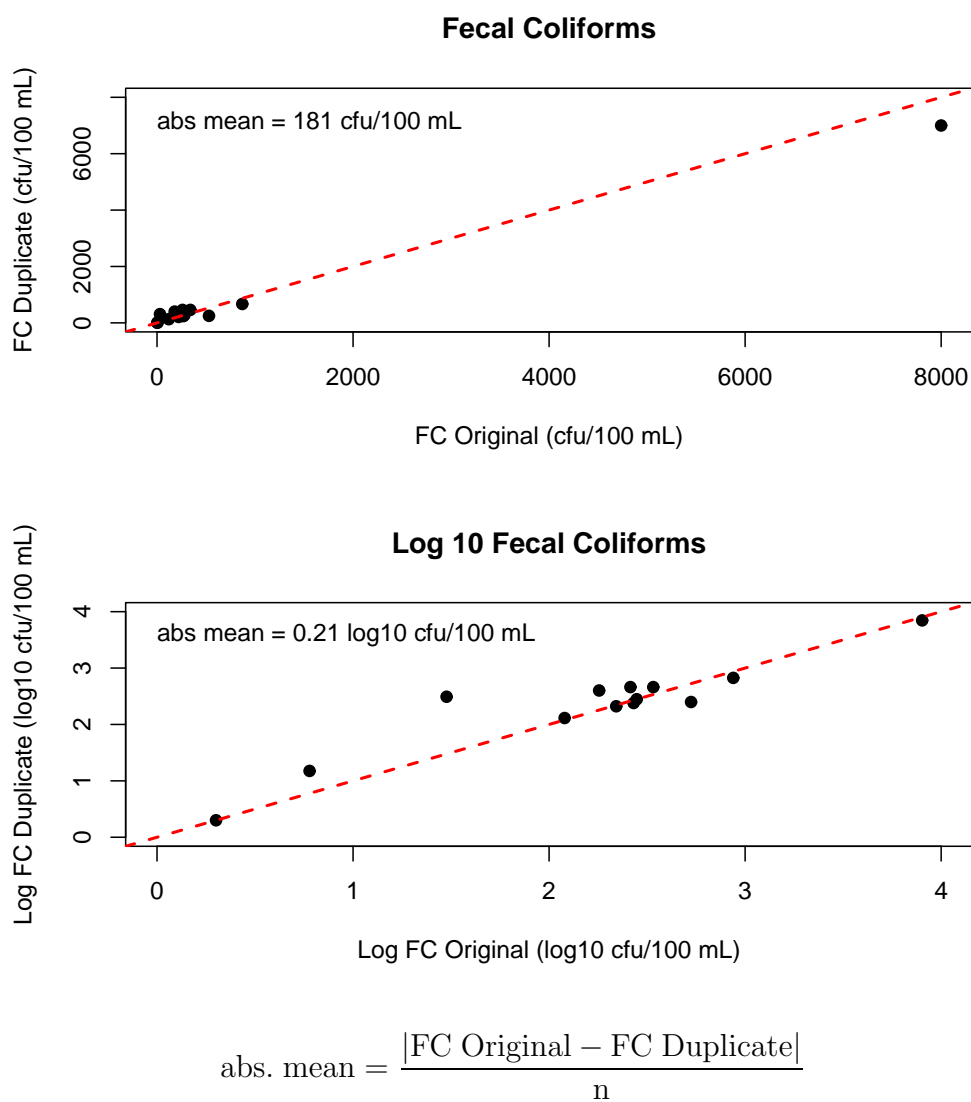


Figure 80: Fecal coliforms field duplicates for the 2002–2004 Abbotsford-Sumas Monitoring Project. Diagonal reference line shows a 1:1 relationship. Results are shown for raw counts and log10 transformed counts.

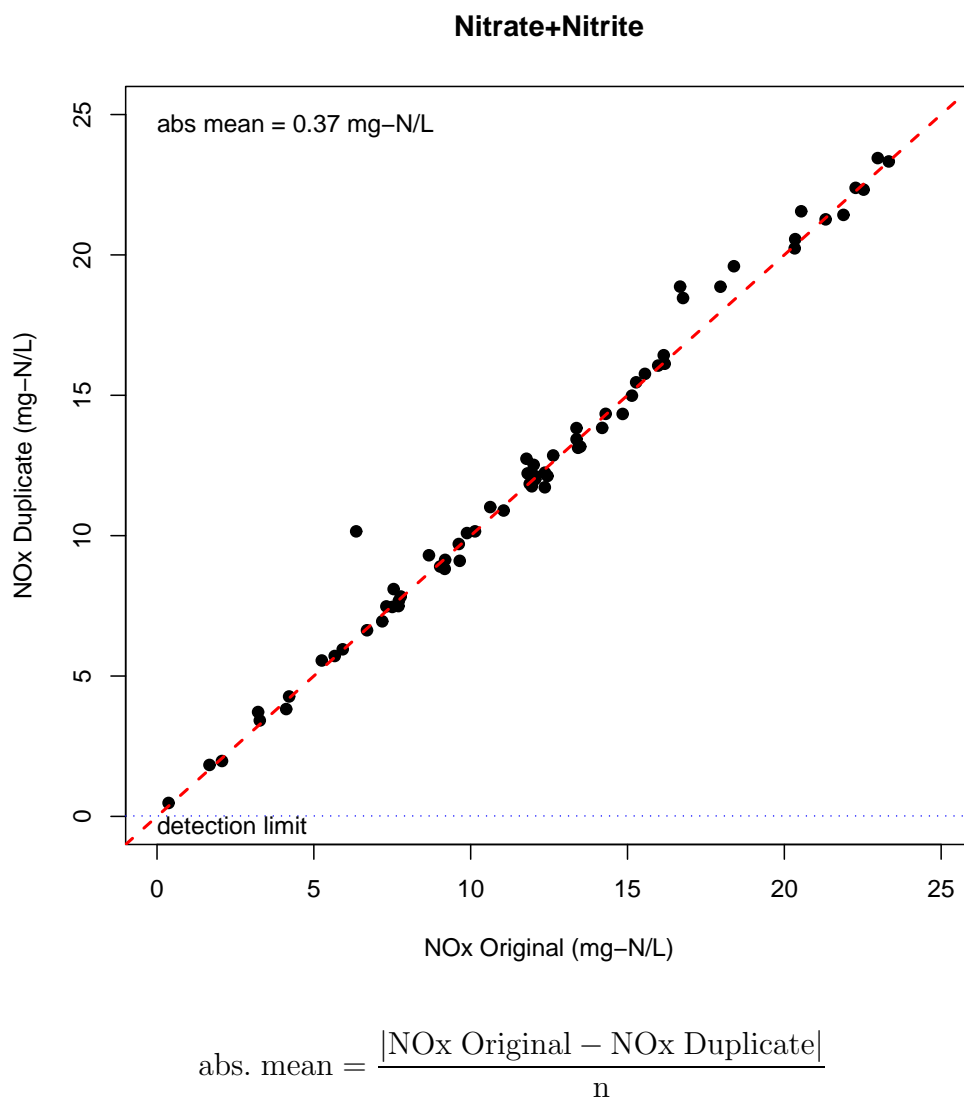
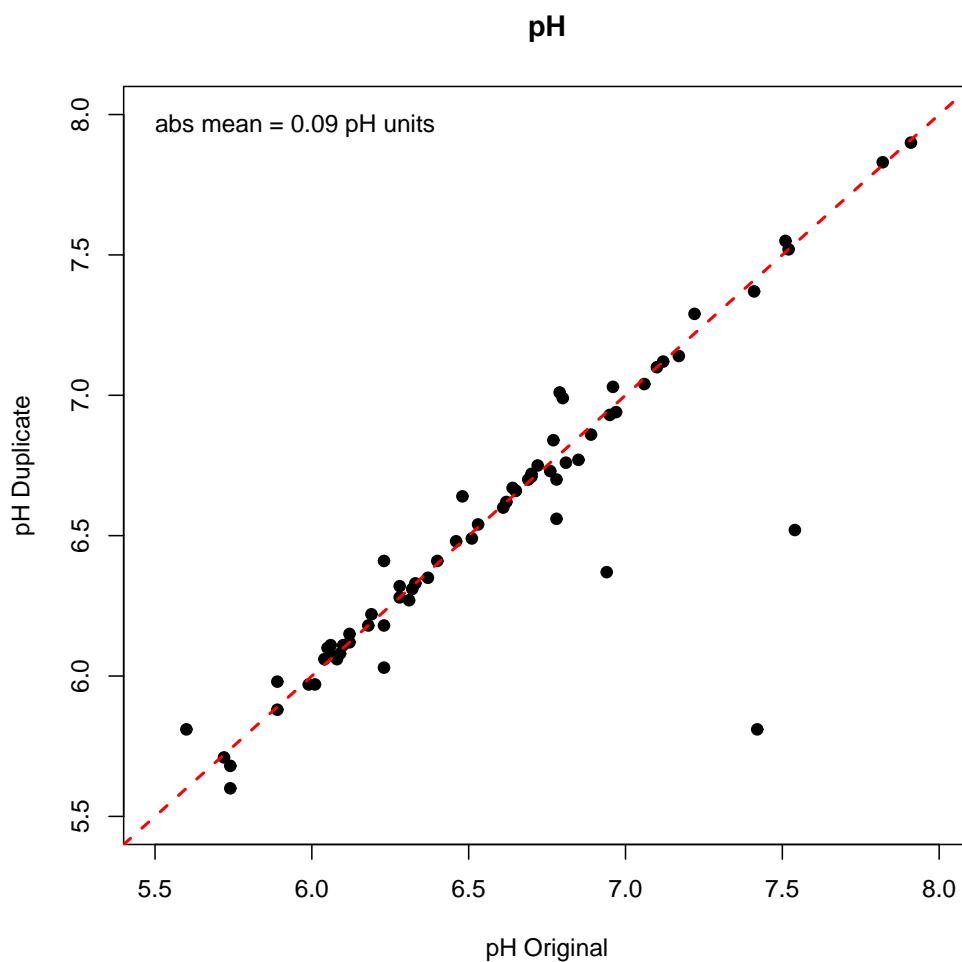


Figure 81: Nitrate+nitrite field duplicates for the 2002–2004 Abbotsford-Sumas Monitoring Project. Diagonal reference line shows a 1:1 relationship.



$$\text{abs. mean} = \frac{|\text{pH Original} - \text{pH Duplicate}|}{n}$$

Figure 82: pH field duplicates for the 2002–2004 Abbotsford-Sumas Monitoring Project. Diagonal reference line shows a 1:1 relationship.

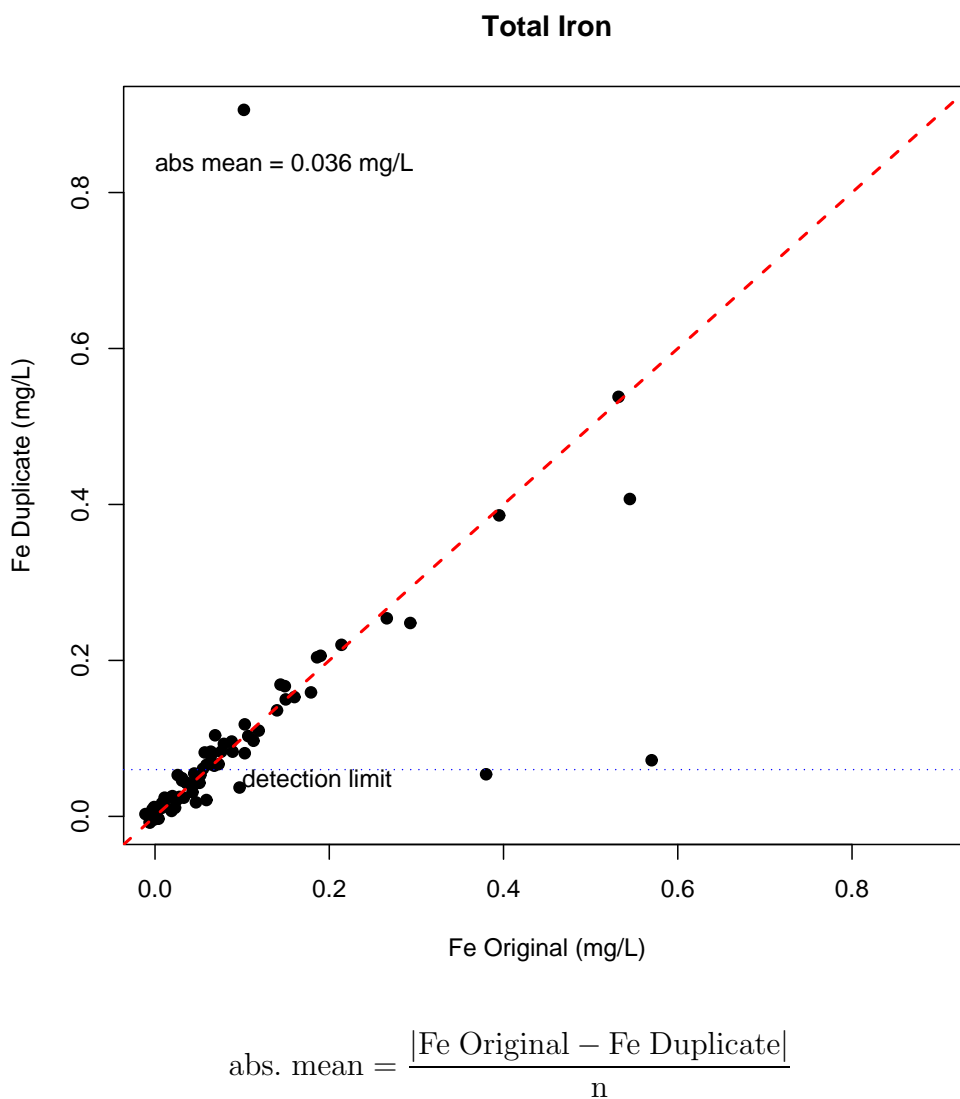


Figure 83: Total iron field duplicates for the 2002–2004 Abbotsford-Sumas Monitoring Project. Diagonal reference line shows a 1:1 relationship.

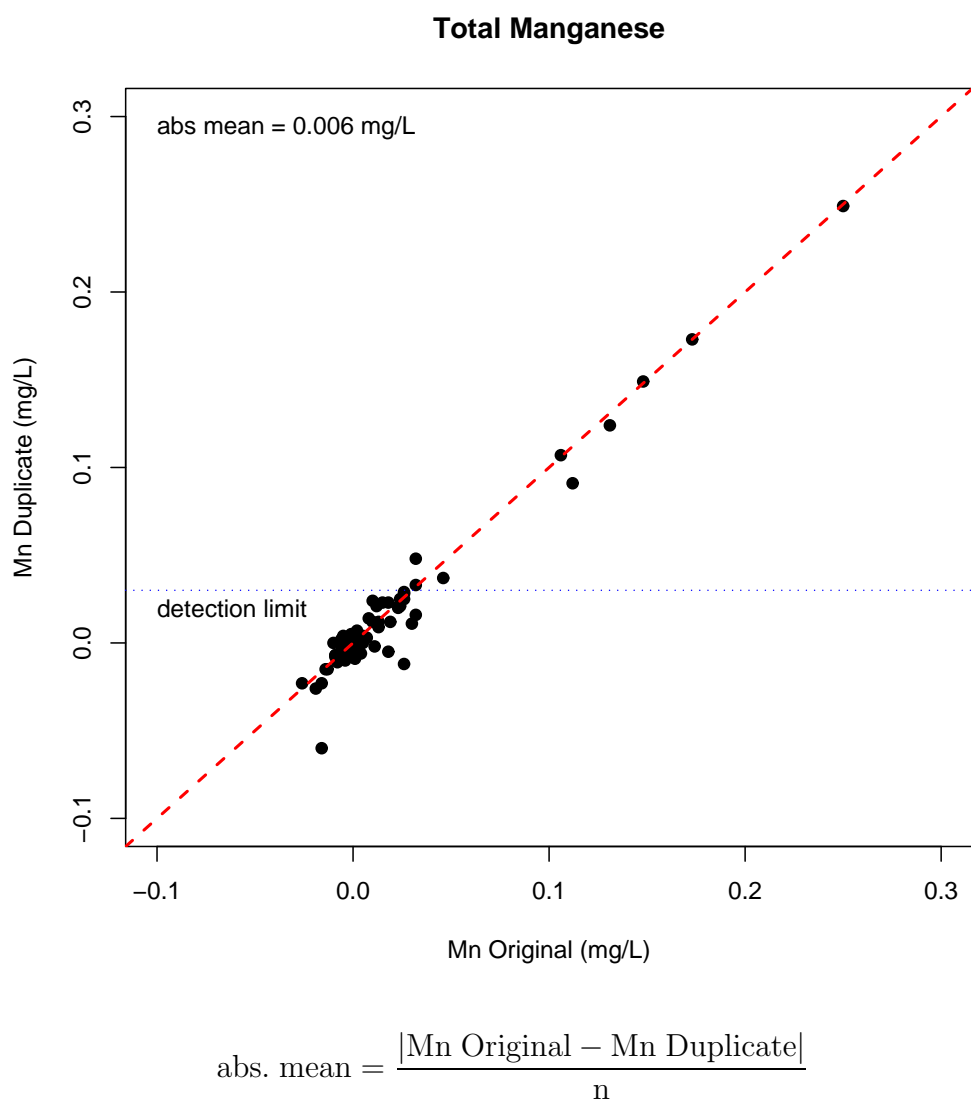


Figure 84: Total manganese field duplicates for the 2002–2004 Abbotsford-Sumas Monitoring Project. Diagonal reference line shows a 1:1 relationship.

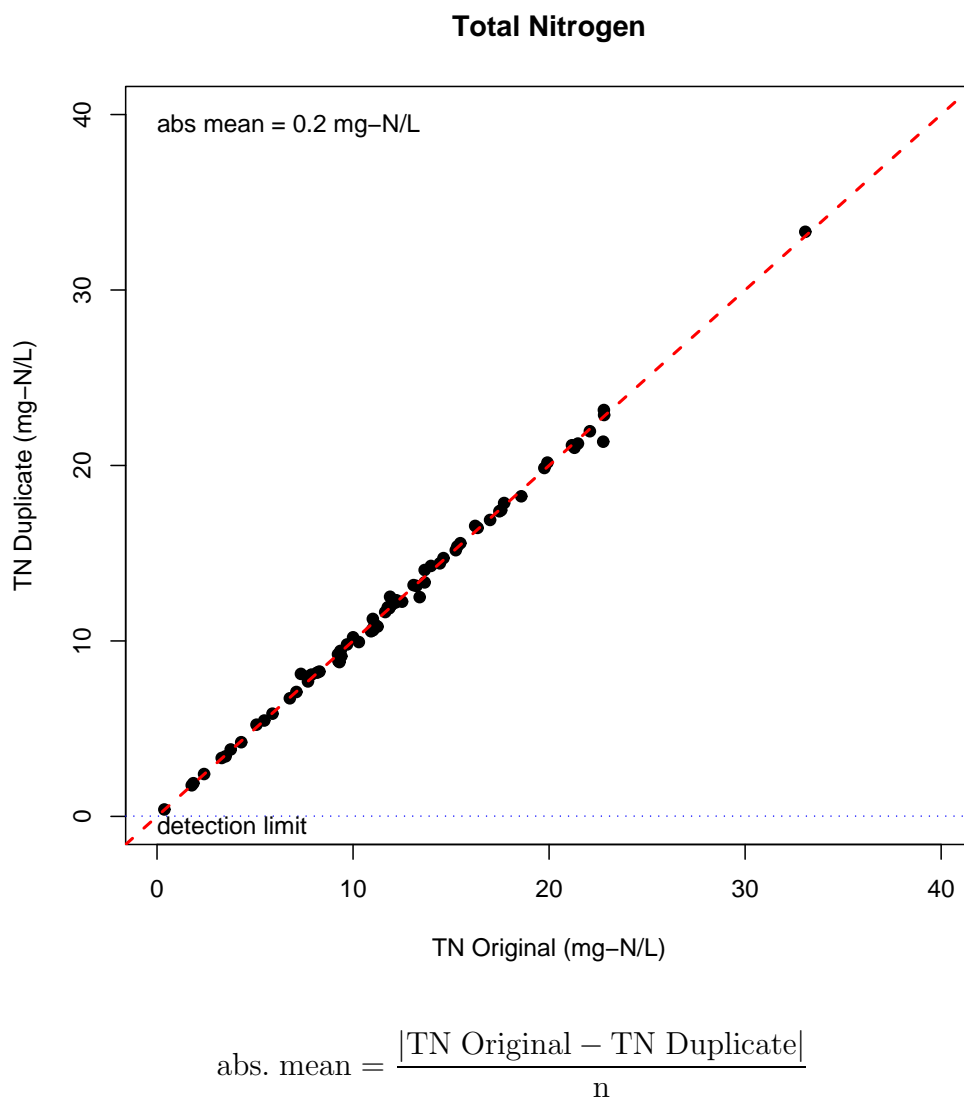


Figure 85: Total nitrogen field duplicates for the 2002–2004 Abbotsford-Sumas Monitoring Project. Diagonal reference line shows a 1:1 relationship.

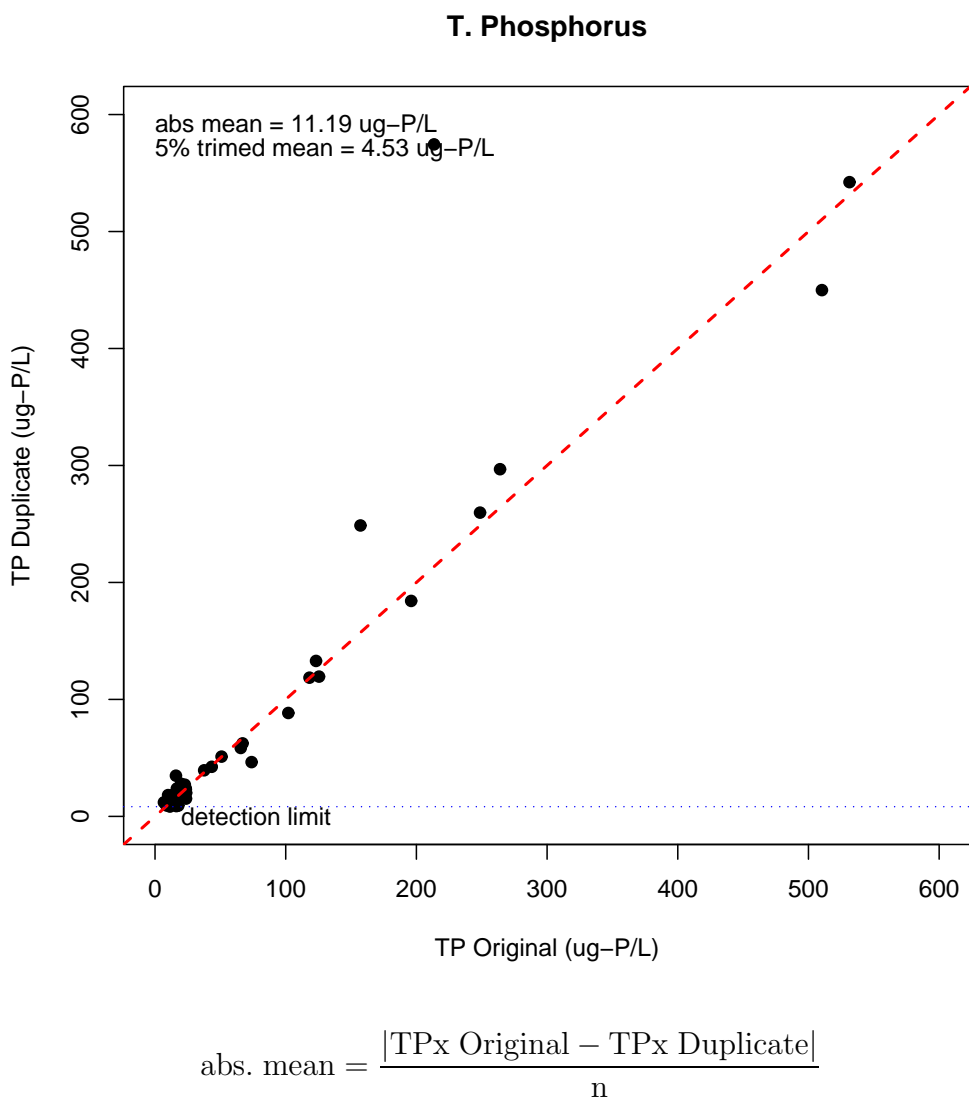


Figure 86: Total phosphorus field duplicates for the 2002–2004 Abbotsford-Sumas Monitoring Project. Diagonal reference line shows a 1:1 relationship.

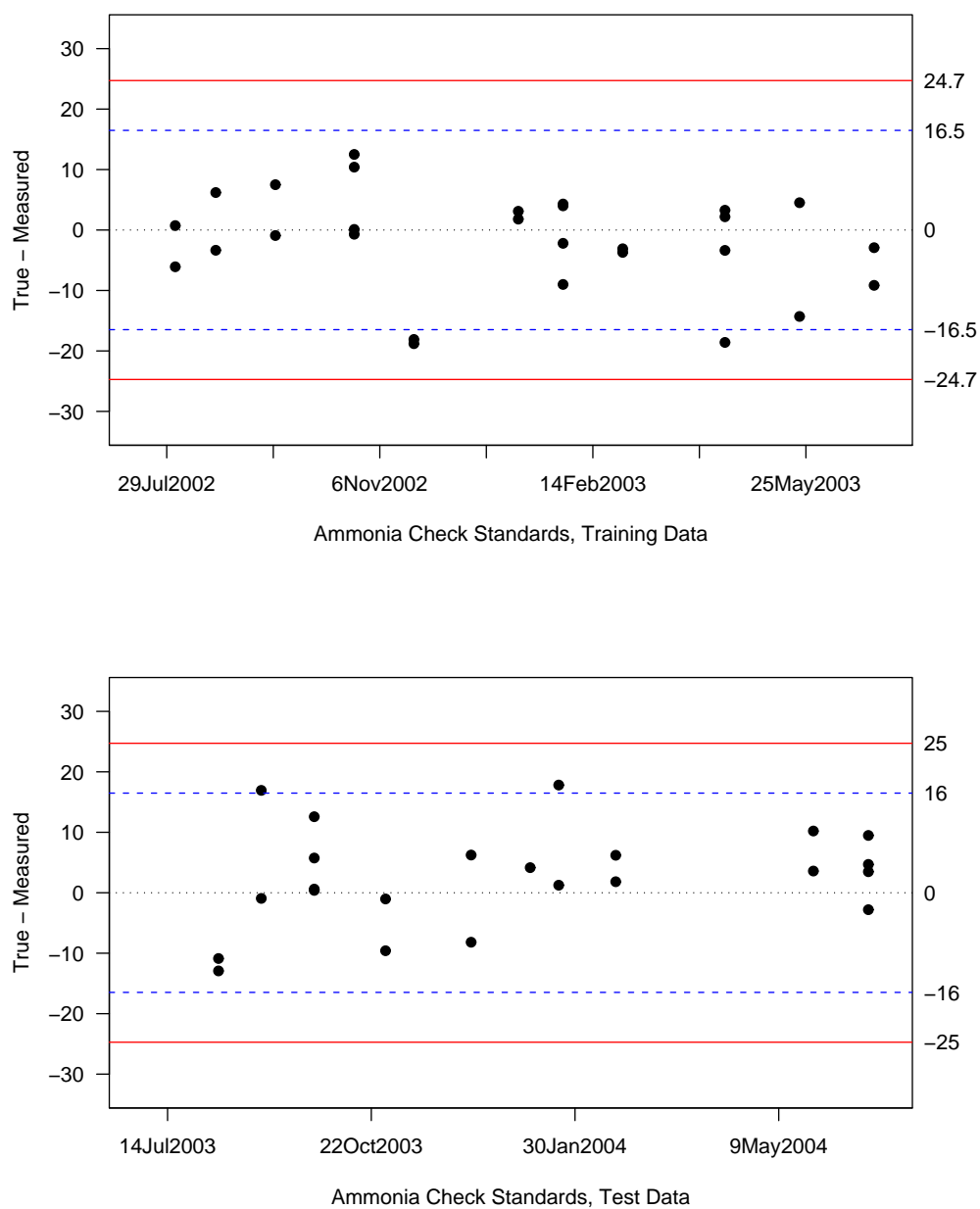


Figure 87: Ammonia check standards for the 2002–2004 Abbotsford-Sumas Monitoring Project. Upper/lower acceptance limits (± 2 std. dev. from mean pair difference) and upper/lower warning limits (± 3 std. dev. from mean pair difference) were calculated using data from July 2002 through June 2003.

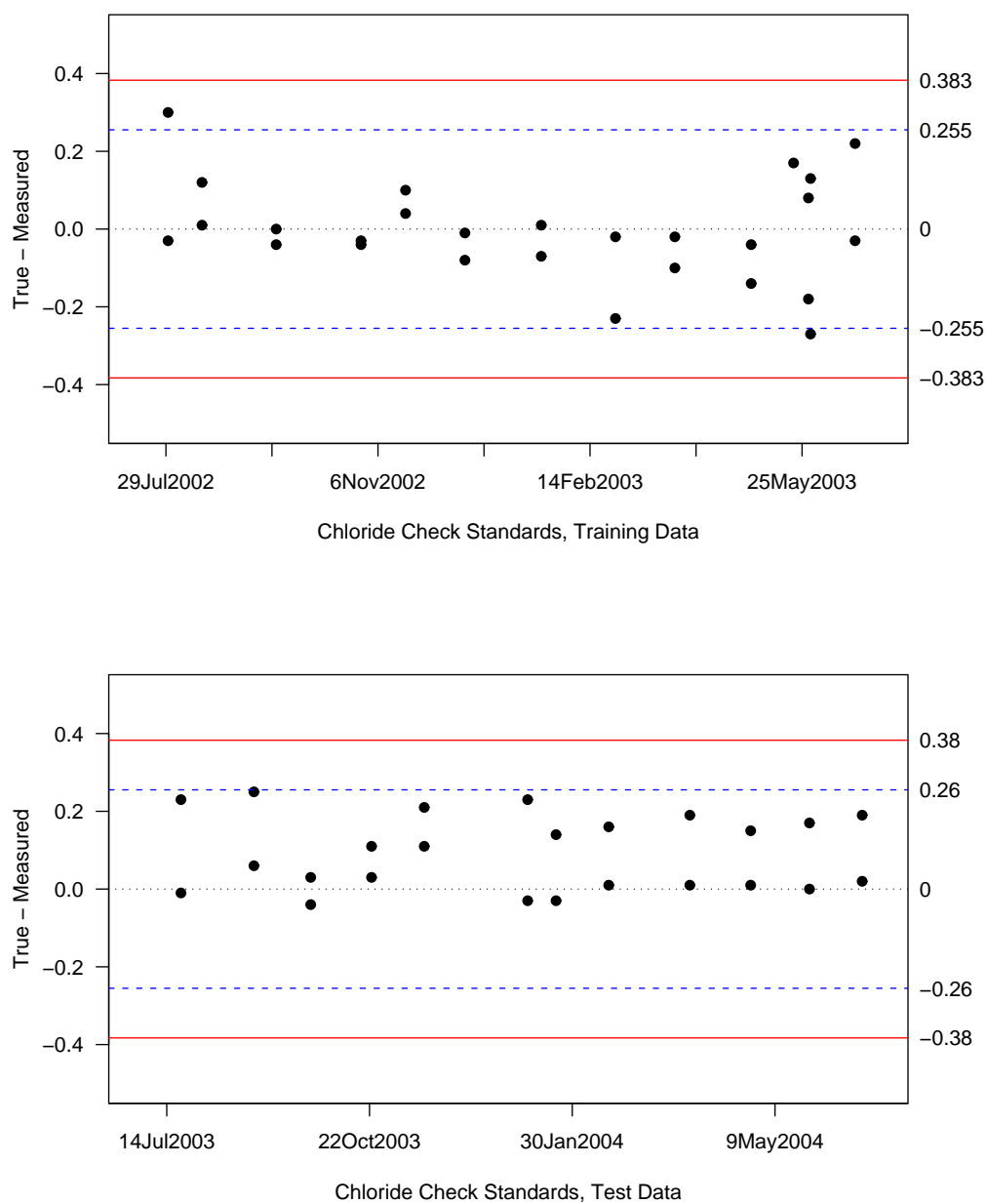


Figure 88: Chloride check standards for the 2002–2004 Abbotsford-Sumas Monitoring Project. Upper/lower acceptance limits (± 2 std. dev. from mean pair difference) and upper/lower warning limits (± 3 std. dev. from mean pair difference) were calculated using data from July 2002 through June 2003.

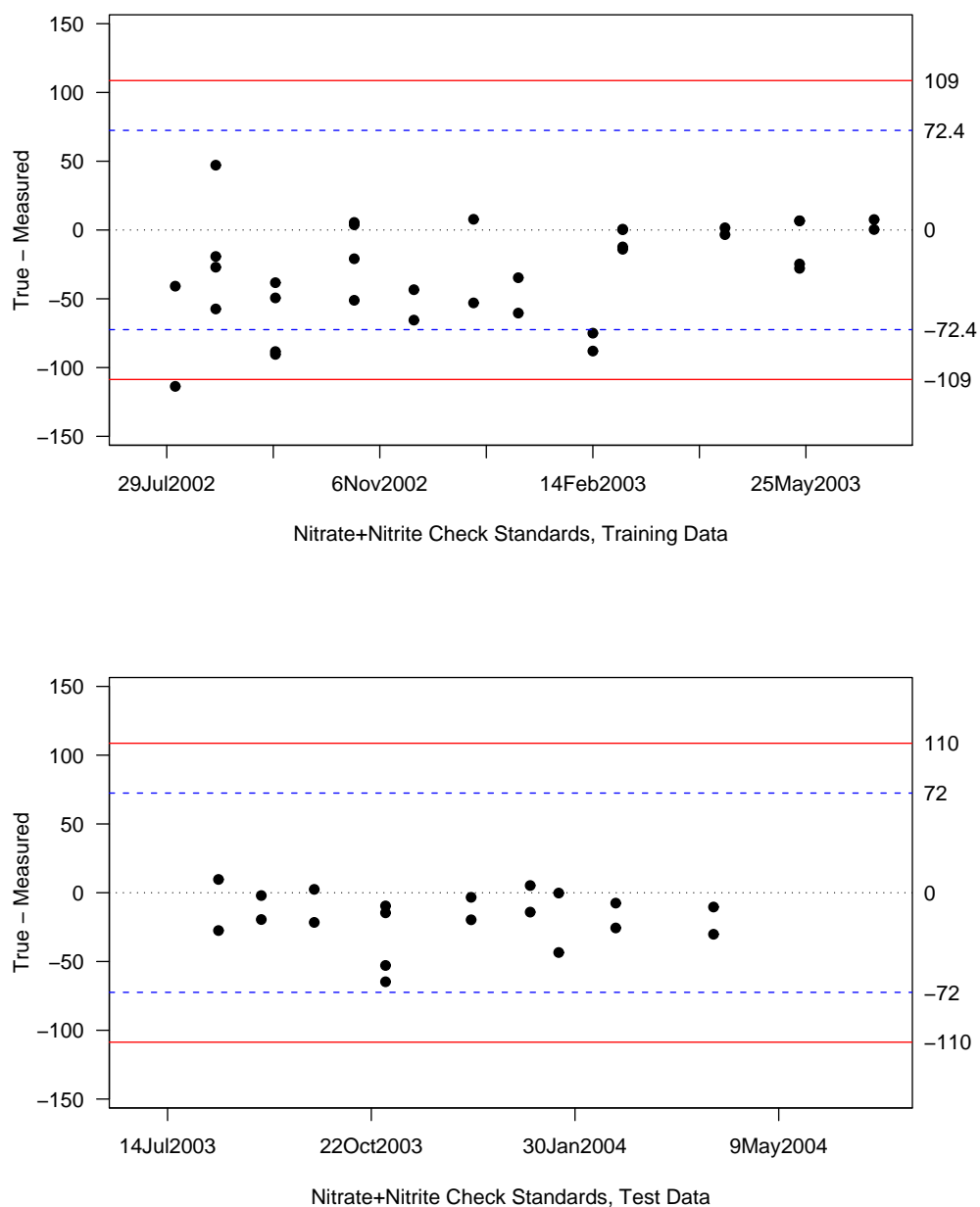


Figure 89: Nitrate+nitrite check standards for the 2002–2004 Abbotsford-Sumas Monitoring Project. Upper/lower acceptance limits (± 2 std. dev. from mean pair difference) and upper/lower warning limits (± 3 std. dev. from mean pair difference) were calculated using data from July 2002 through June 2003.

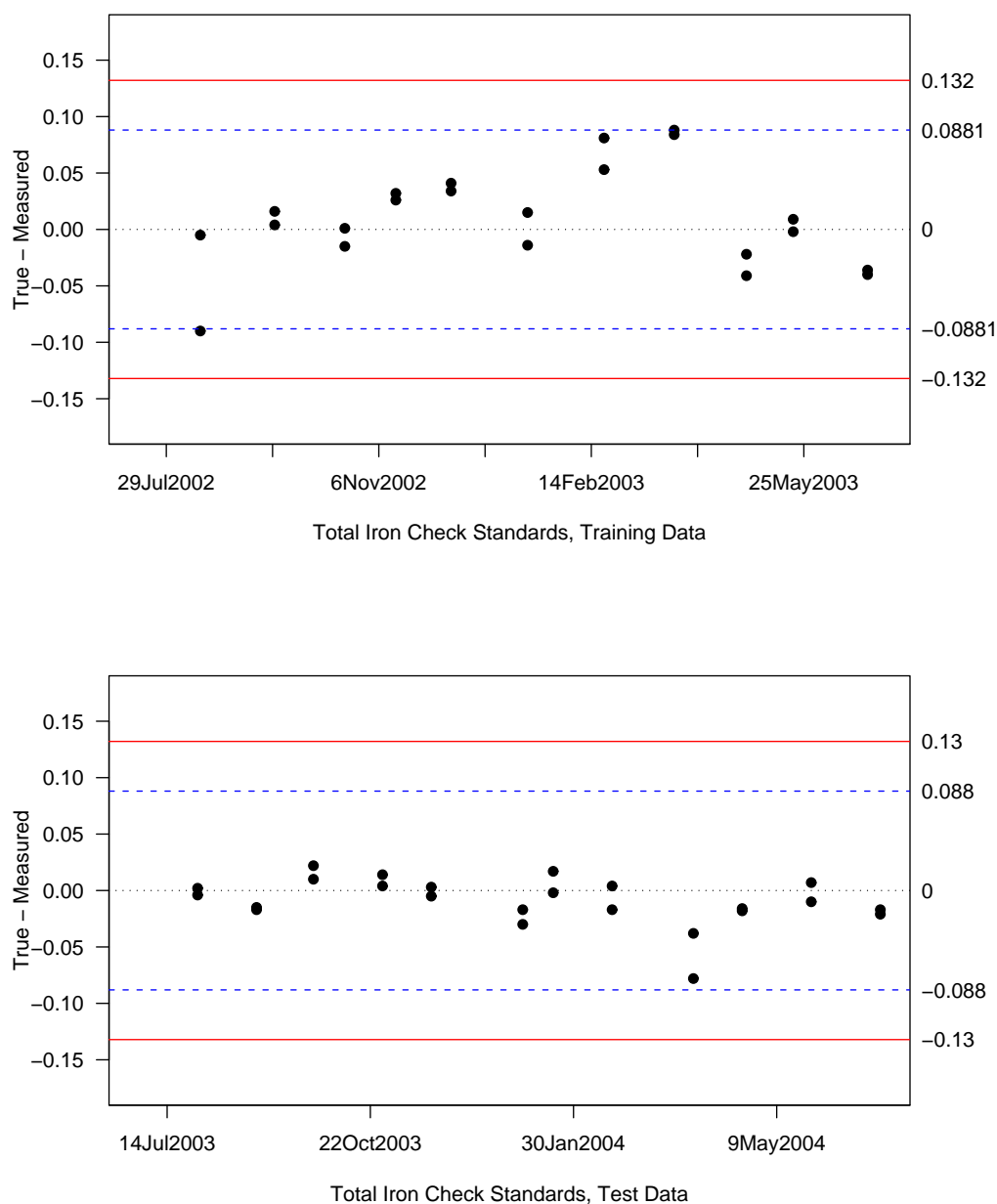


Figure 90: Total iron check standards for the 2002–2004 Abbotsford-Sumas Monitoring Project. Upper/lower acceptance limits (± 2 std. dev. from mean pair difference) and upper/lower warning limits (± 3 std. dev. from mean pair difference) were calculated using data from July 2002 through June 2003.

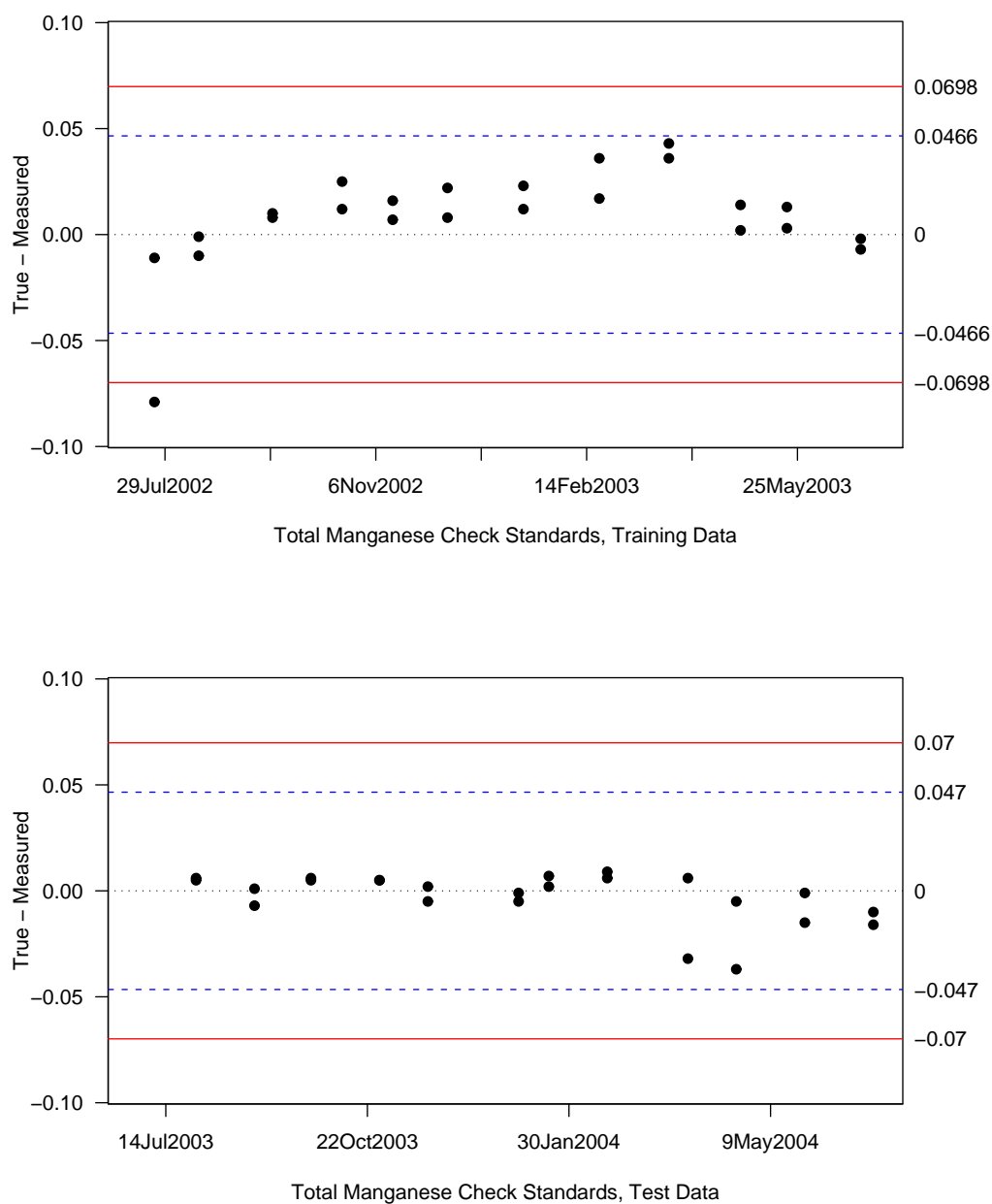


Figure 91: Total manganese check standards for the 2002–2004 Abbotsford-Sumas Monitoring Project. Upper/lower acceptance limits (± 2 std. dev. from mean pair difference) and upper/lower warning limits (± 3 std. dev. from mean pair difference) were calculated using data from July 2002 through June 2003.

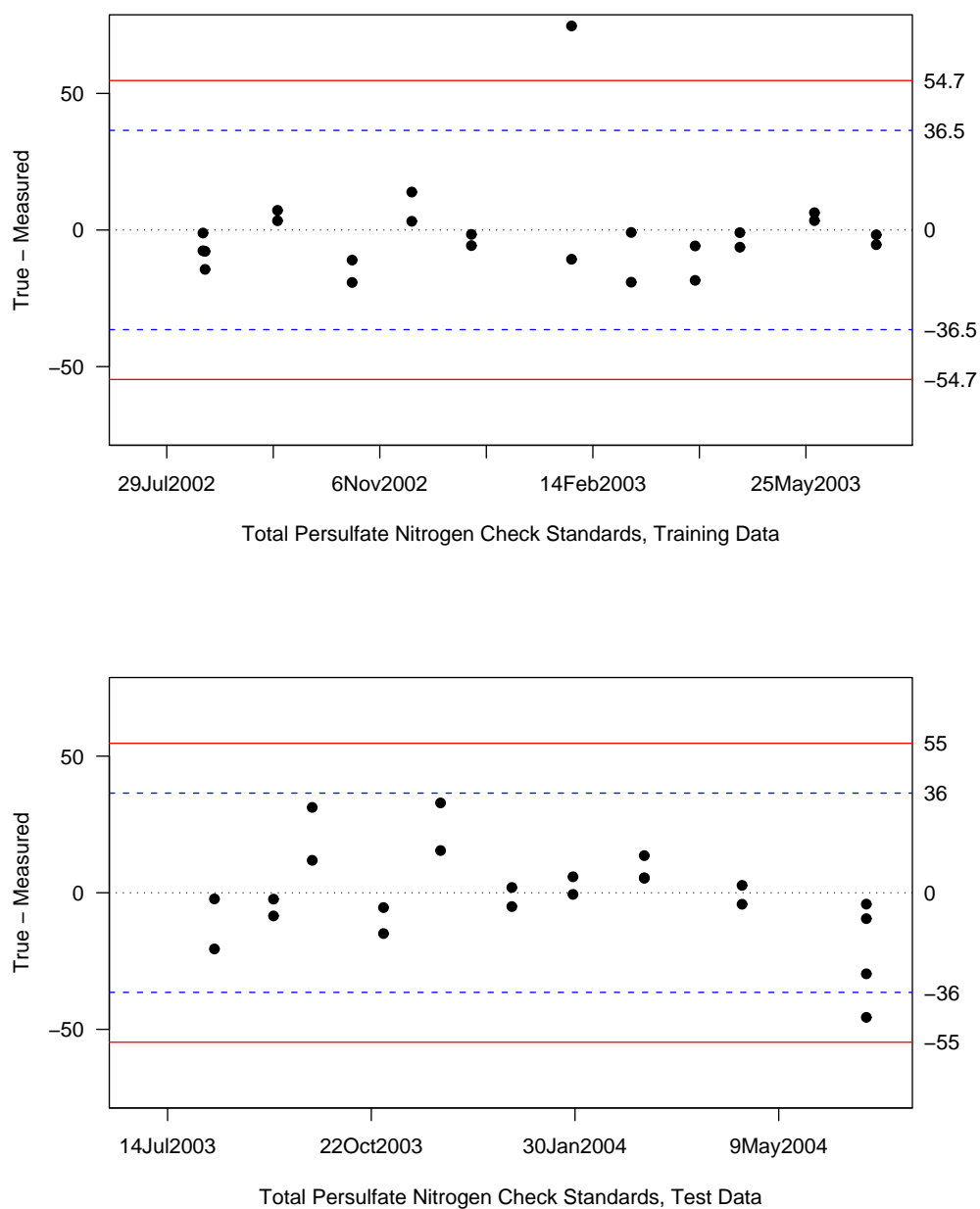


Figure 92: Total nitrogen check standards for the 2002–2004 Abbotsford-Sumas Monitoring Project. Upper/lower acceptance limits (± 2 std. dev. from mean pair difference) and upper/lower warning limits (± 3 std. dev. from mean pair difference) were calculated using data from July 2002 through June 2003.

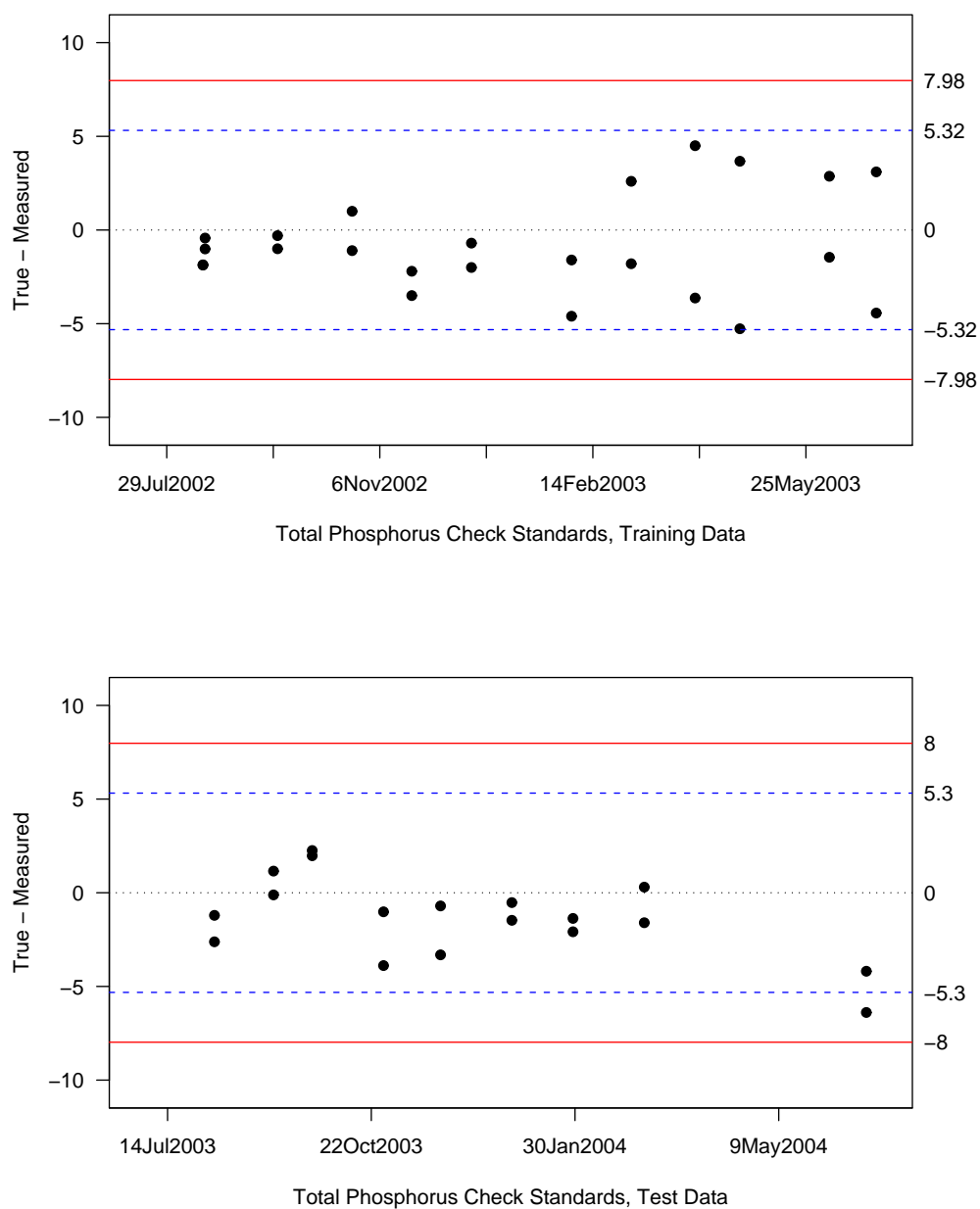


Figure 93: Total phosphorus check standards for the 2002–2004 Abbotsford-Sumas Monitoring Project. Upper/lower acceptance limits (± 2 std. dev. from mean pair difference) and upper/lower warning limits (± 3 std. dev. from mean pair difference) were calculated using data from July 2002 through June 2003.

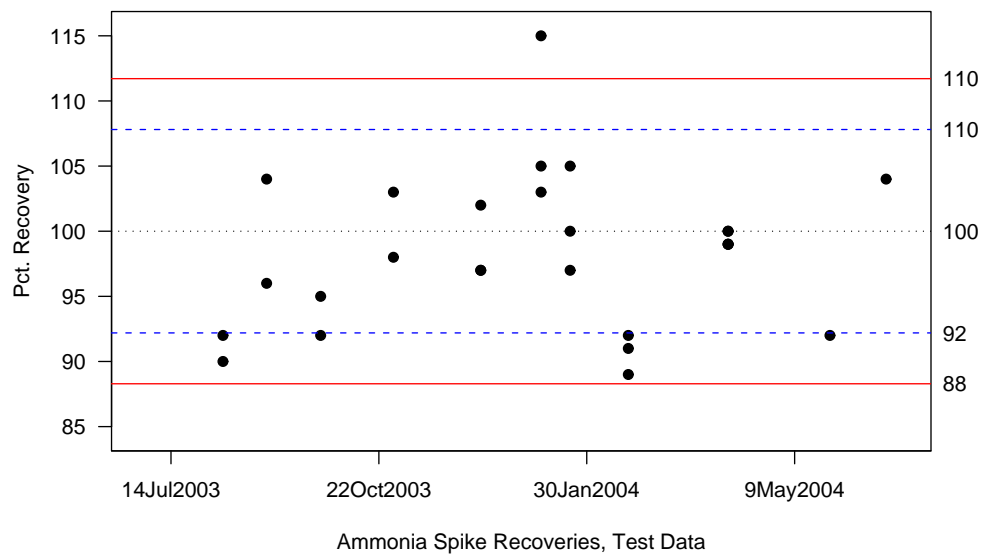
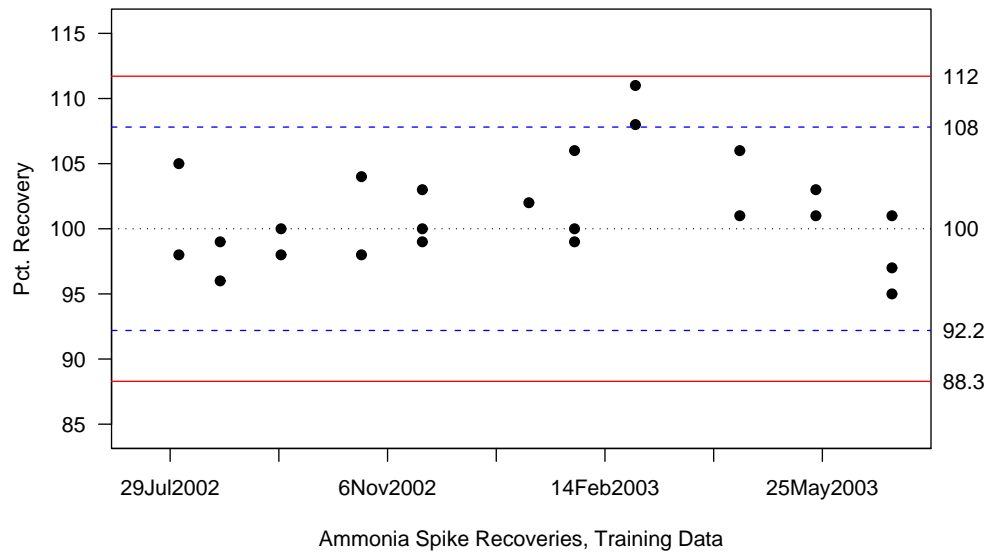


Figure 94: Ammonia spike recoveries for the 2002–2004 Abbotsford-Sumas Monitoring Project. Upper/lower acceptance limits (± 2 std. dev. from mean pair difference) and upper/lower warning limits (± 3 std. dev. from mean pair difference) were calculated using data from July 2002 through June 2003.

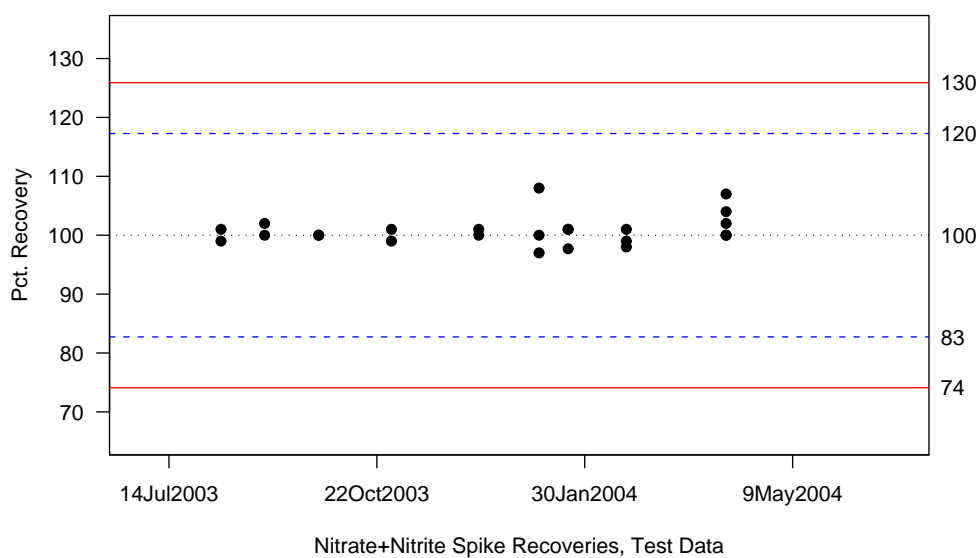
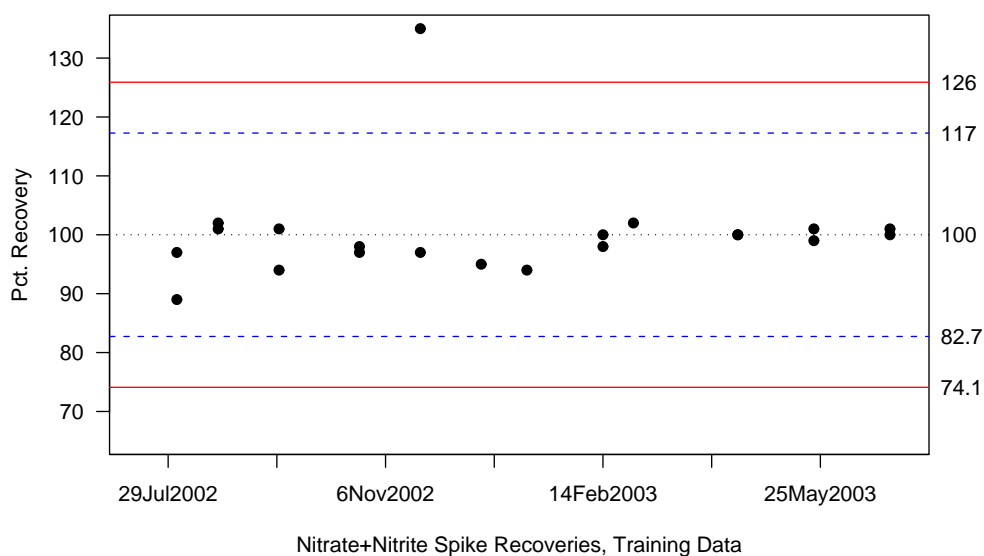


Figure 95: Nitrate+nitrite spike recoveries for the 2002–2004 Abbotsford-Sumas Monitoring Project. Upper/lower acceptance limits (± 2 std. dev. from mean pair difference) and upper/lower warning limits (± 3 std. dev. from mean pair difference) were calculated using data from July 2002 through June 2003.

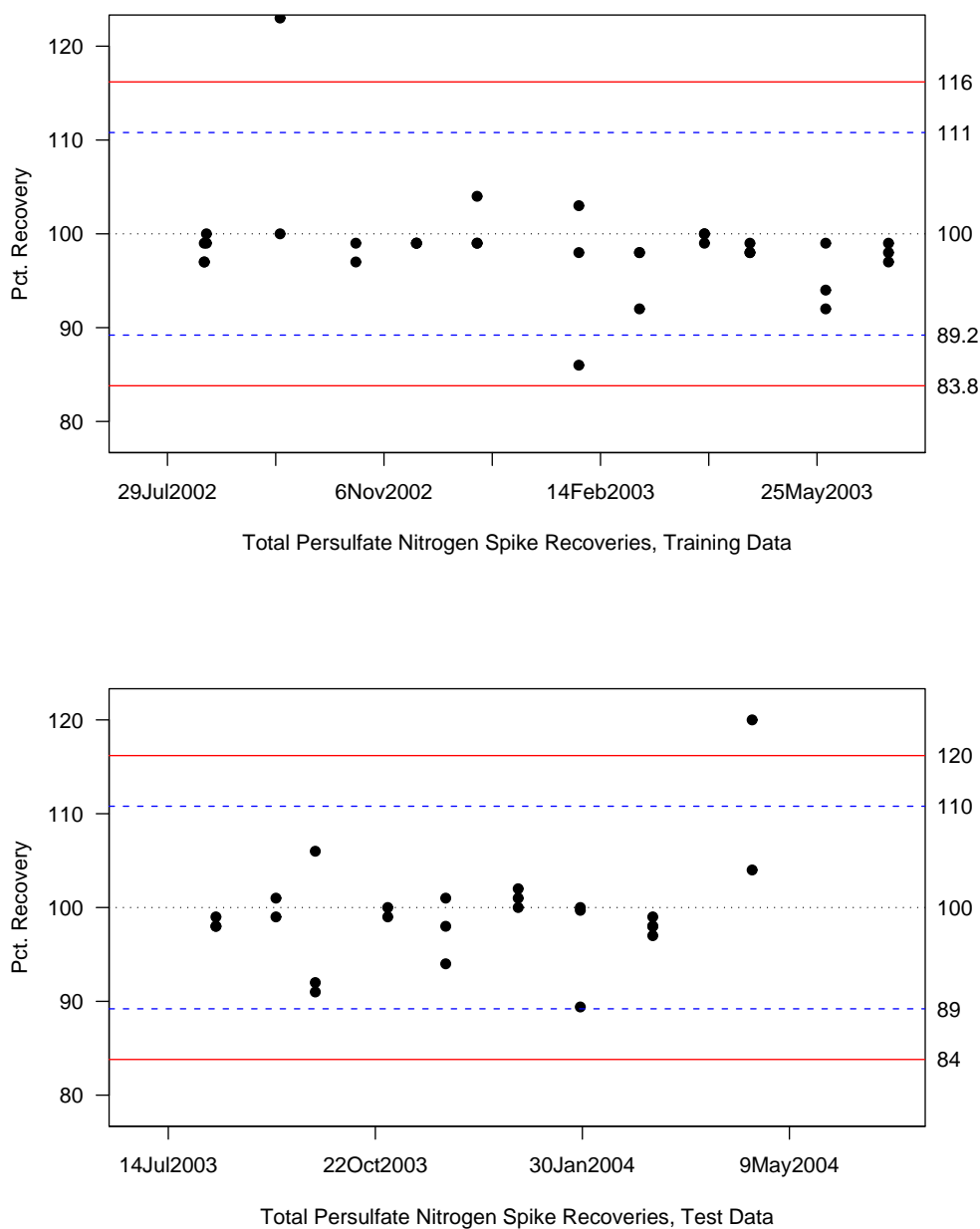


Figure 96: Total nitrogen spike recoveries for the 2002–2004 Abbotsford-Sumas Monitoring Project. Upper/lower acceptance limits (± 2 std. dev. from mean pair difference) and upper/lower warning limits (± 3 std. dev. from mean pair difference) were calculated using data from July 2002 through June 2003.

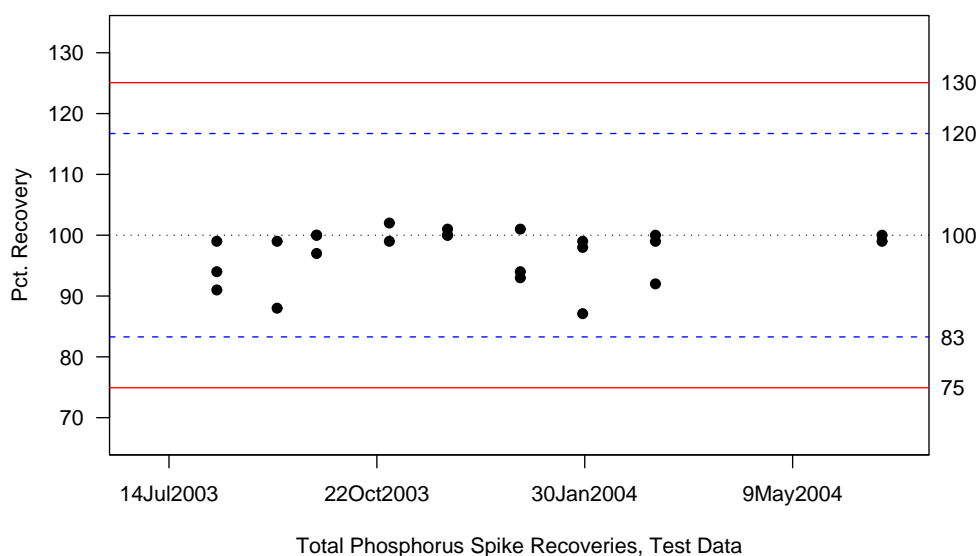
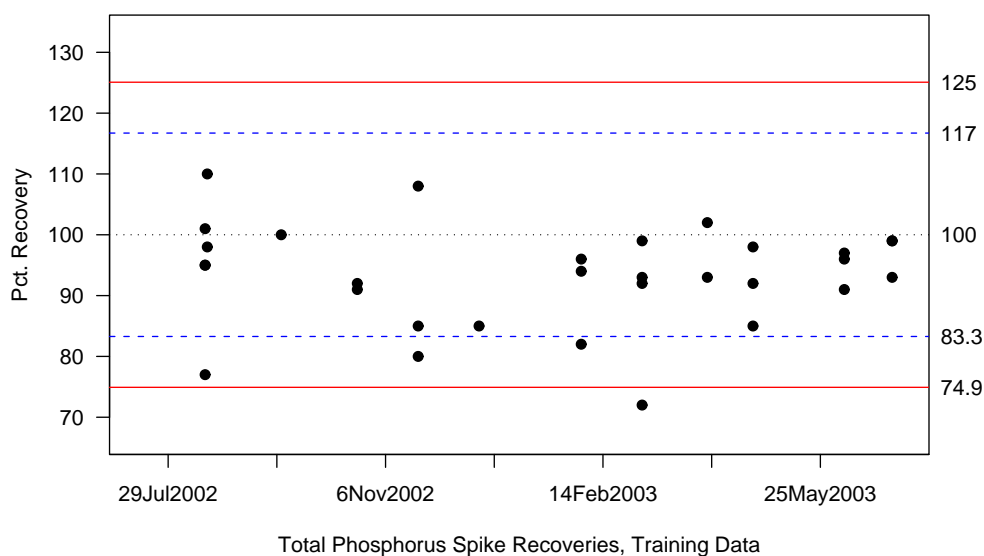


Figure 97: Total phosphorus spike recoveries for the 2002–2004 Abbotsford-Sumas Monitoring Project. Upper/lower acceptance limits (± 2 std. dev. from mean pair difference) and upper/lower warning limits (± 3 std. dev. from mean pair difference) were calculated using data from July 2002 through June 2003.