USGS
science for a changing world

Prepared in cooperation with the U.S. Environmental Protection Agency and the Whatcom Conservation District

Concentrations of Nutrients at the Water Table beneath Forage Fields Receiving Seasonal Applications of Manure, Whatcom County, Washington, Autumn 2011–Spring 2015




Scientific Investigations Report 2018–5124

U.S. Department of the Interior
U.S. Geological Survey

Cover:
Primary photograph: Holstein dairy cattle. Photograph by Nichole Embertson, Science and Planning Coordinator, Whatcom County Conservation District, 2014.
Upper left: Soil core sample, Whatcom County, Washington. Photograph by Stephen Cox.
Middle: Groundwater sampling, Whatcom County, Washington. Photograph by Stephen Cox.
Upper right: Forage grass, Whatcom County, Washington. Photograph by Stephen Cox

Concentrations of Nutrients at the Water Table beneath Forage Fields Receiving Seasonal Applications of Manure, Whatcom County, Washington, Autumn 2011–Spring 2015

By Stephen E. Cox, Andrew R. Spanjer, Raegan L. Huffman, Robert W. Black, Jack E. Barbash, and Nichole M. Embertson

Prepared in cooperation with the
U.S. Environmental Protection Agency and the Whatcom Conservation District

Scientific Investigations Report 2018–5124

U.S. Department of the Interior
U.S. Geological Survey

U.S. Department of the Interior
RYAN K. ZINKE, Secretary

U.S. Geological Survey
James F. Reilly II, Director

U.S. Geological Survey, Reston, Virginia: 2018

For more information on the USGS—the Federal source for science about the Earth, its natural and living resources, natural hazards, and the environment—visit https://www.usgs.gov or call 1–888–ASK–USGS.

For an overview of USGS information products, including maps, imagery, and publications, visit https://store.usgs.gov.

Suggested citation:
Cox, S.E., Spanjer, A.R., Huffman, R.L., Black, R.W., Barbash, J.E., and Embertson, N.M., 2018, Concentrations of nutrients at the water table beneath forage fields receiving seasonal applications of manure, Whatcom County, Washington, autumn 2011–spring 2015: U.S. Geological Survey Scientific Investigations Report 2018-5124, 41 p., https://doi.org/10.3133/sir20185124.

ISSN 2328-0328 (online)

Contents

Figures

Figures—Continued

Tables

Conversion Factors

U.S. Customary Units to International System of Units

Multiply	By	To obtain
	Length	
inch (in.)	2.54	centimeter (cm)
inch (in.)	25.4	millimeter (mm)
foot (ft)	0.3048	meter (m)
mile (mi)	1.609	kilometer (km)
	Area	
acre	4,047	square meter (m^2)
acre	0.4047	square hectometer (hm^2)
	Flow rate	
inch per year (in/yr)	0.0254	meter per year (m/yr)
	Leakance	
foot per day per foot [(ft/d)/ft]	1	meter per day per meter
inch per year per foot [(in/yr)/ft]	83.33	millimeter per year per meter [(mm/yr)/m]

International System of Units to U.S. Customary Units

Multiply	By	To obtain
	Length	
centimeter (cm)	0.3937	inch (in.)

Temperature in degrees Celsius (°C) may be converted to degrees Fahrenheit (°F) as:

$$°F = (1.8 \times °C) + 32.$$

Datums

Vertical coordinate information is referenced to the North American Vertical Datum of 1988 (NAVD 88).

Horizontal coordinate information is referenced to the North American Datum of 1983 (NAD 83).

Altitude, as used in this report, refers to distance above vertical datum.

Supplemental Information

Specific conductance is given in microsiemens per centimeter at 25 degrees Celsius (µS/cm at 25 °C).

Concentrations of chemical constituents in water are given in either milligrams per liter (mg/L) or micrograms per liter (µg/L). Nitrogen containing species such as nitrate (NO_3^-) and ammonia (NH_4^+) are reported as milligram of nitrogen per liter (mg-N/L). Phosphorus containing species, including phosphate (PO_4), are reported as milligram of phosphorus per liter (mg-P/L).

Abbreviations

ARM	Application Risk Management
CON	conventional management
EPA	U.S. Environmental Protection Agency
NRCS	Natural Resources Conservation Service
SBA	Sumas-Blain Aquifer
USDA	U.S. Department of Agriculture
USGS	U.S. Geological Survey
WCD	Whatcom Conservation District

Concentrations of Nutrients at the Water Table beneath Forage Fields Receiving Seasonal Applications of Manure, Whatcom County, Washington, Autumn 2011–Spring 2015

By Stephen E. Cox[1], Andrew R. Spanjer[1], Raegan L. Huffman[1], Robert W. Black[1], Jack E. Barbash[1], and Nichole M. Embertson[2]

Abstract

The U.S. Geological Survey, in cooperation with the Whatcom Conservation District (WCD), collected groundwater-quality data for roughly 3 years (October 2011–May 2015) from near the water table beneath forage fields receiving regular seasonal applications of liquid dairy manure in Whatcom County, Washington. The work was done as part of an evaluation of WCD's prototypical Application Risk Management (ARM) decision support system. The ARM system uses a combination of field-specific hydrology, stage of crop-growth, manure management practices, soil conditions, and precipitation forecast to evaluate the timing of manure application via a set of decision support tools (Manure Spreading Advisory, ARM Worksheet, manure application setback distances) in order to reduce the risk of contamination of surface water and groundwater. The ARM system's effectiveness in reducing leaching of nitrate to groundwater was evaluated by monitoring nitrate concentrations in recently recharged groundwater beneath paired test plots receiving manure application scheduled using either conventional (CON) manure scheduling procedures, which utilize fixed start and end dates for manure application along with projected crop nutrient requirements or ARM manure scheduling procedures using an approach to manure application timing based on projected crop nutrient needs, field conditions, and weather forecast. Water-quality samples from the surface of the water table were collected synoptically from paired test plots (2–5 monitoring wells per test plot) at approximately monthly intervals at three different dairy field sites. Water-quality samples from near the water table were isolated from the underlying aquifer using a combination of an inflatable packer and a fine-grained sand pack encompassing the well-screen interval.

Concentrations of nitrate and chloride measured at the water table beneath test plots were highly variable. Concentrations of nitrate ranged from non-detectable to 116 milligrams nitrogen per liter (mg-N/L), and chloride ranged from 1.15 to 153 mg/L. In each test plot, seasonal variations were much greater than spatial variations. Differences in nitrate concentrations in groundwater between the two treatments were inconclusive. Nitrate concentrations in groundwater at paired treatment plots (Mann Whitney, $p<0.05$) were significantly lower beneath the ARM treatment plot at site B, yet significantly higher beneath the ARM treatment plot at site C. Nitrate concentrations in ground water varied significantly among individual wells at each site (Kruskal-Wallis, $p<0.05$), indicating that leaching of nitrates from soil following manure application is spatially variable at the field scale tested regardless of manure application strategy. At all three paired test plots, average concentrations of nitrate and chloride at the water table were lowest near the end of the growing season (September) and increased rapidly with the onset of autumn rains (October–December). Under both the conventional (calendar-based) and treatment (ARM-based) manure application scheduling systems, high soil nitrate concentrations in autumn were coincident with rising groundwater levels, suggesting that nitrate and chloride were flushed from soil to groundwater by recharge from the seasonal rains. Under both treatments, concentrations of nitrate in shallow (10–25 feet) groundwater beneath forage fields receiving manure applications were greater than the nitrate drinking water standard of 10 mg-N/L in approximately 85 percent of samples. Yearly mass loading of nitrogen to the groundwater system calculated from nitrate concentrations at the water table and estimates of recharge volume ranged from 86 to 196 pounds-N per acre, which was equivalent to approximately 16–37 percent of the recommended manure application rate for projected forage production yield of 7 dry tons per acre per year. Manure nitrogen applied in the autumn, when crop nutrient needs decrease due to reduced sunlight and cooler temperatures and commensurate with ongoing mineralization of soil organic-nitrogen and increased seasonal precipitation, are more likely to exceed the immediate plant nutritional requirements and hence be flushed to groundwater than manure applications occurring near the peak of the growing season.

[1]U.S. Geological Survey.

[2]Whatcom Conservation District.

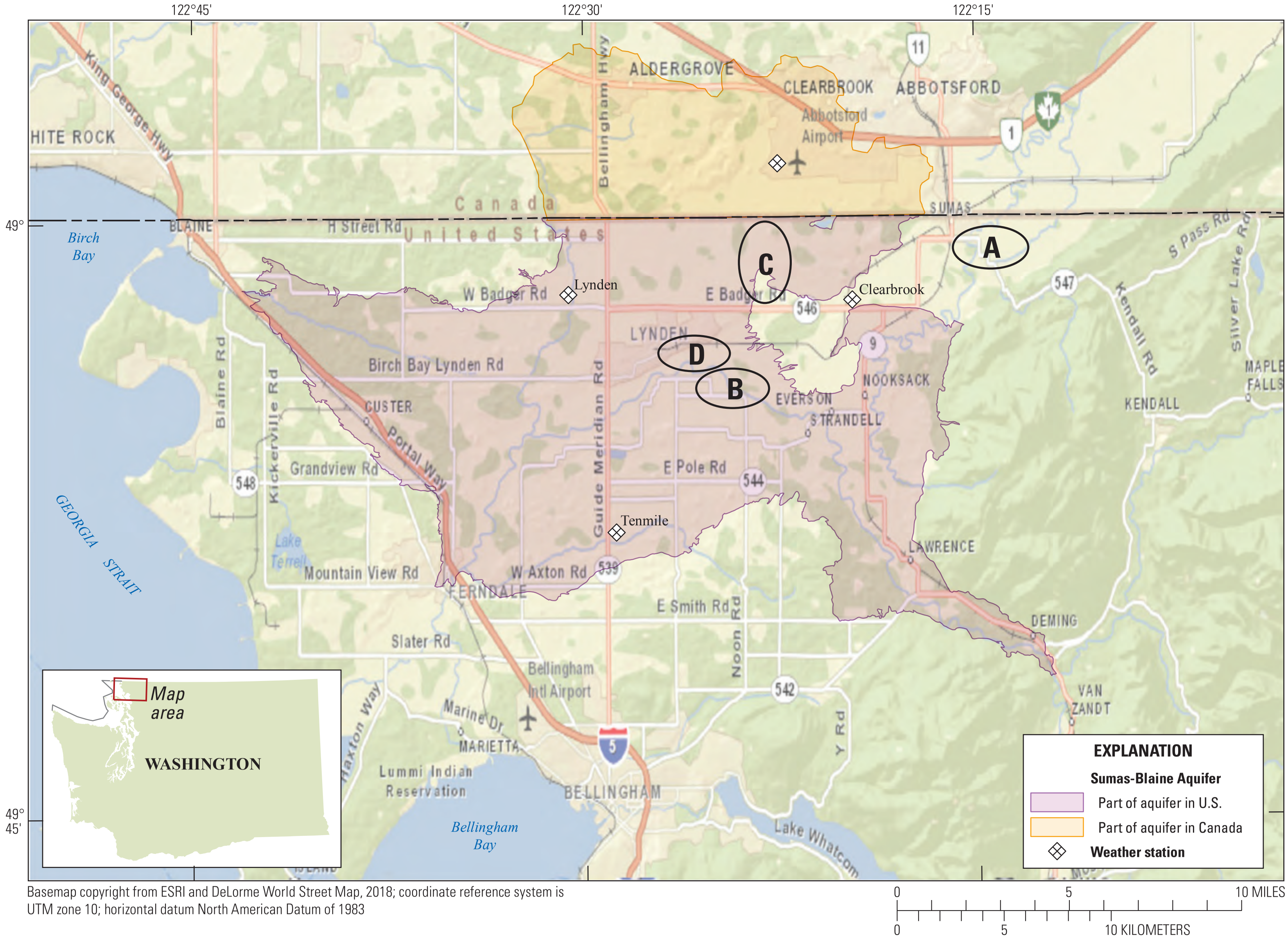
Introduction

Since the mid-20th century, increased production and use of nitrogen amendments in crop production in the United States have fostered significant increases in food production needed to support increasing human populations. Subsequently, concomitant increases of nitrogen released into aquatic environments have resulted in greater incidences of eutrophication and contamination of water resources (Robertson and Vitousek, 2009; Galloway and others, 2013). Excessive nitrogen in aquatic systems can be toxic to individual organisms and harmful to terrestrial and aquatic ecosystem stability (Ward and others, 2005; Compton and others, 2011; Sobota and others, 2013; Clark and others, 2017). Nationwide, the primary source of nitrate contamination of aquatic resources, including groundwater in many agricultural areas of North America, is associated with lengthy periods of high-intensity agricultural (Burkart and Stoner, 2001; Böhlke, 2002; Burow and others, 2010; U.S. Environmental Protection Agency, 2011). High-intensity agricultural practices that focus in part on maximizing crop yield often result in “leaky” soil systems in which a substantial fraction of the nitrogen added to the soil-crop system to improve crop production is lost from the plant-root zone by either leaching of solutes in soil to groundwater or volatilization to the atmosphere (Hermanson and others, 2000; Vitousek and others, 2009; Osmond and others, 2015; Zebarth and others, 2015). Although trends in fertilizer-use efficiency by crops have improved substantially in the United States in recent decades, agricultural activities remain the leading source of nitrogen released to the environment (U.S. Environmental Protection Agency, 2011). Despite these and other efforts to reduce the unwanted input of nitrogen to aquatic environments, the concentrations of nitrate in groundwater in some agricultural areas of the United States have continued to increase steadily over the last several decades (Rupert, 2008; Puckett and others, 2011; Sebilo and others, 2013).

In the area overlying the Sumas-Blain Aquifer (SBA) of northwestern Whatcom County, Washington (fig. 1), concentrations of nitrate greater than the U.S. Environmental Protection Agency’s (EPA’s) maximum contaminant level (MCL) for drinking water (10 mg-N/L) in groundwater have been an ongoing issue for decades (Cox and Kahle, 1999; Redding, 2008; Carey and Cummings, 2013; Carey, 2017). Carey’s (2017) trend analysis of groundwater nitrate concentrations, measured during 2009–16 in 25 wells in 3 localized areas of the SBA known to have large nitrate concentrations in groundwater, showed limited but significant changes in nitrate concentrations. Nine of their wells showed decreasing trends, and only one well showed increasing trends. Nitrate concentrations measured in nearly one-quarter of their study wells sampled in 2016 remained greater than the MCL, and over the course of the study period there was a deceasing trend in the total number of wells exceeding the MCL. While these data suggest decreasing nitrate concentrations in groundwater of parts of the SBA, high nitrate concentrations in community drinking wells drawing from the SBA exhibit large-scale spatial and temporal variation and remain a concern to drinking water providers (Steve Hulsman, Washington State Department of Health, written commun., 2018).

Much of the land area over the aquifer is in agricultural crop production, with a large fraction of acres in silage corn and grass production for dairy operations. Dairy production, although declining in this region, has been an important component of its economy since the middle of the last century. In 2003, the number of producing dairy cows in Whatcom County was 62,700, which has since decreased by 24 percent to 45,500 in 2016 (U.S. Department of Agriculture, 2003, 2016). Utilization of dairy manure as a source of supplemental nutrients for forage crop production is a long-standing component of agricultural practices in Whatcom County, providing nitrogen for crop growth, improvement in soil quality, and recycling options for accumulated manure. Generally, crop yield responses to supplemental nitrogen amendments show large increases at small and intermediate rates of application, with diminishing rates of improvement at higher amendment rates (Viets, 1965; Hermanson and others, 2000; Singer and Moore, 2003).

The use of animal manures as a supplemental source of nitrogen for plant growth is more complex than use of nitrogen fertilizers because much of the nitrogen in animal manures is not in a form that is immediately available for uptake by plants. Conversion of manure nitrogen into plant available forms is largely governed by soil microbial processes that vary with soil temperature and moisture conditions. Nitrogen in dairy manure is roughly equal parts organic matter and inorganic forms of nitrogen including urea and ammonium. Urea and ammonium in manures is rapidly nitrified (microbially converted to nitrate) following application to the soil surface and subsequently available for crop uptake. However, 50 percent of manure nitrogen in the form of organic matter is chemically diverse and more stable than urea or ammonia requiring complex microbial conversion (mineralization) to plant-available forms. The rate and amount of organic matter mineralized from soil and manure changes from year to year depending on seasonal and climate variations. Typically, only about 40 percent of the organic nitrogen in liquid dairy manures is converted to plant available forms in the year in which dairy manure is applied to cropland. Roughly 15, 7, and 3 percent of the original manure application will become plant available in the 3 years following manure application with smaller fractions available in following years (Sullivan, 2008; Sebilo and others, 2013).



Basemap copyright from ESRI and DeLorme World Street Map, 2018; coordinate reference system is UTM zone 10; horizontal datum North American Datum of 1983

Figure 1. Locations of field sites A–D, Whatcom County, Washington.

Much of the nitrogen from applied manure is incorporated in soil organic matter as plant roots and microbial biomass and can remain in the soil for many years. Following a single application of radiolabeled nitrogen fertilizer to a lysimeter, Sebilo and others (2013) found diminishing concentrations in the crop, soil, and drainage water for more than 20 years indicating that once nitrogen is incorporated into soil organic matter, it can be retained for many years.

Cool-season grasses grown for dairy forage in Whatcom County are considered some of the most favorable crops for limiting leaching losses of soil nitrogen from fields because of the high rate of nitrogen uptake by the grass crop and year-round growth (Cherney and others, 2002; Singer and Moore, 2003; Sullivan, 2008). In the Pacific Northwest's mild climate, perennial forage grass crops are expected to continue to grow throughout the year because prolonged periods (greater than 10 days) of sub-freezing weather are rare. In the study area, about 70 percent of annual rainfall occurs from October 1–March 1, while growth curves for cool-season grass during that period indicate just 6–10 percent of the annual growth is produced (Yungen and others, 1977; U.S. Department of Agriculture, 2017). Management strategies to extend crop production beyond the prime growing season may result in nitrate present in soils at the beginning of the seasonal rainy period that are greater than can be utilized by crops prior to being leached by groundwater recharge. Leaching losses maybe particularly large in years when actual crop yield falls short of the expected yield for which manure nitrogen has been applied.

At the time of this study, regulatory guidance on timing of manure application to established dairy grasslands in Whatcom County was developed and updated in conjunction with the Washington Dairy Nutrient Management Act of 1998 (RCW 90.64), Whatcom County "Manure and Agricultural Nutrient Management" Ordinance (16.28) (2011) U.S. Department of Agriculture, Natural Resources Conservation Service (NRCS) Nutrient Management Practice Standard (590) (2013), and USDA/ NRCS Winter Period Application of Manure in Washington State (2014). For established grasslands, manure application was historically allowed starting February 15, or when the summation of the mean daily temperatures (Celsius), beginning on January 1, exceeds 200 (also known as, "TSum-200"). The post-growing season period of non-application of manure begins on October 15 in floodplain areas and October 31 everywhere else. These dates were chosen to coincide with the beginning of the growing season and the beginning of the rainy season, respectively, with no provisions made for near-term hydrologic and weather conditions. Reliance on specific calendar dates for the beginning and end of manure applications, particularly in autumn when crop uptake of nutrients is diminishing and rainfall intensity is increasing, may present increased risks of nutrient loss to leaching (Paul and Zebarth, 1997; Beckwith and others, 1998; Smith and others, 2002; Van Es and others, 2006; Hepperly and others, 2009). Allowing spring manure application to begin on a specific calendar date may also encourage manure application during high precipitation events or when excess soil moisture could contribute to runoff.

In 2010, the Whatcom Conservation District (WCD) initiated a program intended to reduce the potential risk of contamination of water resources from application of dairy manure to cropland. It focused specifically on the timing of manure application, not the application rate, to minimize potential loss of nutrients and pathogens from fields via leaching and/or runoff. The intent of the manure management system, referred to as application risk management (ARM), was to develop a set of online and real-time decision-making tools that draw together the information needed to improve scheduling and management of manure application to cropland (Whatcom Conservation District, 2015). The ARM tools include a real-time manure spreading advisory, ARM Field Evaluation Worksheet, and seasonal manure application setback distances. The ARM Worksheet, used for scheduling and assessing the need for manure application to cropland, includes consideration of hydrologic properties of specific soils and fields, soil physical parameters, and current and forecasted local precipitation for the 3 days immediately following manure application. The WCD evaluated the ARM system (ARM field test) by monitoring soil zone nutrient concentrations in paired test plots at three different dairy forage fields in Whatcom County from 2010 to 2015. At each test site, similar test-plots received manure applications using either conventional (CON) manure scheduling procedures using set dates, or the proposed ARM procedures using field conditions to determine application timing. For the most part, the first and last manure application events (January–February and September–October) differed in timing between the two treatments, while mid-season applications (March–August) were the same. The annual amount of manure applied was not dictated as part of the study (determined by the land manager based on agronomic needs), with the exception that the total annual amount of manure applied to the paired test plots in the same field be equal. Additionally, they assessed leaching losses from the soil using samples of vadose zone water collected from gravitational lysimeters placed at 12, 24, and 36 inches (in.) below the surface at random locations in each sample field, as well as soil samples at the same horizons and forage samples at each harvest event.

This groundwater quality study is a component of the ARM field evaluation. It focused on monitoring changing concentrations of nutrients and fecal bacteria reaching the water table of the underlying shallow aquifer. Samples of groundwater were collected from wells at the water table beneath the paired test plots at the field evaluation sites. Monitoring nitrate concentrations in groundwater was expected to provide broader and more-integrated information regarding the leaching and transport of nutrients and bacteria from agricultural soils to the groundwater system rather than relying on soil zone monitoring alone, which is subject to very high variability due to the heterogeneous nature of soils and soil microbial communities (Viera and others, 1981; Bruckler and others, 1997) and the tendency for soil water to follow preferential flow paths (Close, 2010; Gerke and others, 2010; Nimmo, 2012). In aerobic groundwater systems, nitrogen in the nitrate form is typically conservative; however, under anaerobic conditions, denitrification may remove nitrogen from groundwater in the form of dissolved nitrogen gas or may transform nitrogen to ammonia (Tiedje, 1988; Korom, 1992).

The purpose of this report is to describe the spatial and temporal variation in concentrations of nutrients including nitrate and bacteria in groundwater beneath paired test plots with similar soil and crop conditions and receiving manure applications according to either CON or ARM application strategies. The tests were done to assist in the evaluation of the ARM strategy at three sites from autumn 2011 to spring 2015. Groundwater samples were collected approximately monthly from two to five wells in each test plot. The nitrate concentration data from the water table were used to calculate an estimated loading of nitrate to groundwater from the root zone. Embertson (2016) documented soil data and details of ARM project.

Description of Study Area

Hydrogeology of Region

The study area was in the Nooksack River lowland section of northwestern Whatcom County, Washington, near the U.S.-Canadian border (fig. 1). The extent of the unconfined SBA is roughly 150 mi^2 of northwestern Whatcom County, with an approximately equal area extending north of the International Boundary with Canada. In Canada, the aquifer is referred to as the “Abbotsford-Sumas,” and it faces similar land-use and groundwater-quality issues as the U.S. side (Gleeson and others, 2012; Graham and others, 2015; Zebarth and others, 2015). The SBA has several physical features that are characteristic of aquifers vulnerable to contamination from agricultural land-use activities, including shallow depths (10–25 feet [ft]) to groundwater through highly transmissive soil and unsaturated zone, large fluxes of water moving through the unsaturated zone, and high rates of nutrient amendment applied to the land surface. Soils overlying this aquifer are composed of a mixture of eolian loess and volcanic ash, with the upper 1–2 ft of the soil generally consisting of silty-loam with moderate permeability in the range of 0.6–2 inches per hour (in/hr) (Goldin, 1992). Soils in the area are typically 24–39 in. thick, with cumulative water holding capacity in the soil column of 8–9 in. (Goldin, 1992). Agricultural soils in Whatcom County contain from 3 to 9 percent organic matter in the upper 6–12 in. of soil (Golden, 1992). Using a carbon to nitrogen ratio of 12:1, the nitrogen content of these soils varies from about 3,000 to 9,000 lb-N/acre. However, nearly all this nitrogen occurs in the form of soil organic matter that is unavailable for crop uptake. The potential annual microbial conversion of soil organic matter (mineralization) is about 1–2 percent of soil organic matter to plant available forms of nitrogen such as nitrate (Meisinger and others, 2008). Addition of supplemental nitrogen is thus required to achieve desired crop production yields.

The underlying sediments are glacial fluvial outwash composed predominantly of stratified sand and gravel mixed with cobble and silt containing localized discontinuous lenses of silt and fine-grain deposits. Substantial hydrogeologic heterogeneity occurs in the sedimentary material that results in complex groundwater flowpaths at the scale of individual agricultural fields (Cox and Kahle, 1999). Beneath the highly permeable surficial sediments is a regionally extensive deposit of silty-clay material (interglacial and glacial-marine) that forms an effective regional confining unit. The hydrogeologic characteristics of hydro-stratigraphic units in the Nooksack River lowlands were previously summarized in more detail by Cox and Kahle (1999). Artificial drainage is often required for agricultural activity in soils of many of the flat-lying areas.

Topography and surface relief of the area varies from generally flat-lying to gently sloping. The outwash deposit that comprises the SBA is typically about 40–80-ft thick, with accumulation being thicker north of the international boundary. The aquifer is highly productive, with typical hydraulic conductivities ranging from 74 to 610 feet per day (ft/d), and horizontal groundwater velocities on the order of 2.8–23 ft/d (Cox and Kahle, 1999). Depth to groundwater is typically in the range of 10–25 ft and fluctuates in response to seasonal precipitation. The range of annual fluctuations in groundwater-level altitudes is typically 2–6 ft, depending on local topography, precipitation rates, and soil properties. Groundwater discharge to local streams and surface drainage ditches is also seasonally variable (Cox and others, 2005). Groundwater-level altitudes are typically lowest in early autumn before the seasonal rains begin. Maximum seasonal water levels usually occur during winter months.

Precipitation across the study area ranges from 32 to 60 in/yr and is strongly influenced by marine weather systems from the Pacific Ocean and orographic effects of the Coast Mountains and Cascade foothills east of the study area (Cox and Kahle, 1999). The summer growing season is generally warm with light precipitation. The autumn and winter period that begins late September–October is characterized by a steady progression of low-pressure systems from the Pacific Ocean generating cloudy and rainy conditions. Rainfall intensity is usually light to moderate and often continues for several days. Typically, four to five annual rainfall events exceed 1 in. within 24 hours. Solar radiation varies seasonally, affecting incident solar radiation and evapotranspiration in-turn affecting forage and crop production. The average monthly accumulated incident solar radiation recorded at Lynden and Ten Mile weather stations (http://weather.wsu.edu/) ranges from roughly 680 megajoules per square meter (MJ/m^2) in July to less than 100 MJ/m^2 in December and January. Occasionally, during mid-winter (typically during late January), strong high-pressure systems develop over the continental interior inducing cold northeasterly winds that funnel down the Fraser River system producing a cold snap without significant precipitation that can last 5–10 days.

Recharge of the surficial aquifer is generated as soil moisture from precipitation, and irrigation exceeds the water-holding capacity of the soil column. This process can be complex, occurring as either diffuse flow between sediment grains or as preferential flow through large connected macropore spaces (Beven and Germann, 1982; Nimmo, 2005, 2012). Supplemental irrigation is required for crop production during much of the summer growing season due to the limited precipitation during the warm summer months (fig. 2). Periods of rising groundwater levels and groundwater recharge typically begin in late September or early October and extend through February–March. Estimates of the amount of regional groundwater recharge occur over the study area range from 26 to 30 in/yr (Vacarro and others, 1998; Cox and Kahle, 1999). Soluble chemicals in the soil zone, such as nitrate and chloride, can be mobilized and leached to groundwater during this seasonal recharge process. Numerical simulations of nitrate leaching at a site near the Abbotsford International Airport, British Columbia, north of Whatcom County suggest that nitrate is completely leached from the soil zone to the water table within 3 months of saturation of the soil water column (Chesnaux and Allen, 2008).

Description of Field Sites, Well Installation, and Test Plots

Four field sites were selected for evaluation. These fields were used for forage silage grass production and were expected to remain in forage production for the duration of the project. Each field was at least 10 acres in size, had uniform soils classification across all 10 acres, was generally flat, and had consistent manure application in the preceding 3 or more years. Hydrologic criteria for field site selection included limited surface gradients, proximity to hydrologic divides, absence of surface-water features that might influence groundwater flowlines, and depth to groundwater less than 25 ft. Using these criteria, four field sites were originally selected through coordination with growers interested in participating in the study. Sites are referred to as A, B, C, and D (fig. 1). The predominant soil type and length of time each field remained in the study is listed in table 1. Site A was converted to berry production in the first year of the study, before groundwater-sampling infrastructure was installed, and no usable groundwater-quality data were obtained from this site.

All field test sites were forage fields with a mixture of orchard and tall fescue grasses. Fields received seasonal applications of manure (January/February–September/October) and irrigation water (May–September). Forage was typically harvested five to six times per growing season (February–October), depending on weather and growth rates. All manure applications were scheduled to occur during time periods of non-precipitation based on weather forecasts and timing of forage harvesting. Conventional (CON) manure applications were done five to six times per year during the growing season starting mid-February (TSum-200 or February 15) to approximately October 31. The same number of manure applications were applied to ARM treatment plots. However, the first manure application of the season occurred during a mid-winter dry period in mid-late January instead of February as in CON, and the last application of the year was in September versus October for CON. Manure applications from March to September were typically done on both plots at the same time. Manure applications made on the ARM treatment plots took into consideration field conditions (soil moisture, forage condition, water table depth) in addition to 24-hour and 72-hour precipitation forecast. Application timing for the CON treatment took only date and current (24 hour) precipitation into consideration. Manure was typically applied with a splash-plate drag hose system, although a “big gun” or aerator option were occasionally used early season. Manure application rate was not dictated in this study, but rather determined and monitored by the landowner.

Sites B and D were located on a relic channel floodplain of the Nooksack River, and site C was located on the Lynden glacial terrace, all near the town of Lynden in Whatcom County, Washington. The sandy soils at sites B and C were classified as Kickerville silt loam, characterized as very deep, well-drained soils comprised of loess and volcanic ash overlaying gravelly glacial outwash (Goldin, 1992). The top 36 in. of Kickerville soils are a silty loam with high organic matter and moderate permeability. Below 36 in., the soil tends to be gravelly or cobbled, with high permeability but reduced water holding capacity. Runoff is characterized as very slow, and erosion is very low due to gentle topography. The seasonal high water table tends to be less than 6 ft from the surface in the winter months. The silty soil at site D was classified as Puget silt loam, characterized as a very deep, poorly drained soil. The top layer is silt or silty clay loam with mottled silt loam beneath. The permeability is moderately slow, and there is a relatively high water table (at a depth of 1–3 ft) present from November through April. Nearly flat topography limits surface runoff, which is slow and typically affected by soil saturation and the high water table.

At each field site, paired test plots of approximately 5 acres each were delineated. A total of five to nine wells were installed at each completed test site, with two to five wells per treatment plot (table 2). Site B was enrolled in the study in 2011, site C in 2012, and site D in 2013.

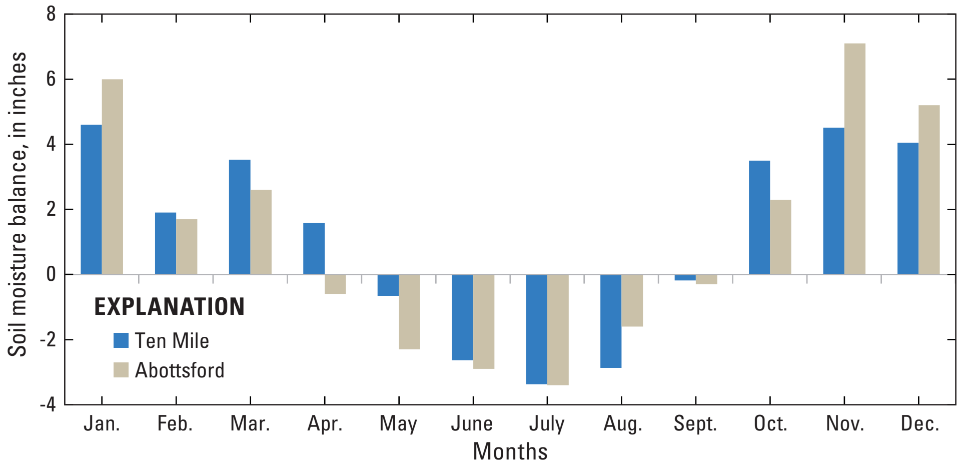


Figure 2. Long-term cumulative net monthly soil-moisture balance computed as the sum of daily precipitation minus daily reference evapotranspiration for weather stations at Tenmile Creek (Tenmile), Washington, and Abbotsford International Airport (Abbotsford), British Columbia, Canada. Locations of weather stations are shown in figure 1.

Table 1. Description of four field sites, Whatcom County, Washington.

[**Site:** Location of sites shown in figure 1]

Site	Soil type	Soil group	Number of monitoring wells installed	Dates enrolled	Reason for exit
A	Sand	Mount Vernon	3	October 2011–July 2012	Conversion to berry production
B	Sand	Kickerville	8	October 2011–May 2015	End of project
C	Sand	Kickerville	5	October 2012–April 2015	End of project
D	Silt	Puget	9	October 2013–May 2015	End of project

Groundwater sampling locations were generally co-located with pan-lysimeter sites used in the companion WCD study. Monitoring wells were installed in the test fields following the last cutting of forage grass for the season (October), generally just before seasonal low water-table altitudes.

Wells ranged in depth below land surface from 4.1 to 24.5 ft, with screen lengths ranging from 0.5 to 15.0 ft. Wells were constructed with 1.0- or 2.0-in.-diameter flush thread polyvinyl chloride (PVC) pipe and screen. Slot size of the screen intervals was 0.010 ft. Fine grained (20–40 mesh size) quartz sand was used to fill annular space between the well screen and aquifer material. The fine-grained sand pack placed in the annular space between the well and the aquifer formation using a tremie tube was intended to inhibit groundwater flow around the inflatable packers used to isolate the sampling screen interval. Eight-inch flush mount well enclosures were set in 20-in. concrete pads at each well. Concrete and grout surface seals extended to 2 ft below land surface. Well construction information is listed in table 2. Each well was developed by pumping groundwater at flows of 1–2 L/min until turbidity cleared from the pumped water. During the first year of water table sampling, extended purging was done for the isolated water table zone to remove silt from undeveloped sand pack around the isolated screened area. Well development continued through the first autumn as rising water level encountered undeveloped sand-pack around the well screen.

At site B, the 5-acre CON and ARM treatment plots were located along a gentle rise on the eastern margin of the field with a 30-ft buffer strip between adjacent raspberry fields and a gentle slope from north to south. Eight wells were installed at the site in October 2011, four wells in the CON treatment plot and four wells in the ARM treatment plot (fig. 3; table 2). Well depths at site B ranged from 8.8 to 24.5 ft below land surface. All wells terminated in a silt-clay confining layer that is at least 1–5 ft thick. Wells were developed and an isolation packer was tested in November 2011. The first reliable sampling of the isolated water table occurred on December 8, 2011, after water levels had risen in all wells following the beginning of heavy seasonal rainfall.

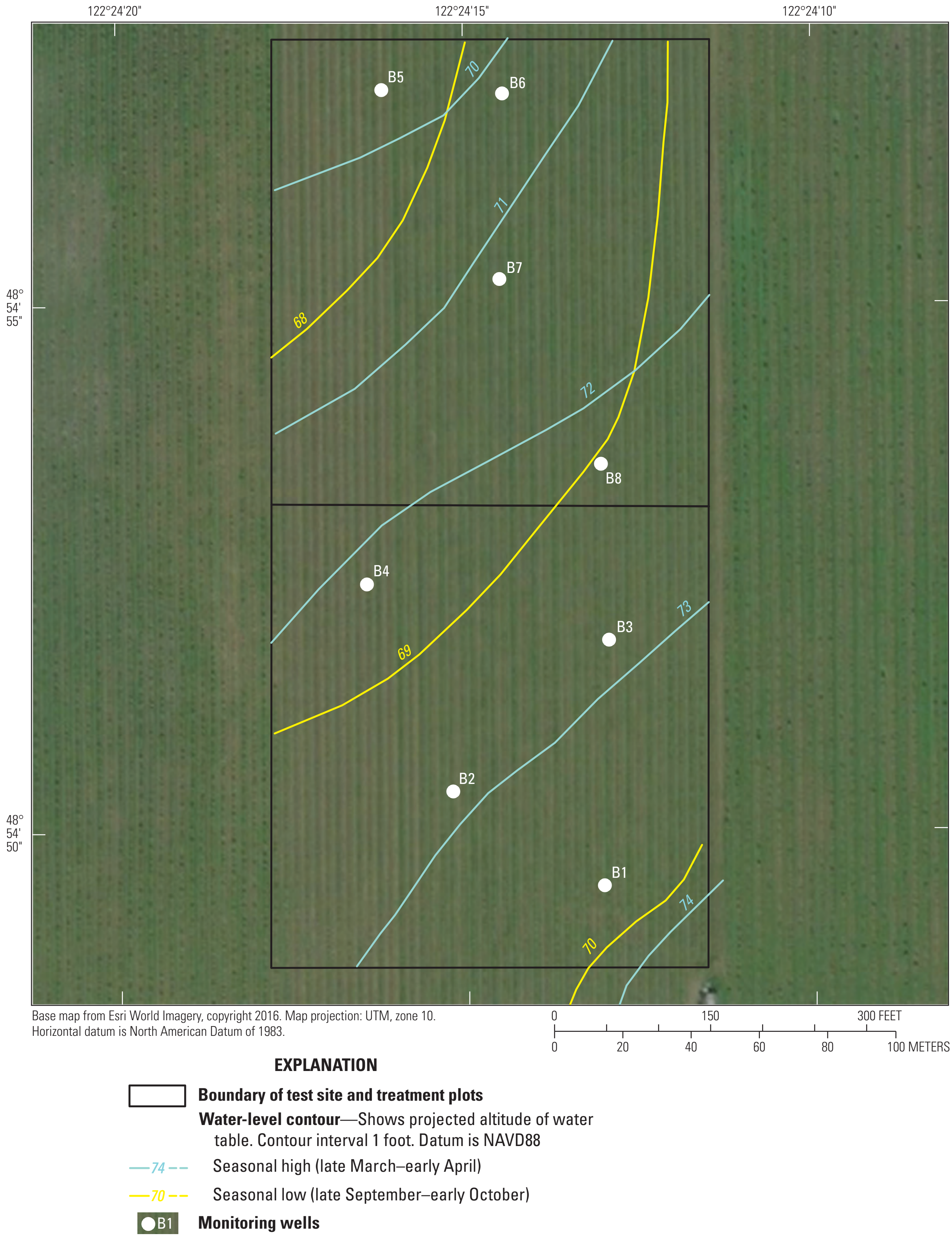


Figure 3. Location of wells and seasonal high and low water-level altitudes at site B, Whatcom County, Washington. Location of site B shown in figure 1.

Table 2. Well construction information at field sites B, C, and D, Whatcom County, Washington.

[Location of sites shown in figure 1. **Treatment:** ARM, application risk management; CON, conventional. **Abbreviations:** NAVD 88, North American Vertical Datum of 1988; ft, foot, –, not applicable]

Well identifier	Treatment	Well depth (ft below land surface)	Land-surface altitude (ft above NAVD 88)	Screen interval (ft below land surface)	Length of open interval (ft)
Site B					
B1	ARM	24.5	85.00	4.5–19.5	15.0
B2	ARM	19.8	84.12	5.5–19.5	14.0
B3	ARM	14.9	80.34	4.9–14.9	10.0
B4	ARM	19.4	80.91	4.4–18.4	14.0
B5	CON	9.8	76.70	3.6–9.8	6.2
B6	CON	8.8	76.79	3.8–8.8	5.0
B7	CON	10.5	77.78	3.5–10.5	7.0
B8	CON	10.8	78.52	3.8–10.8	7.0
Site C					
C1	CON	19.5	146.62	9.5–19.5	10.0
C2	CON	23.0	146.88	13.0–23.0	10.0
C3	ARM	22.0	145.37	13.0–22.0	10.0
C4	ARM	24.0	146.95	14.0–24.0	10.0
C5	ARM	23.0	149.79	13.0–23.0	10.0
Site D					
D1	CON	6.7	59.00	2.1–6.7	4.6
D2	CON	5.3	60.08	4.8–5.3	0.5
D3	CON	4.9	58.49	2.0–4.9	2.9
D4	CON	4.1	58.09	1.6–4.1	2.5
D5	CON	4.1	58.30	1.5–4.1	2.7
D6	ARM	8.6	60.72	2.5–8.6	6.1
D7	ARM	6.1	59.42	1.4–6.1	4.7
D8	ARM	6.3	59.19	2.3–6.3	4.0
D9	ARM	7.7	60.11	1.7–7.7	6.0

Initially, the southern plot was the CON treatment plot receiving calendar-based manure applications, and the northern plot was the treatment plot receiving ARM-based applications. However, unanticipated field maintenance was done in spring 2012, and the entire field was plowed and reseeded with the same mix of fescue and orchard grasses. However, the designation of CON and ARM treatment plots switched. Granular fertilizer was applied to both plots as part of the reseeding process. The overall effect of the switch likely did not significantly affect study conclusions. Data are presented for the entire study period; however, data used in the statistical tests were limited to data for the recharge period following the switch of treatment plots. There was no difference in manure application the previous autumn, and only one manure application event had occurred on the original ARM treatment plot before the switch. Any measurable effect would likely be limited to the first years, as annual groundwater recharge flushes soluble constituents out of the unsaturated zone.

Five wells were installed at site C in October 2013 (fig. 4 and table 2). Heavy rain halted the installation of additional wells at that time. Three wells were completed in the ARM treatment plot, and two wells were installed in the CON treatment plot. Wells were constructed of 1-in. flush threaded 1-in. diameter PVC pipe and screen with a screen slot size of 0.010 ft.

Nine wells were installed at site D in October 2014, five in the CON treatment plot and four in the ARM treatment plot (fig. 5 and table 2). The wells were constructed of 2-in. flush threaded PVC pipe with 0.010 ft slot screen with concrete surface seals extending 2 ft below ground surface. During some sampling events, the water-table altitude was above the screened interval rising into the surface seal interval. There was often standing surface water in the western one-third of the site adjacent to an agricultural drainage ditch. Due to field conditions, manure application scheduling was the same for both CON and ARM treatment plots during most of the time water samples were collected at site D. Near the end of the period of field investigations, the CON plot (wells D1-D4) received a manure application while the ARM plot did not.

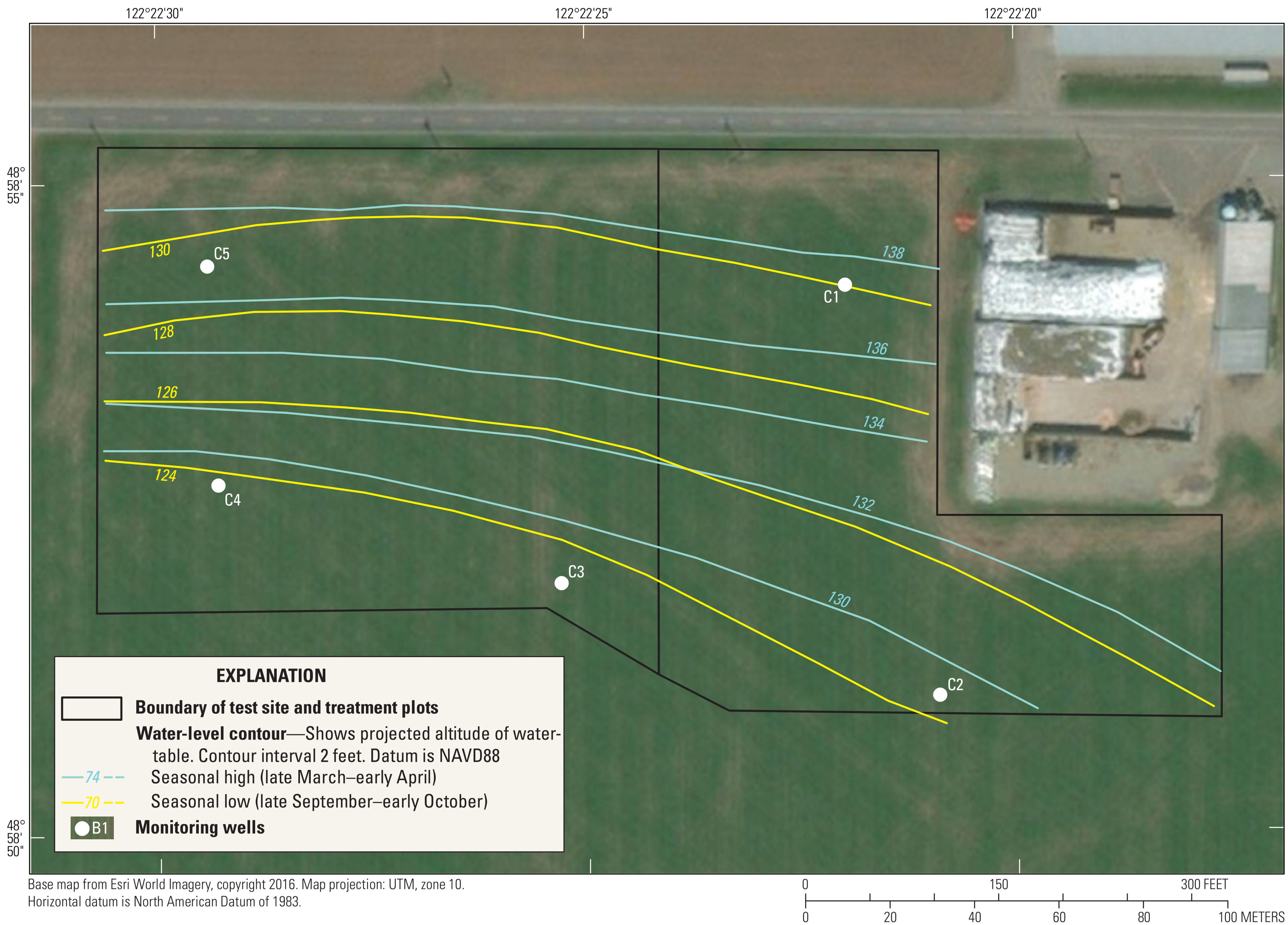


Figure 4. Location of wells and seasonal high and low water-level altitudes at site C, Whatcom County, Washington. High groundwater altitudes typically occur in late March to early April. Low groundwater elvations typically occur in late September to early October. Location of site C shown in figure 1.

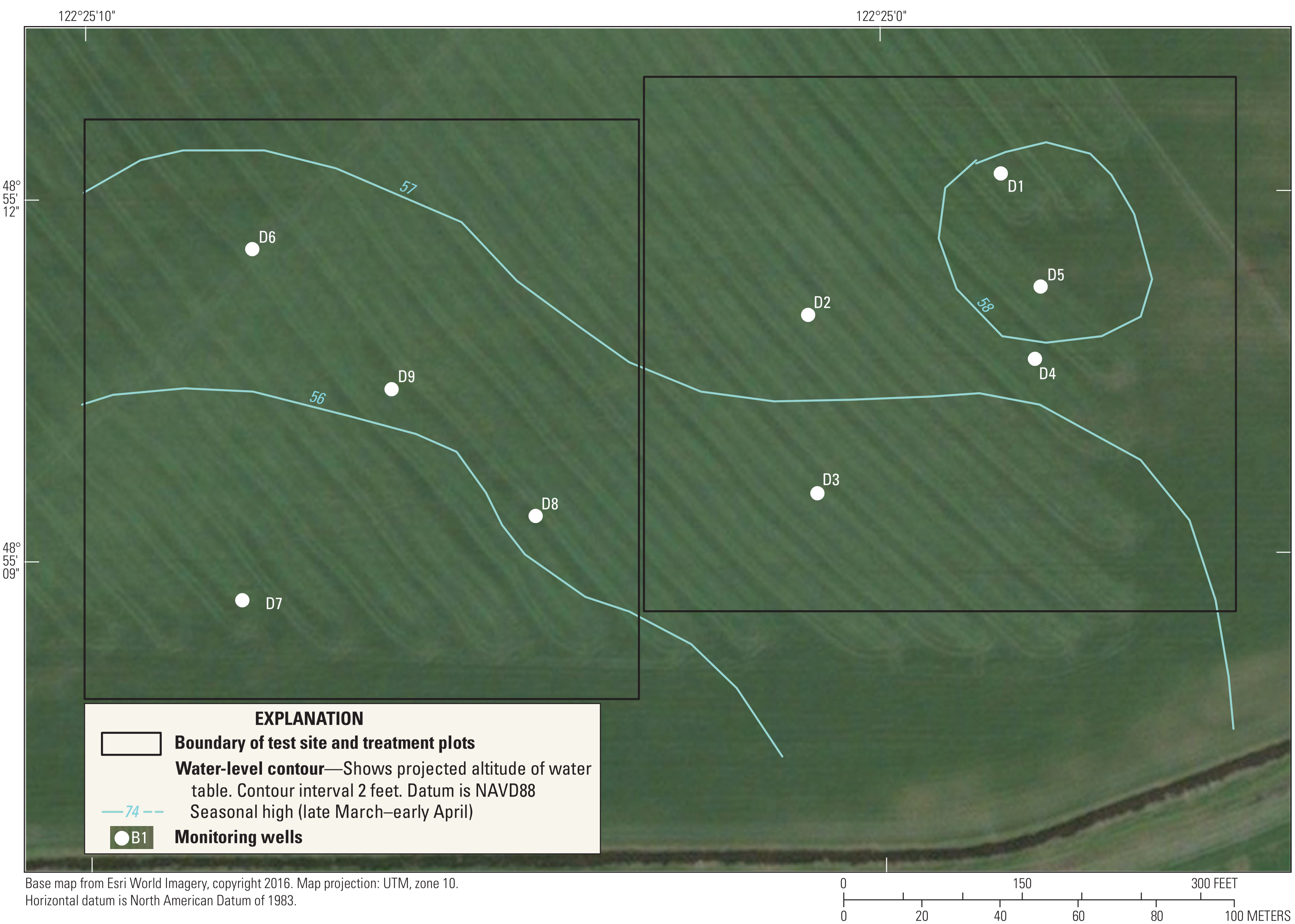


Figure 5. Locations of wells and seasonal high water-level altitudes at site D, Whatcom County, Washington. Location of site D shown in figure 1.

Methods

Collection of Groundwater Samples

Samples of groundwater from near the water table were collected from wells at each of the paired test plots at sites B, C, and D. Sampling procedures incorporated a movable packer installed inside the well to isolate the upper 6 in. of the saturated screened zone. During the first 6 weeks of sampling, a sterilized, cellulose sponge packer attached to a support rod was placed in each well to isolate the sampling zone. The sterilized sponge packer was held in place using a threaded rod and large-diameter fender washers. This assembly was placed in the well 6 in. below the water level. The sterilized tubing was placed in the well so that the obtained groundwater sample was from roughly 1–2 in. above the top of the cellulose packer. The use of the cellulose packer precluded simultaneous collection of water samples from above and below the isolation point in the well and was replaced with an inflatable bladder packer beginning in January 2012 that allowed continuous monitoring of field parameters above and below the isolation point during the well-purging process. Simultaneous samples confirmed that the water table samples were distinct and presumable not mixed with groundwater from the deeper saturated zone open to the well below the packer. A schematic diagram of the inflatable packer that was constructed of stainless steel and Viton® rubber is shown in figure 6. The packer-based method was used to collect all water quality samples, except when water levels were in the bottom 6 in. of the screened interval of the well, precluding the need for the packer. At site D, where summer water levels declined below the depth of the well screen, a drive-point assembly constructed with 6-in. stainless steel screen attached to 3/8 in. polyethylene tubing was driven into the ground to extend the depth of the sampling interval so that a water sample could be collected. Water-level data were not collected on these sampling dates.

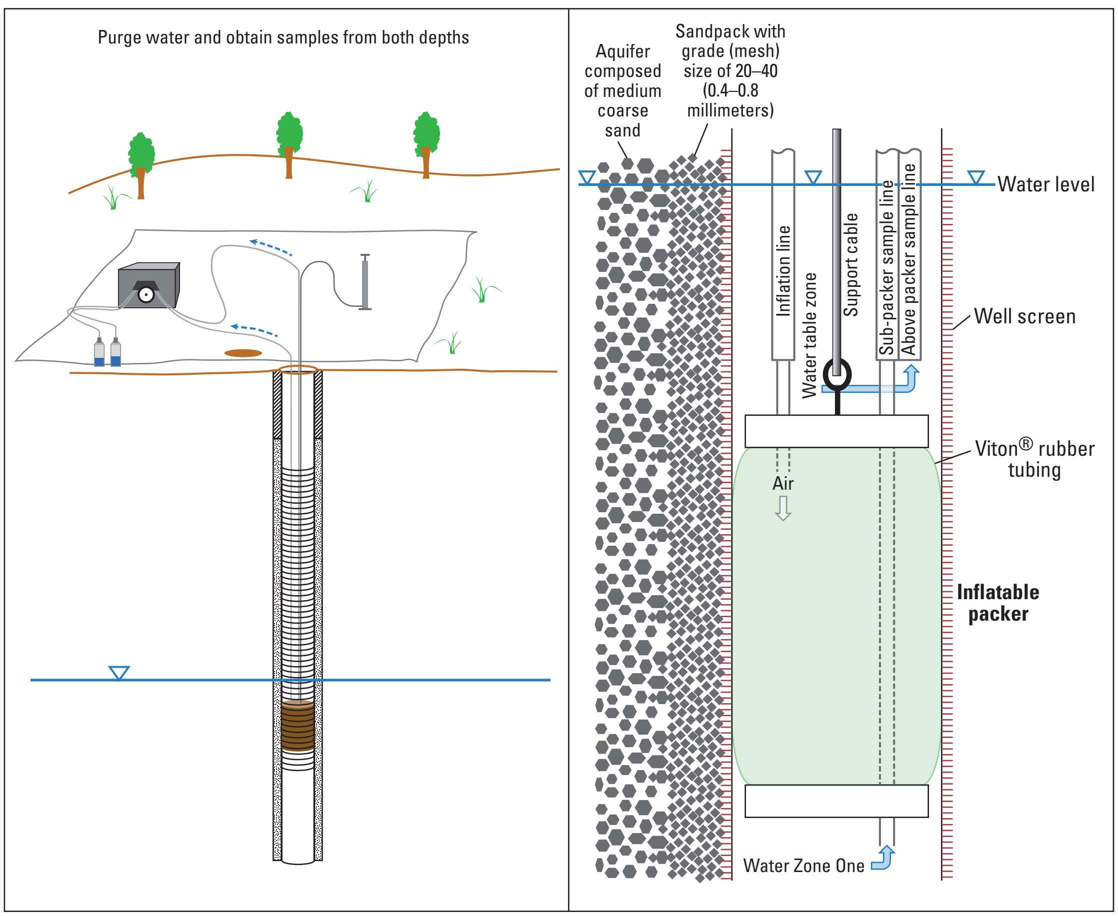


Figure 6. Schematic diagram of sampling wells and inflatable packer used to isolate upper 6 inches of water-table zone for sampling most recently recharged groundwater, Whatcom County, Washington.

Just before sampling, the static water level was measured to the nearest 0.01 ft using an electric tape following procedures outlined in Drost and others (2005). The static water level was used to determine the placement of the packer. Two lines of pre-cleaned and sterilized polyethylene tubing (1/4 or 1/8 in. diameter) were attached to the sterilized packer, one for the water table (WT) zone (top 6 in. of the saturated aquifer) and the other for the zone beneath the sealed packer (Zone 1). After placement in the well, the packer was inflated using a bicycle pump (>20 PSI) to form a tight seal and keep the packer in place. The pressure was continually monitored to ensure isolation during sampling. Water was sampled with a low-flow peristaltic pump operating at 30–60 mL/min during purging. Field parameters of water temperature, specific conductance (SC), pH, and DO were monitored in separate beakers from both the water table and Zone 1 to monitor stabilization of parameter values during well purging. Additionally, a standard purge volume of at least 0.9 L was used to ensure removal of the standing water in the 6-in. isolated zone of the well and the pore volume of the sand pack surrounding the well screen (assuming an annular diameter of 7.25 in. and a saturated porosity of sand pack material of 0.25 percent). The purge rate was minimized to prevent the development of strong hydraulic gradients that might induce vertical flow through the sand pack along the well screen.

Aseptic procedures were used to install the packer and sample tubing. Following sample collection, the inflatable packer assembly was retrieved and sterilized by soaking in a 0.005 percent solution of sodium hypochlorite Clorox® bleach for 30 minutes, followed by soaking in sterile 0.005 percent solution of sodium thiosulfate and then sterilized deionized water (U.S. Geological Survey, variously dated). Precleaned and sterilized tubing were prepared for each well so that only the packer assembly necessitated sterilization in the field.

Field parameters were monitored in an open container exposed to the atmosphere using a Yellow Springs Instruments (YSI®) 600XLM data sonde. The sonde was calibrated before use as described in the National Field Manual (U.S. Geological Survey, variously dated). The SC sensor was calibrated daily with standard reference solutions (1,000 µS/cm and checked with solutions ranging from 250 to 750 µS/cm), and the DO sensor was calibrated daily using the air-saturated water method and occasionally verified with a zero-DO solution. The temperature probe was confirmed to ±0.2 °C through quarterly comparisons against a NIST-certified thermistor.

After purging, a raw sample was collected for nutrients and chloride into a pre-cleaned 1-liter Nalgene® bottle. Bacteria samples for analysis of *Escherichia coli (E.coli)* were collected into autoclaved 99 mL dilution bottles. All samples were immediately placed on wet ice for transport to a laboratory. Analysis of bacterial samples was initiated within 24 hours of sample collection. Sample processing consisted of filtering through a 0.45 µm membrane filter into polyethylene bottles for subsequent analysis of chloride and nutrients comprised of nitrate + nitrite, ammonia, total nitrogen, and phosphorus. Laboratory analyses of chemical constituents were done at the U.S. Geological Survey (USGS) National Water Quality Laboratory (NWQL) in Lakewood, Colorado. Analysis of bacteria samples were done at either the Washington Water Science Center laboratory in Tacoma, Washington, or at Exact Scientific Services in Bellingham, Washington (table 3).

Bacteria samples were analyzed by an enzyme substrate most-probable-number method using the IDEXX Colilert media Quanti-Tray® enumeration procedure for total coliforms and *E.* coli (Myers and others, 2007). Samples were incubated at 35±0.5 °C for 18–24 hours. Enumeration of *E. coli* was done using long-wave 366 nm ultra-violet light.

Table 3. Analytical parameters, method, reporting level, preservation, and analytical hold time for water samples.

[**Method:** Method identification used by the National Water Quality Laboratory (NWQL). **Abbreviations:** 1 µS/cm, 1 microsiemen per centimeter; °C, degrees Celsius; mg/L, milligram per liter; 1 mpn/100 mL, 1 most probable number of coliform forming units per 100 milliliters; mg-N/L, milligram of nitrogen per liter; <, less than; –, not applicable]

Analyte	Method	Instrument/method description	Reporting level	Preservation	Hold time (day)
		Field parameters			
Specific conductance	–	Ysi 600 xlm	1 µS/cm	Unfiltered, chilled	Immediate
Temperature	–	Ysi 600 xlm	°C	Unfiltered, chilled	Immediate
pH	–	Ysi 600 xlm	pH units	Unfiltered, chilled	Immediate
Dissolved oxygen	–	Ysi 600 xlm	mg/L	Unfiltered, chlled	Immediate
Escherichia coli	Iso 9308-2:2012	Idexx/colilert-18 defined-substrate	1 mpn/100 mL	Chilled	<1
		Laboratory analytes			
Nitrate plus nitrite	1-2547-11	Colorimetric, enzyme reduction	0.002 mg-N/L	Filtered, chilled	30
Ammonia as nitrate	1-2525-89, 1-2522-90	Colorimetric	0.010 mg-N/L	Filtered, chilled	30
Chloride	1-2057-85	Ion chromatography	0.06 mg/L	Filtered	180
Total nitrogen	1-2650-03	Alkaline persulfate digestion	0.05 mg-N/L	Filtered, chilled	30
Phosphorus	1-1630-85	Inductively coupled plasma atomic emission spectroscopy	0.022 mg-P/L	Filtered, acidified	180

Quality Assurance

Standard USGS quality-assurance and control procedures were incorporated throughout the sampling and analysis procedures to assure generation of data of known and acceptable quality (Wagner and others, 2007; Mueller and others, 2015). Potential bias in the data generated was assessed using laboratory blanks, field equipment rinse blanks, and references samples. Variability was assessed from an analysis of replicate laboratory and environmental samples.

The USGS Branch of Quality Systems (BQS) provides continuous checks of laboratory bias and variability of data generated by the NWQL through the regular submission of blind environmental samples. During this study, the NWQL analyzed 223 laboratory blank samples and 330 reference samples submitted by BQS. Results from the BQS blank samples submitted for analysis of nitrate indicated the minimal potential for laboratory bias from contamination in the analytical process, which had a laboratory quantification limit of 0.04 mg-N/L. Four percent of the 223 blank samples had reportable concentrations within five-fold of laboratory reporting limit (<0.2 mg-N/L), and one blank sample had a reported concentration greater than 1.0 mg-N/L. The average recovery of reference samples was 100.4 percent, with six samples exceeding the target recovery range of 80–120 percent. Environmentally relevant concentrations in this study were more than two times larger than the laboratory reporting limit; thus, potential laboratory bias is not considered to affect study results significantly.

Field quality-control samples submitted blind to the analyzing laboratory along with the monitoring samples included 35 field equipment blanks, 53 replicate environmental samples, and 21 reference samples of known composition. Field equipment rinse blanks included samples of deionized water collected through the sampling equipment after field-sterilization of the inflatable-packer assembly using sodium-hypochlorite. Measurable chloride concentrations less than 1.0 mg/L were common in field equipment blanks and were within project quality-assurance target goals. In the field blank of one sample set, chloride was measured at 14.1 mg/L, suggesting incomplete rinsing of the packer assembly following field sterilization. Because of that contaminated field blank, chloride data for that sampling event was qualified as estimated values. The corresponding concentration of nitrate in the equipment field blank sample was less than laboratory reporting levels of 0.04 mg-N/L. Analysis of field-blank sample results using procedures for skewed data described by Mueller and others (2015) and Hahn and Meeker (1991) provide confidence measures of the minimum measurement levels for which 90 percent of the data generated are unaffected by potential bias (table 4). Environmentally relevant concentrations of constituents in this study greatly exceeded (>10 times) the range of contamination typically existing in equipment blank samples. Reference solutions for nitrate analysis, at concentrations ranging from 5.0 to 50 mg-N/L, were submitted in triplicate with environmental samples on 17 sampling events. The percent recovery of the expected reference solution concentration ranged from 95 to 104 percent, with an average of 99.8 percent. The average standard deviation of the triplicate analysis was 0.17.

Sampling variability was assessed to distinguish differences between measured concentrations resulting from laboratory measurement and environmental temporal variability. In this study, differences between replicate or paired samples, presumed to have identical concentrations, were used to assess laboratory measurement variability in water-quality concentration (table 5). The relative percent difference (RPD) of 63 replicate analyses of nitrate plus nitrite averaged 1.2 percent, but ranged up to 17.7 percent. The median RPD of replicate samples collected sequentially was 1.2 for nitrate and 0.7 for chloride. During the early phase of the study, several wells were sampled on repeated days to

Table 4. Summary of dissolved constituent concentrations in field blank samples and calculated upper confidence limit of inherent contamination associated with sample collection of groundwater samples using an inflatable packer system to isolate water table zone, Whatcom County, Washington.

[**Abbreviation:** mg/L, milligram per liter]

Constituent	Reporting level (mg/L)	Number of blank samples	Number of values less than the reporting level	Range of quantified values (mg/L)	Concentration of 90th percentile (mg/L)	Evaluated percentile of contamination	Achieved level of confidence
Ammonia	0.01	30	19	0.01–0.07	0.07	90	95.8
Nitrate	0.04	30	26	0.04–0.09	0.90	90	95.8
Phosphorus	0.01	30	27	0.06–0.11	0.11	90	95.8
Chloride	0.10	29	12	0.06–14.1	0.9	90	81.6

Table 5. Summary of measurement variability of selected water-quality constituents, Whatcom County, Washington.

[**Abbreviations:** mg/L, milligram per liter; BQS, U.S. Geological Survey Branch of Quality Systems]

Constituent	Detection level (mg/L)	Number of replicate pairs measured	Percentage of uncensored concentrations	Median relative percentage of difference
Recovery of expected concentrations of blind reference samples (BQS)				
Chloride	0.1	634	100	2.3
Ammonia	0.01	1,232	100	2.6
Nitrate	0.1	354	100	2.3
Total nitrogen (persulfate)	0.1	648	100	3.0
Phosphorus	0.1	212	100	1.8
Environmental replicate samples (this study)				
Chloride	0.1	51	100	0.7
Ammonia	0.01	50	37	0.0
Nitrate	0.1	63	100	1.2
Total nitrogen (persulfate)	0.1	48	100	2.5
Phosphorus	0.1	52	13	10.9

further assess difference from small-scale temporal and spatial variation that might be associated with placement of the packer assembly and sampling variability, and relative percent difference in analyzed concentration was less than 5 percent.

All field and laboratory data collected for this study were stored, archived, and made available at the U.S. Geological Survey National Water Information System database and in Cox and others (2016).

Statistical Analysis

Water-quality concentrations at the water table were not normally distributed; therefore, non-parametric statistical tests were used to compare data from ARM and CON treatment wells to evaluate if differences in manure application timing strategy affected differences in chemical concentrations at the water table. The analysis intended to determine if ARM-based manure application scheduling resulted in a significant difference in nitrate concentrations originating in the pore water leaving the vadose zone and measured in the water table. Each of the three field sites with paired plots had slightly different land management details, soil properties, and depths to water. Also, site D only received one differing manure application between the CON and ARM treatment plots during the study (autumn 2014) due to field conditions and the short length of the study period at that site. Therefore, comparisons of aggregated data from all sites were not instructive. Instead, water-quality concentrations at the water table beneath the ARM and CON treatment plots in individual sites were compared.

Differences in treatment regime and chemistry outliers necessitated the use of data screening. For site B, only data taken after March 1, 2013, were used for analysis to account for the switch in treatment plots. This date was chosen to correspond with the end of the recharge season and was confirmed by consistently declining water table levels after this date. For site C, data from all wells except well C5 were used in the analysis. The anaerobic conditions measured in well C5 indicated it was an outlier compared to oxygenated aerobic water-table conditions found in all other monitoring samples (for detailed description see section, "Water-Quality Concentrations Measured at the Water Table").

A non-parametric Mann-Whitney test was used to determine if significant differences were present in nitrate, total nitrogen, and chloride concentrations between the ARM and CON treatment plots at each site. Analyses were run using all the data, along with a separate set of analyses using only seasonal data from the recharge period October 1 through the following March 31, for all years for which data were collected. To further understand spatial variability of nitrate, total nitrogen and chloride concentrations, a non-parametric Kruskal-Wallis analysis of variance was used to test for significant differences among wells followed by a post-hoc Conover-Iman pairwise test to determine significant differences between individual wells.

Nitrate Loading Calculation

Nitrate loading to groundwater was estimated to provide information on the annual and seasonal rates of nitrate leaching from the soil zone. Nitrate concentrations measured at the water table were assumed to represent recent recharge without significant mixing with lateral groundwater flow. Nitrate loading was estimated based on the volume of recharge passing through the soil zone and the concentration of nitrate measured in recently recharged groundwater samples collected at the water table. Nitrate concentration reported from the laboratory in milligrams nitrogen per liter was converted to traditional farm scale units of pounds per acre-inch from the relation (assuming groundwater density of 1.0 g/mL):

$$1\frac{mg}{L}\times 2.2028\times 10^{-6}\ \frac{lb}{mg}$$

$$\times 1.028\times 10^{5}\ \frac{L}{ac-in}=0.226\frac{lb}{ac-in} \qquad (1)$$

At the field scale of an acre, each milligram per liter of nitrate measured in groundwater at the water table to a water-equivalent depth of 1 in. (allowing for the porosity of aquifer material) requires a nitrate leachate mass of 0.226 pound. An equivalent calculation is embedded in NRCS Nutrient management standard Nutrient Budget worksheet to calculate input of nitrogen from irrigation water (U.S. Department of Agriculture, 2014b). Annual mass loading was calculated from the annual average nitrate concentration at each site, and estimated annual recharge was determined by Cox and Kahle (1999). Annual nitrogen loading was estimated for a recharge period from July 1 through June 30.

Monthly nitrate loading at each site was estimated from monthly nitrate concentrations and monthly estimates of potential recharge based on soil water balance data. Monthly recharge was estimated as the positive sum of the daily water balance calculated from precipitation minus potential evapotranspiration (fig. 2). Recharge was assumed to be zero during months with a negative soil water balance sum. Potential evapotranspiration was calculated as the average of reference evapotranspiration calculated using the Penman-Monteith equation (Allen and others, 2005) for grass crop at 12 and 50 cm height. Monthly nitrate concentrations were averaged or interpolated from adjacent months. Precipitation and reference evapotranspiration data used to calculate monthly recharge were obtained for local weather stations (Ten Mile and Lynden [Washington State University, 2018] and Clearbrook, Washington [Western Regional Climate Center, 2018]). Precipitation data from Ten Mile were used directly at sites B and D. Precipitation data for site C were taken as the average of precipitation measured at Lynden and Clearbrook. The variation in reference evapotranspiration data for stations at Ten Mile and Lynden was comparatively small and not available for the Clearbrook weather station, so evapotranspiration data for Ten Mile was used in all calculations.

Variation of Water-Level Altitude and Nutrient Concentration at the Water Table

Water Levels

Annual precipitation during the study period ranged from 92 to 109 percent of the long-term average for the Abbottsford weather station, suggesting that hydrologic conditions and variations in groundwater altitudes in the surficial aquifer were typical of average conditions for the study area. The pattern of seasonal water-level variations was similar between sites with high water levels typically occurring in mid-March to late May and lows in late September–October (fig. 7). The range in water-level altitudes in individual wells was 2.6–5.3 ft at site B; 5.5–8.5 ft at site C; and 2.3–4.3 ft at site D. Water-level contours for sites B, C, and D (figs. 3, 4, and 5) show more variation at sites B and D than at site C. The direction of groundwater flow, inferred as perpendicular to water-level contour lines, has greater variability at sites B and D. At site C the water level contours are similar in shape at periods of both high and low groundwater level altitudes indicating direction of groundwater flow is less variable. Likewise, hydraulic gradients and, hence, groundwater velocities, vary spatially and temporally both within and among sites. Water-level gradients were higher during periods of rising water levels. At site B, average calculated groundwater-flow velocity ranged from 2.3 to 7.9 ft/d, and at site C, average calculated groundwater-flow velocity ranged from 20 to 36 ft/d. At site D, water-level contours indicate average calculated groundwater-flow velocity of 3.2–4.7 ft/d during rising water-level periods. The water-level altitude declined below the screen zone of several wells at site D during the summer low water-level period.

Spatial variation in water levels was observed among different wells at sites B and C and was consistent with topographic variations of the test plots. Areas adjacent to the test plots were more steeply sloped downhill, which lowered water-level altitudes and increased the thickness of the unsaturated zone. Water-level altitudes were larger at upslope wells with higher land-surface altitude. At site B, the water level was highest in well B1, where the land surface was highest, and lowest at well B5, where the land surface was lowest. Differences were smallest during the summer

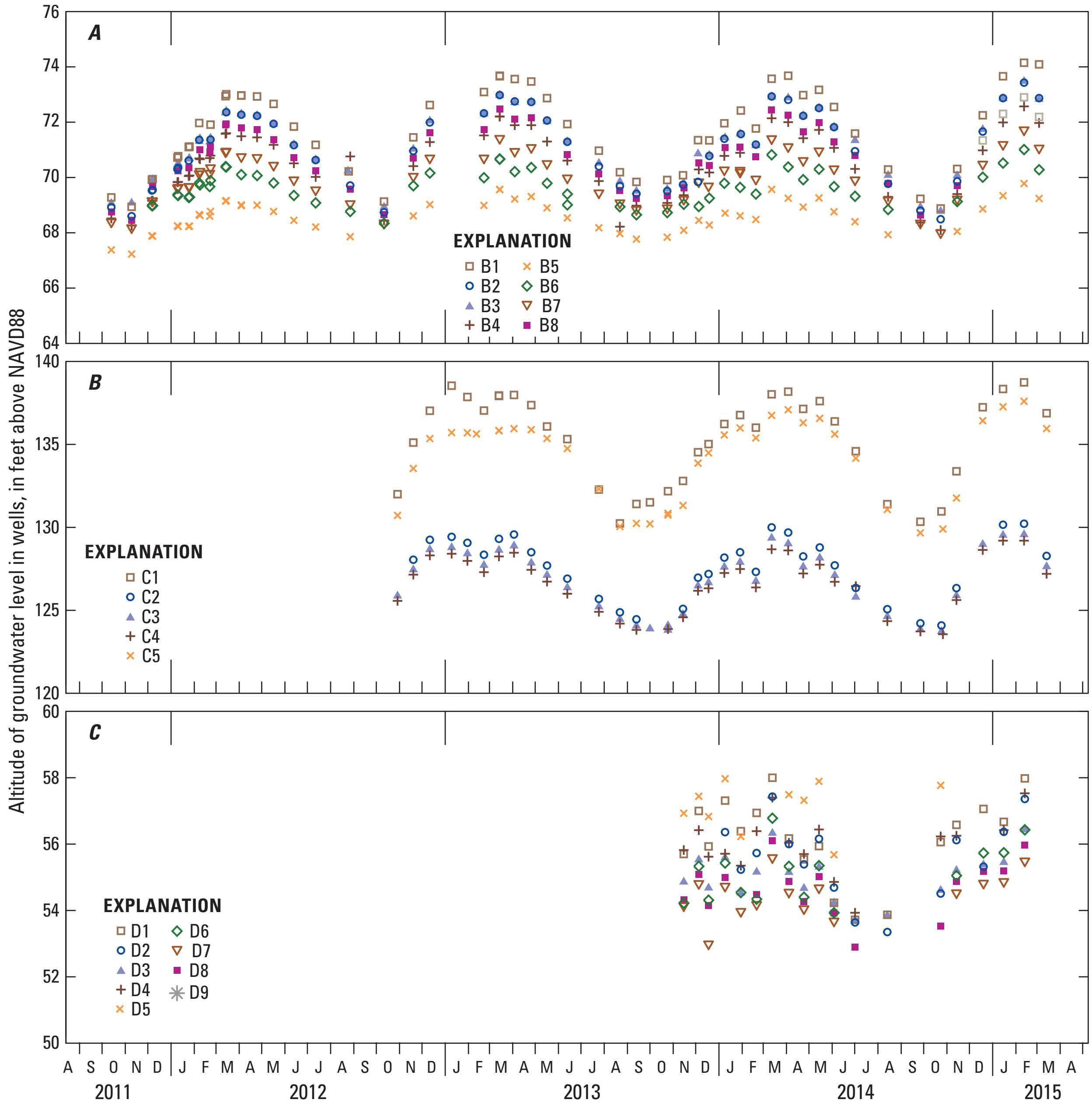


Figure 7. Water-level altitudes in monitoring wells at sites (*A*) B1–B8, (*B*) C1–C5, and (*C*) D1–D9, Whatcom County, Washington. Symbols indicate individual wells shown in figures 3–5.

period (about 2 ft) and greatest in mid-winter (about 5 ft). An increased horizontal hydraulic gradient is present at times of increased differences in groundwater levels (fig. 7). At site C, water levels in the two upslope wells (C1 and C5) were typically 5 ft higher than the downslope wells. Topography in the field area adjacent to wells C2, C3, and C4 sloped away substantially from the edge of the test plots at site C.

Water-table fluctuation at all sites was less than might be anticipated based on annual recharge rates of 26–30 in/yr. However, given the porosity of the sandy aquifer material, which ranges from 25 to 30 percent, and the location of the field sites near topographic high points, groundwater would be expected to flow away from these areas at all times. Thus, the measurement of seasonal water-table fluctuations in this setting would underestimate annual recharge. The seasonal pattern of groundwater level change coincided with the seasonal patterns of precipitation and natural net soil-moisture (fig. 2). Rising groundwater levels begin after the onset of increasing frequent precipitation and declining levels of potential evapotranspiration that typically begin in late

September to early October. Water levels typically began to rise in late October or early November, although this varied with depth to groundwater. This lag between the onset of more frequent precipitation and rising water-level altitude is the result of moisture accumulation to saturation levels in the soil zone resulting in transport through the vadose zone. Water levels rose rapidly through late autumn (November–December) and remained near maximum altitudes during much of the winter period before beginning to steadily decline during March–April. Water levels continued to decline through the spring and summer (fig. 7). Overall, the seasonal variations in water levels observed during the test period at sites B, C, and D were similar with previously reported water-level variations for surficial aquifer in the area (Cox and Kahle, 1999; Carey and Harrison, 2014).

Water-Quality Results

Nitrate concentrations in samples of groundwater from near the water table in all wells from both test plots at all field sites were consistently larger than average concentrations reported in groundwater from the SBA. These results indicate that nitrate was being leached from the soil zone and present in water recharging the SBA aquifer beneath fields receiving manure application using either the ARM or CON manure application scheduling procedures. Concentrations of nitrate, total nitrogen, ammonia, phosphorus, chloride, and *E. coli* measured in groundwater samples at the water table were highly variable (table 6). Concentrations of nitrate ranged from less than 0.1 to 116 mg-N/L. More than 90 percent of the measured nitrate values exceeded 10 mg-N/L, the EPA’s MCL for nitrate. At sites B and D, the median concentration exceeded the MCL by 2.5 and 4.0 times, respectively. Concentrations of ammonia, phosphorus, and *E. coli* were typically low, often at or near the laboratory reporting level. Ammonia was not detected at concentrations greater than 0.01 mg-N/L in 80 percent of samples analyzed, and exceeded 1.0 mg-N/L in only 5 of 726 samples analyzed. The concentration of chloride ranged from less than 1.2 to 153 mg/L, with a median value of 14.6 mg/L. The few detections of phosphorus and *E. coli* at the water table indicated that these constituents were largely retained in the unsaturated zone and were not being transported with recharge. The pH values were typically near 5.8, as is expected of shallow water table conditions in agricultural settings. Dissolved oxygen concentrations were typically in the range of 10 mg/L, measured in an open container exposed to the atmosphere. However, measurements of DO at well C5 ranged from 2 to 5 mg/L, indicating that DO concentrations at the water table were smaller than expected for saturated atmospheric conditions. Complete results are available in the companion data release (Cox and others, 2016).

In the groundwater sampled beneath the test plots, nitrogen was present almost exclusively in the nitrate form indicating nearly complete oxidation of any nitrogen in manures applied to fields that reached the water table . The plot of nitrate and total nitrogen analyzed by persulfate oxidation shows an almost perfect one-to-one correlation (fig. 8) at all three of the field sites. About 50 percent of the nitrogen in manure that is applied to the field is in the ammonium-N form, which is either volatilized or microbially converted to nitrate before arrival at the water table. When ammonia was detected in groundwater samples, concentrations were almost always less than 1.0 mg-N/L and composed less than 5 percent of the nitrogen present in the water sample. The ratio of nitrate-to-chloride (NO_3/Cl) in samples co-varied because both are components in manure. Chloride is more conservative in soil moisture because comparatively small amounts of chloride are taken up by plants compared to nitrogen, which can also be lost to the atmosphere by volatilization. Comparison of NO_3/Cl ratios were generally consistent among wells at each study site with occasional outliers suggesting general uniformity in the composition of dairy manure source material. At site B, median NO_3/Cl ratio was 1.9, and the range between the 10th and 90th percentile of nitrate-chloride ratios was 3.4–0.97, similar to ranges observed in studies of shallow groundwater beneath dairy forage fields in Thurston and Whatcom Counties (Erickson and Mathews, 2002; Carey and Harrison, 2014). At sites C and D, the distribution of NO_3/Cl ratios were similar (including occasional outliers) to the pattern measured at site B, though median values were smaller at 1.1 and 0.94, respectively. However, at well C5, NO_3/Cl ratios were often undefined, as nitrate concentration decreased to less than the detection level.

Comparison of Water Quality in Paired Water Table and Zone 1 Samples

Persistent and variable differences in specific conductance, nitrate, and chloride concentrations were measured in samples collected simultaneously from the water table and Zone 1, indicating that the packer system reliably isolated the water table sample from the saturated screened zone below. Differences in measurements of nitrate and chloride between paired samples from the water table and Zone 1 exceeded measurement resolution of field and laboratory instrumentation as well as analytical variability generated from analysis of environmental replicate samples (table 5). Large differences in nitrate and chloride concentrations were observed between paired samples from water table and Zone 1 (see well B2 as an example, fig. 9). Chloride measurements provided the greatest resolution between water table and Zone 1, with 59 percent RPD. The RPD for nitrate and specific conductance between water table and Zone 1 was 36 and 18 percent, respectively. Laboratory analysis of the Zone 1 samples was discontinued in June 2014, and specific conductance measurements were used to confirm sample isolation. The vertical stratification of groundwater quality reported here was previously observed in similar hydrologic and land-use settings in the SBA (Dasika, 1996; Kuipers and others, 2014).

Table 6. Summary statistics of nitrogen and other groundwater-quality data collected near the water table at the study sites, Whatcom County, Washington, October 2011–March 2015.

[**Site:** Location of sites shown in figure 1. **Laboratory reporting level:** Smallest concentration of a substance that can be reliably measured by using a given analytical method for typical analysis; may be higher for some analysis runs. **Percentage of uncensored concentrations:** Analytical results less than the laboratory reporting level are considered censored values. **Range and median of measured concentrations:** Concentrations in milligrams per liter unless noted as mpn (most probable number) for *Escherichia coli* (*E. coli*). **Abbreviations**: <, less than; mpn, most probable number of coliform forming units per 100 milliliters]

Site	Constituent	Laboratory reporting level	Number of measured samples	Percentage of uncensored concentrations	Measured concentrations, in milligrams per liter	
					Range	Median
B	Nitrate	0.1	414	100	5.34–88.9	24.6
B	Total nitrogen	0.1	355	100	5.57–86.6	23.9
B	Ammonia	0.01	401	42	<0.01–39.1	<0.01
B	Phosphorus	0.1	354	9	<0.1–13	<0.1
B	Chloride	0.1	414	100	1.18–82.1	14.6
B	*E. coli*	1 mpn	136	2	<1–28 mpn	<1.0
C	Nitrate	0.1	172	95	<0.1–60.7	11.4
C	Total nitrogen	0.1	143	100	<0.25–60.6	9.34
C	Ammonia	0.01	169	38	<0.01–1.68	<0.01
C	Phosphorus	0.1	137	6	<0.1–0.12	0.01
C	Chloride	0.1	171	100	1.99–37.3	11.8
C	*E. coli*	1 mpn	49	2	<1–2 mpn	<1.0
D	Nitrate	0.1	158	96	<0.4–116	40.5
D	Total nitrogen	0.1	127	98	0.6–109	42.2
D	Ammonia	0.01	156	72	<0.01–6.68	0.02
D	Phosphorus	0.1	123	35	<0.1–1.35	<0.1
D	Chloride	0.1	158	100	7.04–153	47.3
D	*E. coli*	1 mpn	56	38	<1–54 mpn	<1.0

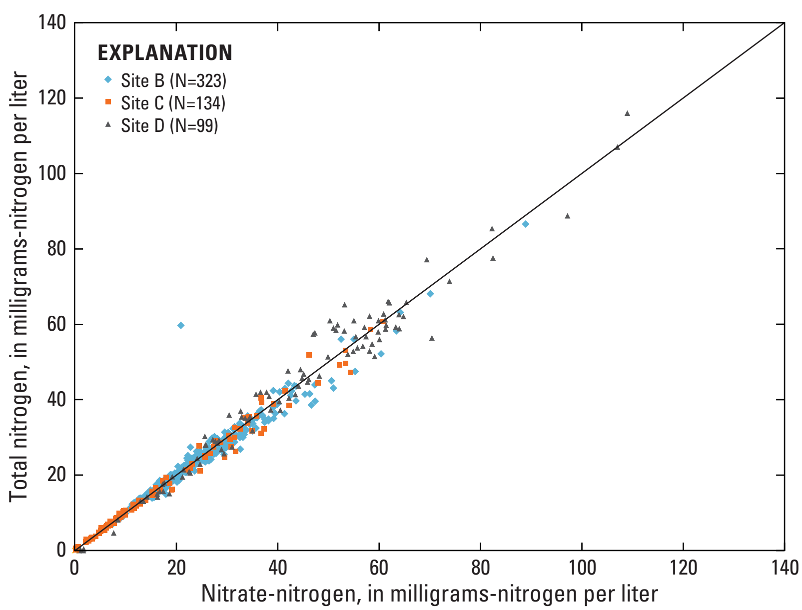


Figure 8. Concentrations of nitrate-nitrogen and total nitrogen (persulfate digestion) along one-to-one correspondence line in water-table samples from beneath forage fields in sites B, C, and D, Whatcom County, Washington. N, number of samples.

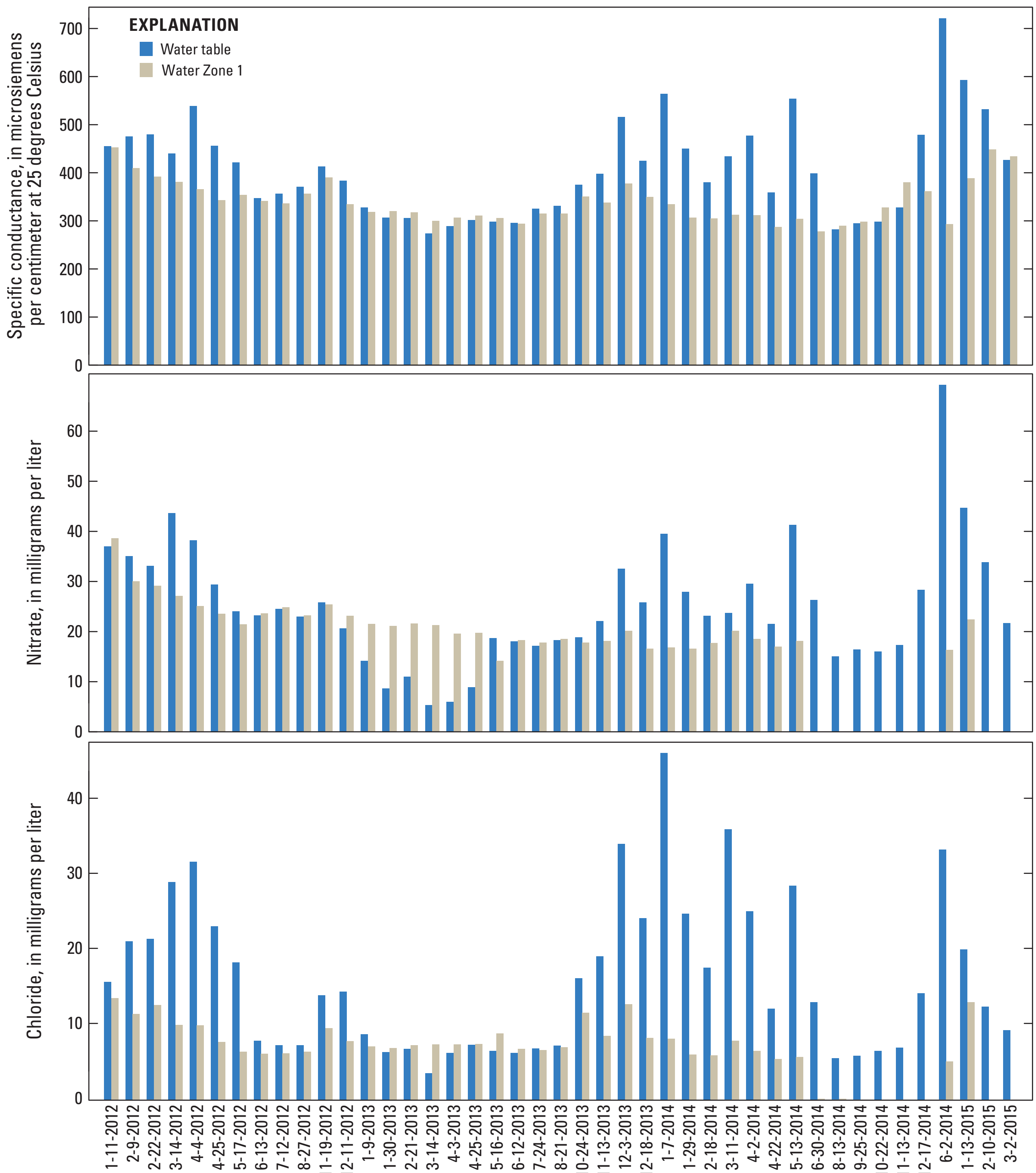


Figure 9. Comparison of specific conductance, nitrate, and chloride concentrations measured in samples from the water table and Zone 1 at well B2, Whatcom County, Washington, January 11, 2012–March 2, 2015.

Although differences in water table and Zone 1 water quality were measured consistently in wells at each site, the pattern of variation was often dissimilar from well to well even within the same test plot (an example of this difference can be seen in fig. 10). For most wells, chloride concentrations in water table and Zone 1 samples are similar during the summer months. However, autumn groundwater concentrations typically rise in both water table and Zone 1, often very synchronously (for example, well B3), but not always (for example, well B7). At well B2, there was minimal variation in Zone 1, while the water table zone showed a doubling in concentration during the winter period of 2012 and 2014. The lack of a consistent pattern may result from the large variability in soil properties that affect the transport and leaching of soil water (Biggar and Neilsen, 1976). Additionally, small differences in the length of water table isolated by the packer, differences in the contributing zone of the aquifer open to Zone 1, or spatial variation in concentrations of constituents in water recharging the aquifer, may have also contributed to variability. Despite these factors, consistent seasonal patterns were generally present in all wells at all sites. Differences in concentrations between the water table and Zone 1 were generally smallest during the summer months when soil moisture deficits were greatest and potential for groundwater recharge was smallest (see well B2 as an example, fig. 8). Larger water table concentrations were often measured in the autumn after the start of the seasonal rainy period. It is expected that water table concentrations are greatest at the beginning of recharge events following periods of high evapotranspiration when soluble salts may have accumulated in the soil zone. Zone 1 concentrations did not show this temporal variability, and differences in concentrations measured at the water table and Zone 1 were greatest during this time.

Water Quality Concentrations Measured at the Water Table

Concentrations of nitrate and chloride at the water table beneath test plots were highly variable. Concentrations of nitrate ranged from non-detect to 116 mg-N/L, and chloride ranged from 1.15 to 153 mg/L. The ranges of nitrate and chloride concentrations for ARM and CON treatment wells are presented in figure 11. Concentrations were more similar among wells at individual sites and show greater variability between sites. For both nitrogen and chloride, concentrations were generally lower at site C and higher at site D, as shown in summary statistics in table 6.

Concentrations of nitrate and chloride in well C5 were much less variability and are typically smaller than measured in other wells, suggesting different conditions were present at that well. Other differences in measured concentrations in well C5 include lower dissolved oxygen and measurable ammonia. Interestingly, at the time of installation of well C5, and for the next 11 months, concentrations of nitrate at the water table in well C5 were similar to those in other wells at site C, around 8.0 mg-N/L nitrate and total-nitrogen, along with non-detectable concentrations of ammonia. However, in the early summer of 2013, ammonia began to be consistently detected at concentrations less than 0.2 mg-N/L, and then in the autumn a decrease and then absence of nitrate. During the winter 2014, concentration of nitrate increased to levels similar to other wells at site C. Dissolved oxygen concentrations were roughly one-half those detected at other site C wells. Together, these data suggest that microbial denitrification was occurring intermittently in the groundwater upgradient of this well. Occasionally, these conditions were observed in single samples at site D, but not consistently in any of the other wells; as such, well C5 was excluded from the comparison of ARM and CON treatment plots at site C.

Temporal Variation in Groundwater Quality

Large seasonal variations were measured in concentrations of nitrate and chloride at all three sites. Chloride is a conservative water-quality constituent that is associated with cattle manure and as such provides a solute tracer of nutrients from manure (Chang and Entz, 1996). The pattern of variation in the concentrations of chloride and nitrate in water table samples were generally similar in both the ARM and CON treatment wells, although differences were apparent in some individual wells (figs. 12–16). Concentrations of nitrate and chloride at the water table were typically lowest in summer and highest in winter. Periods of rapidly increasing concentrations of chloride and nitrate were often nearly synchronous with periods of rising groundwater levels associated with precipitation-mediated groundwater recharge. The largest concentrations measured in groundwater often occurred in samples collected in the period surrounding the highest seasonal water levels. Winter concentrations of both nitrate and chloride at the water table were often larger by one-fold or more than summer concentrations. At all three sites, average nitrate concentrations in shallow groundwater were lowest at the end of the growing season and increased rapidly with the onset of seasonal rains in October.

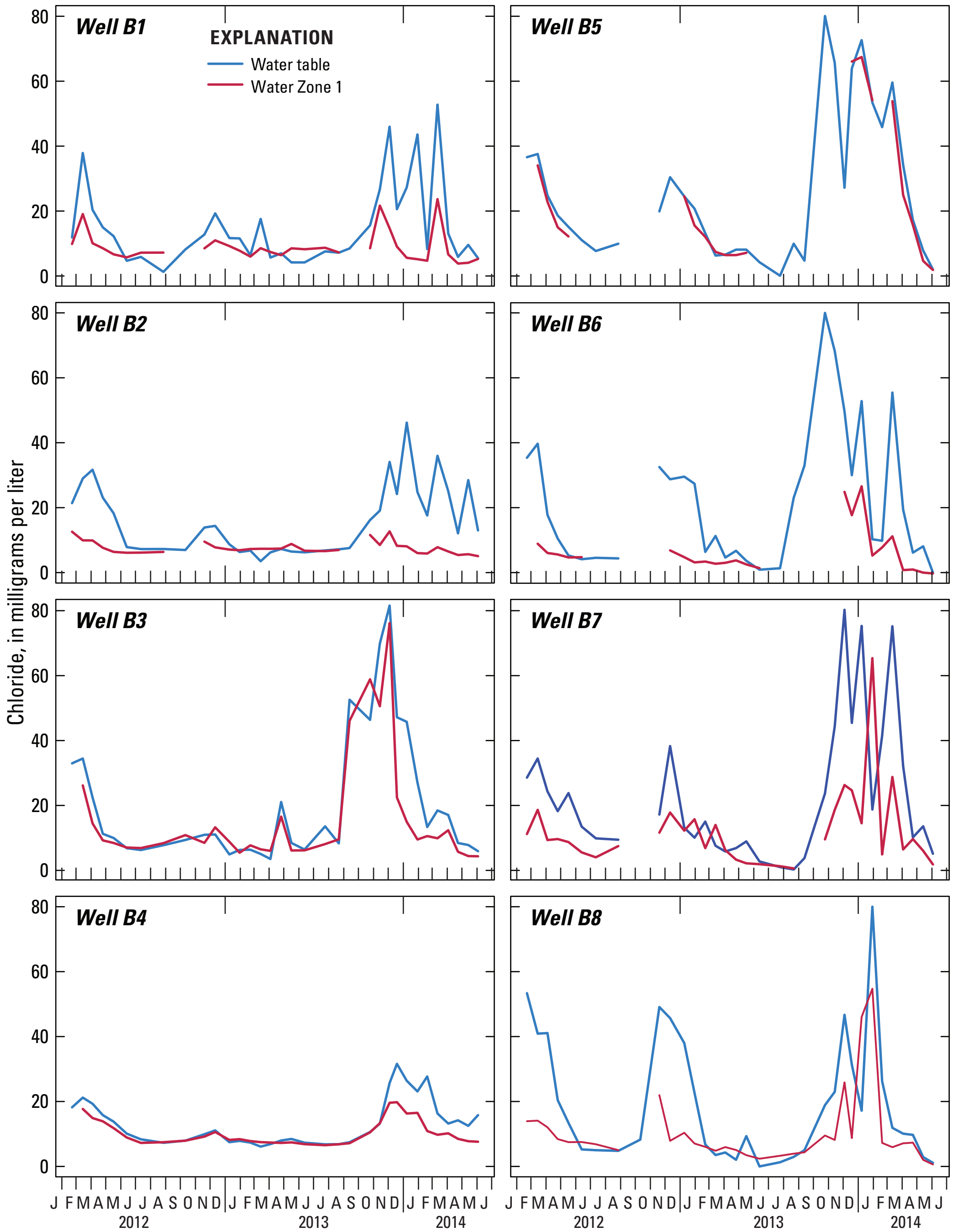


Figure 10. Chloride concentrations in the water table and Zone 1 from individual wells at site B, Whatcom County, Washington. Wells B1–B4 were in the conventional management plot and B5–B8 were in the application risk management plot.

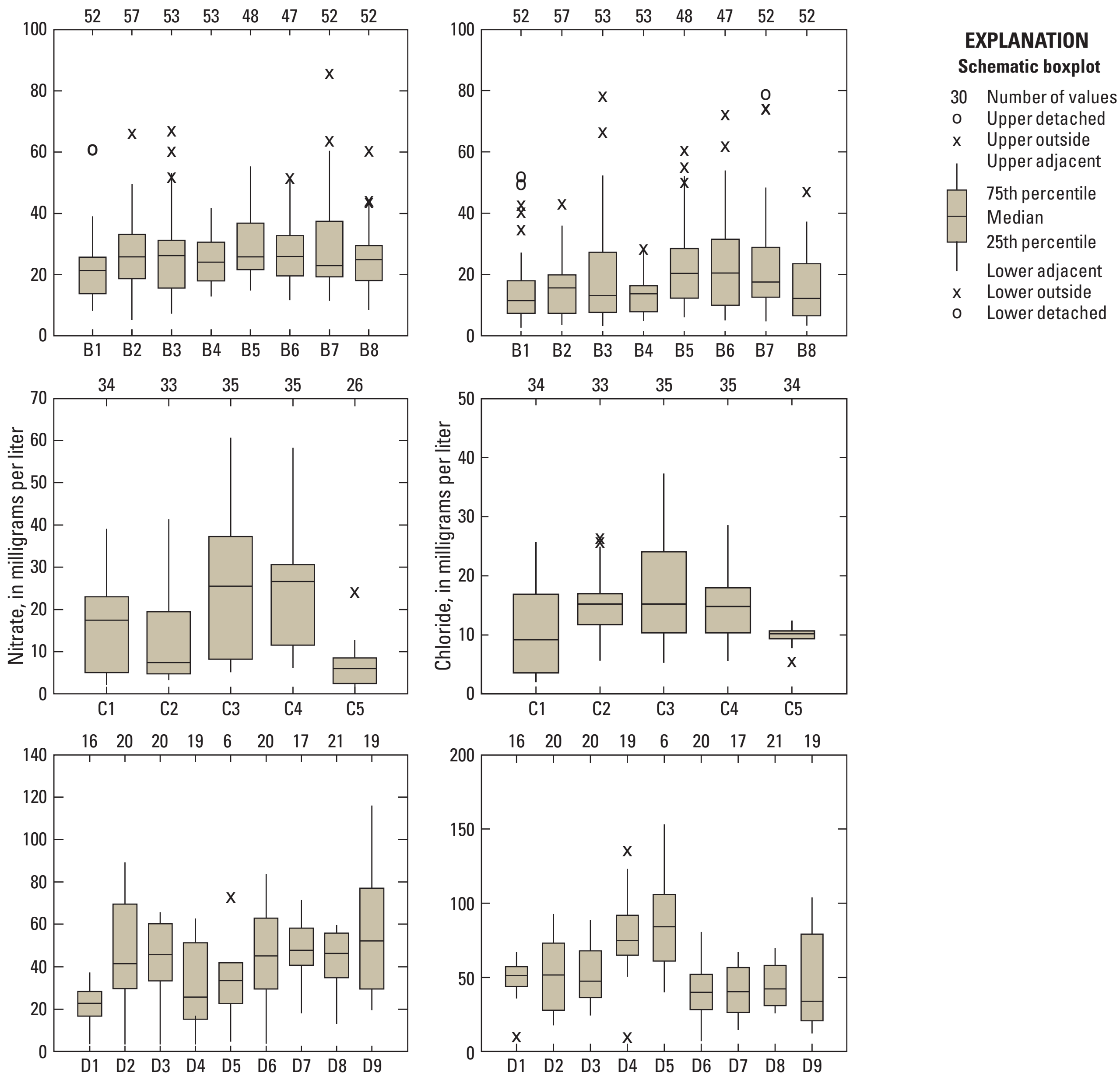


Figure 11. Nitrate and chloride concentrations from the water table zone (top 6 inches of the aquifer) of individual wells from three field sites, Whatcom County, Washington. Locations of the field sites are shown in figure 1.

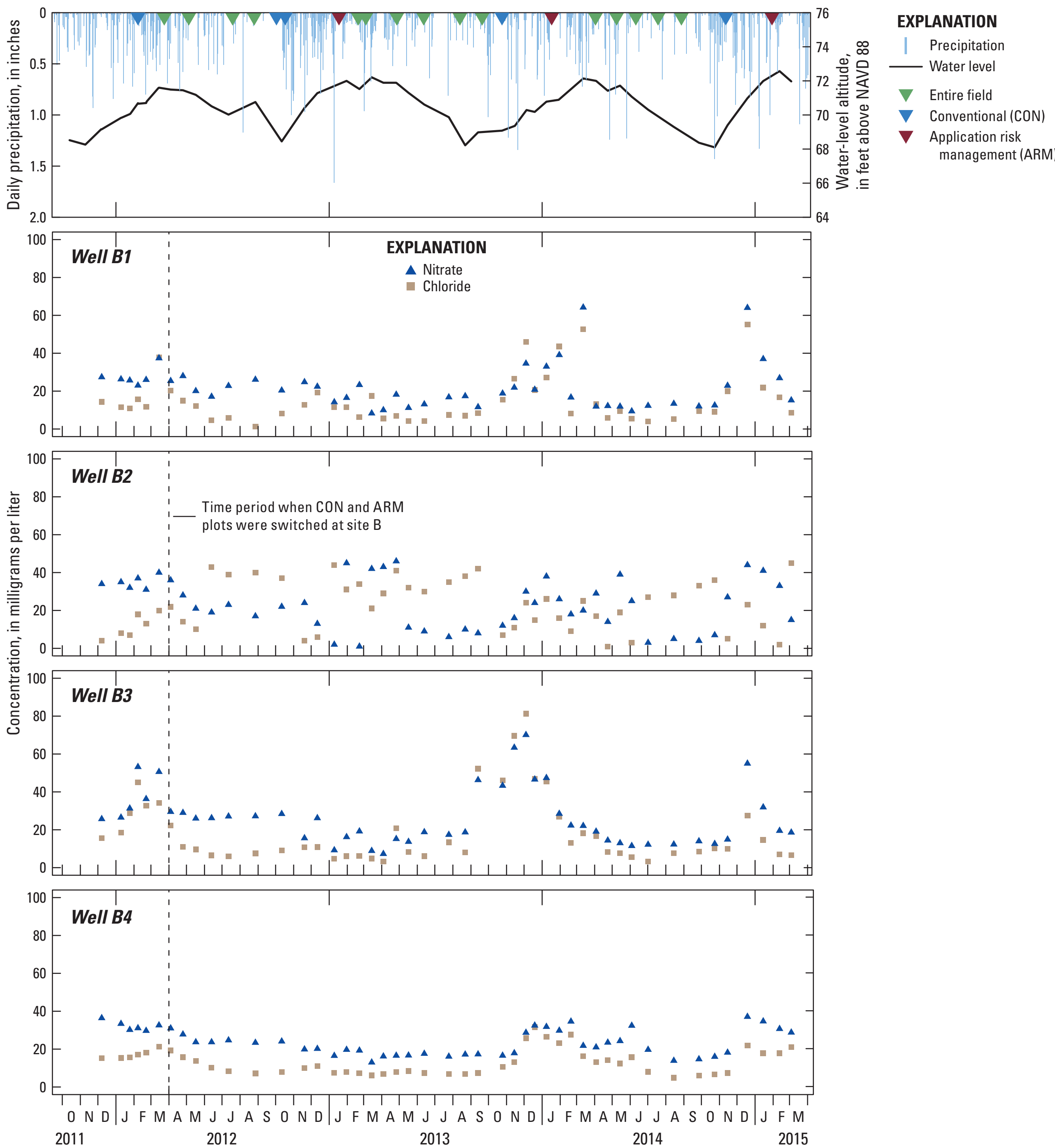


Figure 12. Time series plot showing (top) daily precipitation at Ten Mile weather station and plots showing manure application dates, water level (well B4), and nitrate and chloride concentrations in four monitoring wells at field site B, Whatcom County, Washington. Triangles in top plot represent manure application date and location by color. Wells B1–B4 were under conventional management from November 2011 to March 2012 and under application risk management from April 2012 to March 2015.

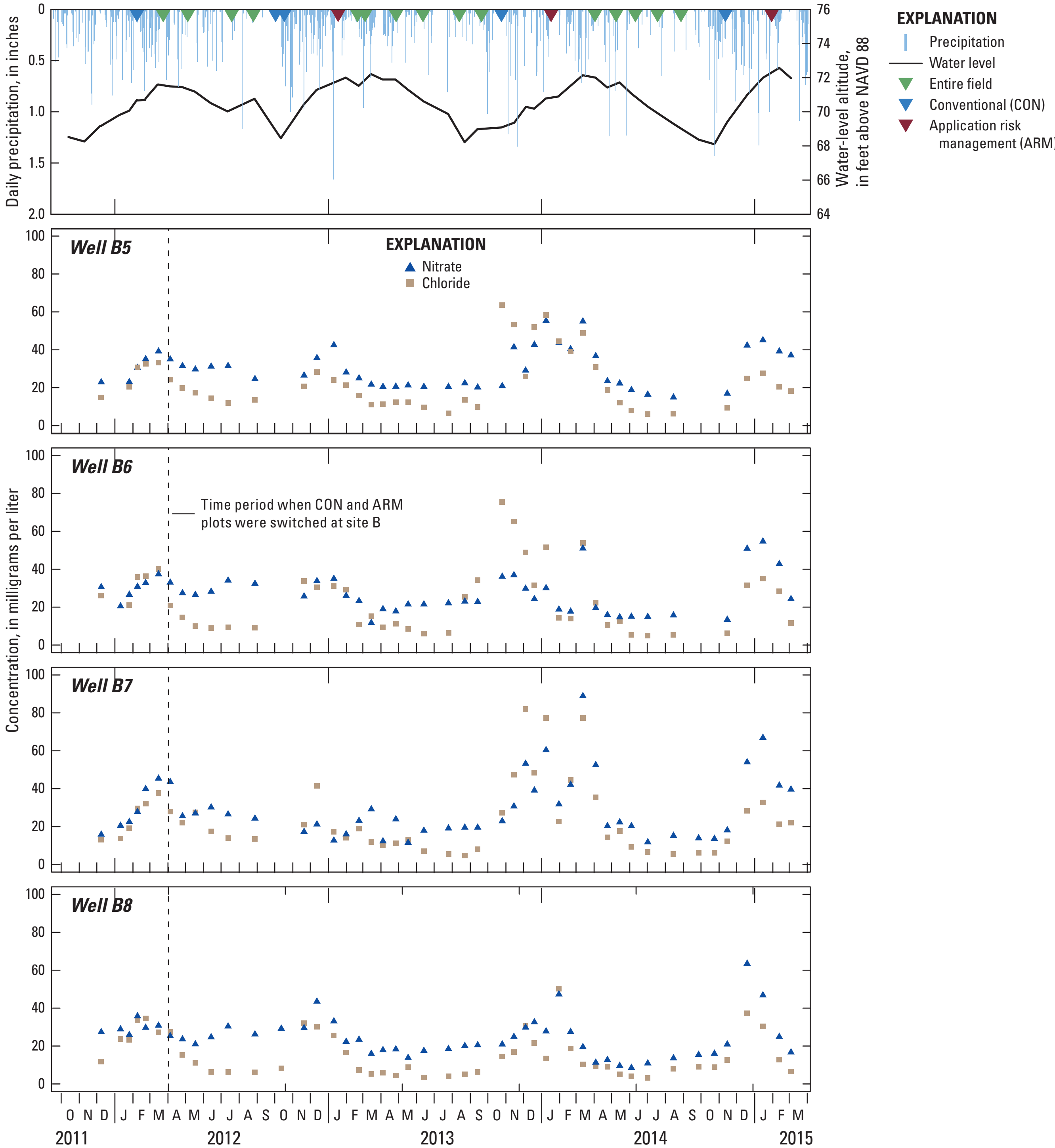


Figure 13. Time series plots of (top) daily precipitation (Ten Mile weather station), manure application dates, and water level (well B4), and nitrate and chloride concentrations in four monitoring wells at field site B, Whatcom County, Washington. Triangles in top plot represent manure application date and location by color. Wells B5–B8 were under ARM treatment from November 2011–March 2012 and under CON treatment from April 2012 to March 2015.

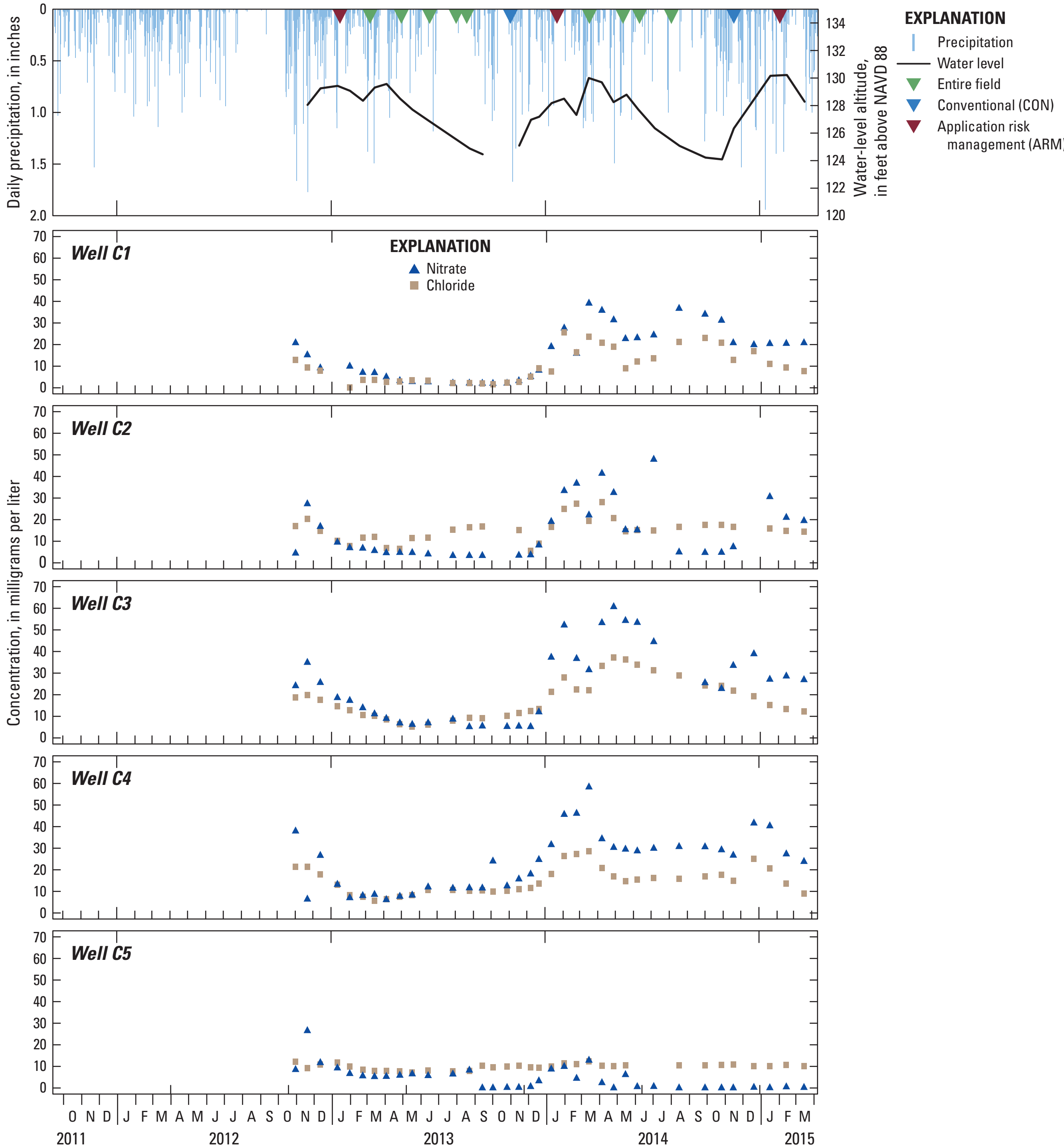


Figure 14. Time series plots of (top) daily precipitation (Clearbrook weather station, Western Region Climate Center; http://www.wrcc.dri.edu/cgi-bin/cliMAIN.pl?wa1484), manure application dates, and water level (well C2), and nitrate and chloride concentrations in five monitoring wells at field site C, Whatcom County, Washington. Triangles in top plot represent manure application date and location by color. Wells C1–C2 were under CON treatment and wells C3–C5 under ARM treatment.

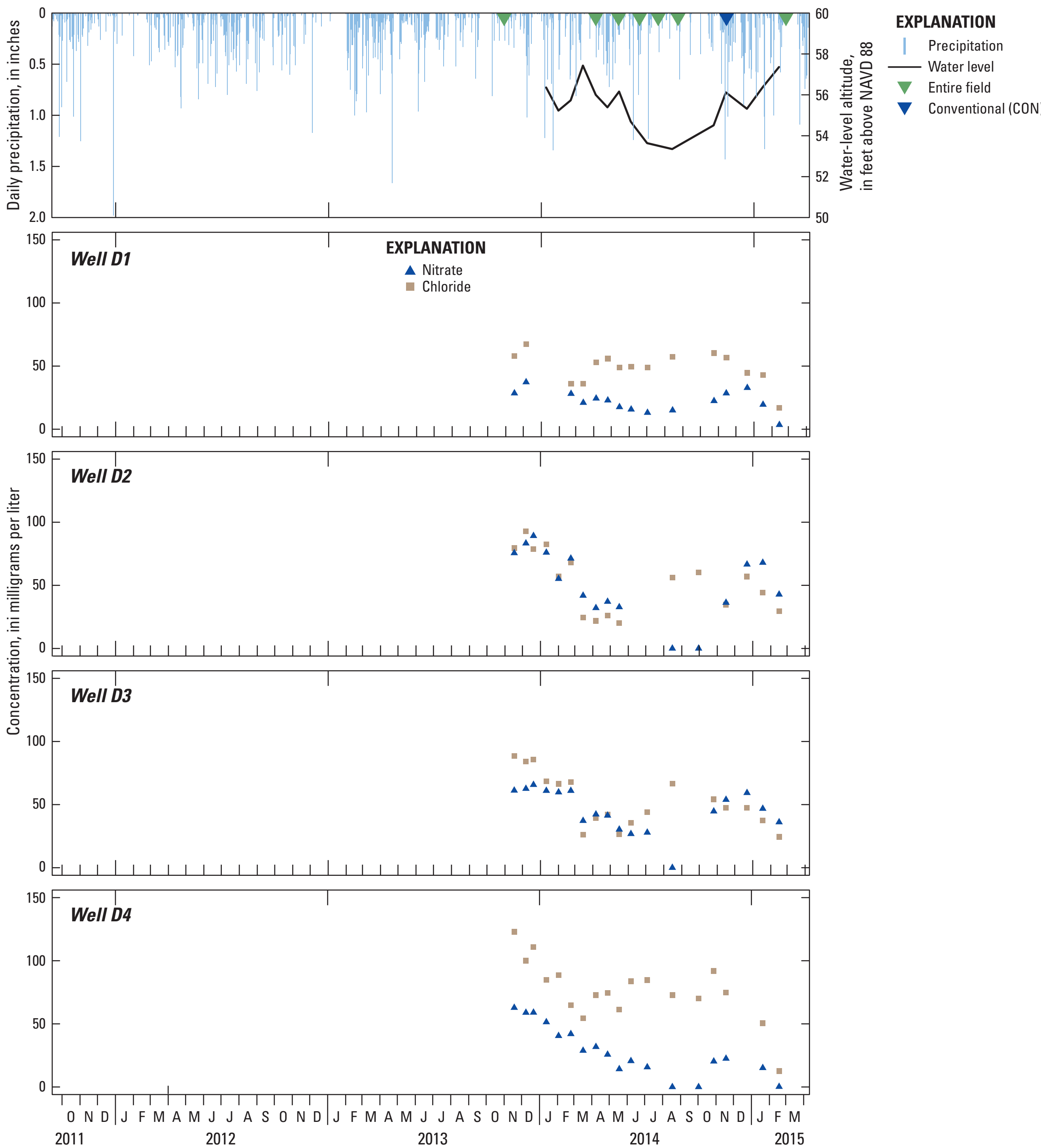


Figure 15. Time series plots of (top) daily precipitation (Ten Mile weather station), manure application dates, and water level (well D2), and nitrate and chloride concentrations in four monitoring wells at field site D, Whatcom County, Washington. Triangles in top plot represent manure application date and location by color.

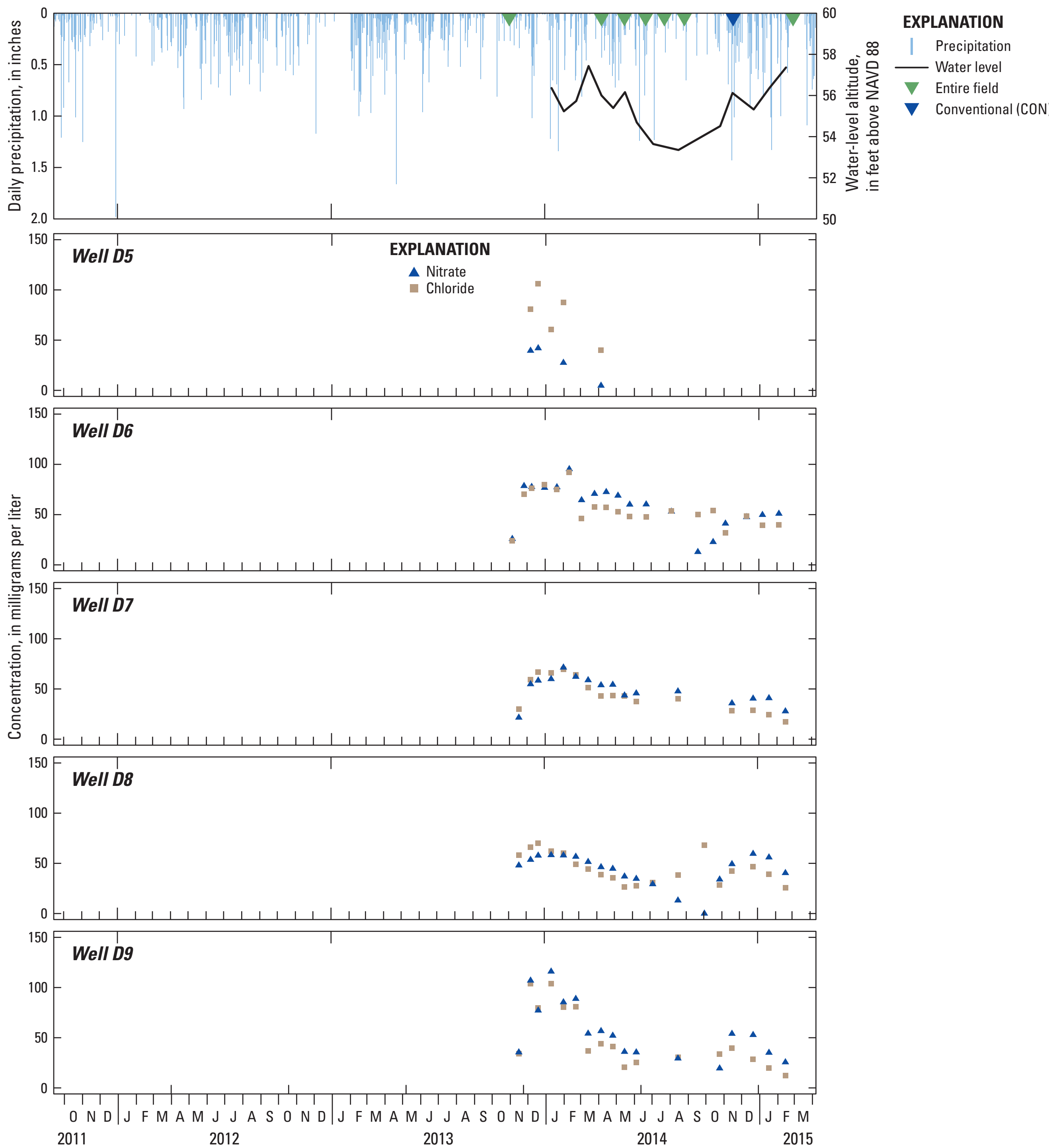


Figure 16. Time series plots of (top) daily precipitation (Ten Mile weather station), manure application dates, and water level (well D2), and nitrate and chloride concentrations in five monitoring wells at field site D, Whatcom County, Washington.

Nitrate concentrations in summer at site B were typically in the range of 15–25 mg-N/L and increased in winter to about 30–40 mg-N/L. site B, which had the longest record of measurements (41 months), showed distinct increases in nitrate and chloride concentrations occurring the autumn recharge period for 2011, 2013, and 2014. However, in 2012, the characteristic concurrent sharp increase in nitrate and chloride concentrations at site B was not observed; instead, the concentration of nitrate continued to decrease in autumn. This decrease probably resulted from the field being tilled and reseeded in summer 2012, and manure was only applied once at reduced rates in autumn that year with a light application of granular fertilizer used in the summer. Small increases in chloride concentration during this time indicated the arrival of recharge. The concentrations of nitrate and chloride in Zone 1 were typically larger than concentrations at the water table during the period of declining water levels. At site C, measurements were obtained over 29 months from November 2012 to March 2015, with characteristic seasonal increases in nitrate and chloride concentrations in 2012, 2013, and 2014, often occurring later during the winter months as transport through a greater thickness of the unsaturated zone likely requires more time. Site D measurements were collected over 17 months from October 2013 to February 2015, with characteristic increases in nitrate and chloride concentrations in autumn 2013 and 2014.

Variability in concentrations of chloride and nitrate among wells increased substantially during recharge periods, indicating that solute transport in recharge is spatially variable. Variability in soil concentrations of nitrate, and in soil properties that affect transport and leaching of soil water, is known to be very large and is expected (Biggar and Neilsen, 1976; Rible and others, 1976). Additionally, even though the soil type was the same throughout the fields, slight variability in land surface level, pockets of soil compaction, ponded areas, and other physical characteristics can cause variability in the soil surface characteristics and thus the underlying soil texture and nutrient transport mechanisms. Transport of solutes through the soil and the unsaturated-zone is both diffuse and preferential, where flow occurs through such features as wormholes, fractures, fingers of enhanced wetness, and contact regions between dissimilar parts of the medium (Nimmo, 2012).

The increase in nitrate and chloride concentrations measured during the recharge period represents a flux of nitrogen in recharge mixing with the existing nitrate in groundwater. The multiple measurements made during the recharge period indicate that the inflow of increased concentrations occurs through much of the recharge period. The concentrations of nitrate and chloride measured in the water table zone beneath the packer (fig. 11) indicate that, during the recharge period, increased concentrations in groundwater from the water table are somewhat restricted to the area near the water table. Concentrations from Zone 1 are typically less variable and more closely represent average conditions.

The large increase in nitrate and chloride concentrations measured at water table coincident with rising groundwater levels (figs. 13–15) shows the input of solutes from the soil zone with groundwater recharge. Agricultural soils at the field sites typically receive seasonal inputs of precipitation in the autumn that far exceed the water-holding capacity of the soil-column, providing several pore-volume of water for leaching soluble constituents from the soil and transporting it to shallow groundwater. Leaching of soil nitrates in autumn following the summer growing season has been reported in studies on both fallow fields (Chichester, 1977; Kowalenko, 1987) and fields with perennial grasses (Smith and others, 2002). While nitrate leaching is expected to be much greater under fallow conditions, Smith and other (2002) measured significant loss of nitrate from perennial grass field receiving manure slurries applied in September and October. In those studies, and in the study reported here, nitrate measured in groundwater collected at the water table beneath forage fields resulted largely from leaching of unassimilated nitrate from the overlying soil zone (assuming limited lateral groundwater flow into the study sites). Soil nitrogen budget studies utilizing labeled ^{15}N nitrogen soil amendments indicate that the source of nitrogen in leachate is a mixture of mineralized soil organic matter and nitrogen from soil amendments (Dowdell and Webster, 1980; Kowalenko, 1989; Meisinger and others, 2008). Kowalenko's isotope studies (1987 and 1989) done in similar soils in the Fraser River valley north of this study area showed low potential for nitrate leaching during the summer months, incorporation of about 17 percent of applied nitrogen in soil organic matter, and leaching of all residual soil nitrate over the winter period. The amount of nitrogen released from soil organic matter and available to crops in any year varies with temperature and moisture and thus is very difficult to predict accurately.

Statistical Comparison of Water-Quality Data from Treatment and Conventional Plots

The results of the Mann Whitney tests (table 7) indicate that concentrations of nitrate, total-nitrogen, and chloride were statistically different ($p<0.05$) between ARM and CON treatment wells at sites B, C, and D, yet the direction and seasonality of these differences varied by site (table 7). At site B, nitrate, total nitrogen, and chloride concentrations in shallow groundwater were significantly higher in the CON treatment plot than in the adjacent ARM treatment plot when utilizing all of the data or the seasonal data after the treatment switch. At site C, nitrate, total nitrogen, and chloride concentrations in shallow groundwater were significantly higher in the ARM treatment plot than in the CON treatment plot, utilizing all data or just the seasonal data. At site D, nitrate and total nitrogen concentrations were significantly higher in the ARM treatment plot, and chloride concentrations were significantly higher in the CON treatment plot when utilizing all data; seasonally, only chloride concentrations were significantly different, which were also higher in the CON treatment plot. Limited treatments at site D constrain the ability to measure application driven differences in concentrations in groundwater.

At site B, there was significant variability in constituent concentrations among all wells (Kruskal-Wallis, $p<0.05$; Conover-Iman post-hoc, $\alpha=0.05$; fig. 17). The significantly lower chemical concentrations in the ARM treatment presented in table 7, and discussed above appear to be heavily influenced by wells B1 and B5. ARM well B1 is significantly lower than three of the four CON wells for nitrate, total nitrogen, and chloride, but not significantly different than any of the other ARM wells (fig. 17*A*–17*C*). Concentrations of nitrate, total nitrogen, and chloride from well B5 (CON treatment plot) are significantly higher than any of the ARM wells at site B. Results were similar for concentrations measured only during the recharge period (October 1–March 31) (fig. 18*D*–18*F*).

At site C, there was significant variability in constituent concentrations among all wells using all data (Kruskal-Wallis, $p<0.05$). Pairwise comparisons (Conover-Iman post-hoc, $\alpha=0.05$) using all data showed that for nitrate and total nitrogen, all CON treatment wells had significantly lower concentrations than ARM wells (fig. 18*A*–18*C*) and were not significantly different from one another within treatment type. Concentrations of chloride in well C1 were significantly lower than all other wells, regardless of treatment. The remaining wells were not significantly different from one another (fig. 18*A*–18*C*).

At site C, the Kruskal-Wallis test using seasonal data found significant differences for nitrate and chloride among individual wells ($p<0.05$), and no difference in total nitrogen among any of the wells (fig. 18*D*–18*F*). Pairwise comparison (Conover-Iman post-hoc, $\alpha=0.05$) results indicate that nitrate concentrations in both CON treatment wells were significantly lower than either of the ARM treatment wells and were not significantly different from one another for annual and seasonal analysis (fig. 18*A* and 18*D*). Chloride concentrations in well C1 were significantly lower than either of the ARM treatment wells (fig. 18*F*); well C2 was not significantly different than either of the two ARM treatment wells.

At site D, there was significant variability in constituent concentrations among all wells using all data and using only the seasonal data (Kruskal-Wallis, $p<0.05$), suggesting high spatial variability. Results from the pairwise comparisons (Conover-Iman post-hoc, $\alpha=0.05$; fig. 19) indicated that nitrate and total nitrogen concentrations were significantly lower in wells D1 and D5 than in the remaining wells for both the entire study period and the seasonal recharge period. Chloride concentrations in wells D4 and D5 were significantly higher than all other wells for both periods (fig. 19).

Nitrate Loading to Groundwater

Estimates were made of annual and monthly loading rates of nitrate transported to groundwater, on a per-acre basis, at sites B and C, by combining the average annual and monthly concentrations of nitrate in recent recharge with estimates of annual and monthly recharge volumes. Estimates of annual groundwater recharge for areas that include sites B and C range from 26 to 30 in/yr (Cox and Kahle, 1999) and were based on regional scale spatial variability in annual precipitation, surficial geology, and regional groundwater flow modeling of Vacarro and others (1998). The high and low recharge estimates were used to bracket estimates of annual loading of nitrogen to groundwater at sites B and C. At sites B and C, the average nitrate concentration at the water table ranged from 9.1 to 31.9 mg-N/L, and estimated annual nitrate loading ranged from 53 to 216 lb-N/acre (pounds of nitrogen per acre) (table 8). The loading estimates are sensitive to both the amount of recharge and concentration of nitrate measured at the water table. Concentration differences were proportionally larger, ranging from 9.1 to 31.9 mg-N/L and having a greater effect on loading than estimated recharge, which ranges from 26 to 30 in/yr. The resulting difference of estimated loading was about 9 lb/acre for each 10.0 mg-N/L measured in groundwater at the water table. Given annual recharge ranging from 26 to 30 in/yr, for concentrations of nitrate in groundwater to be less than 10 mg-N/L, annual loading of nitrogen in leachate from the soil zone would need to be reduced to less than 59–68 pounds per acre-year (lb/acre-yr).

Table 7. Summary of Mann-Whitney test statistics and p-values for comparisons of chemical concentrations in wells from ARM and CON treatment plots by site and datasets used, Whatcom County, Washington.

[Locations of field sites are shown in figure 1. Overall comparisons were for all data. Seasonal data were for data from October 1 to March 31. Chi-Square approximations are presented followed by p-values in parentheses. Values in **bold** indicate significance (p<0.05); letter indicates which treatment was higher (A, application risk management; C, conventional)]

Condition	Mann-Whitney		
	Nitrate	Total nitrogen	Chloride
Site B overall	**12.139 (<0.001)C**	**11.340 (0.001)C**	**12.613 (<0.001)C**
Site B seasonal	**11.249 (0.001)C**	**12.194 (<0.001)C**	**16.655 (<0.001)C**
Site C overall	**17.716 (<0.001)A**	**13.428 (<0.001)A**	**4.870 (0.027)A**
Site C seasonal	**10.359 (0.001)A**	**5.969 (0.015)A**	**4.604 (0.032)A**
Site D overall	**8.214 (0.004)A**	**6.275 (0.012)A**	**19.9 (<0.001)C**
Site D seasonal	0.535 (0.464)	0.293 (0.589)	**12.107 (0.001)C**

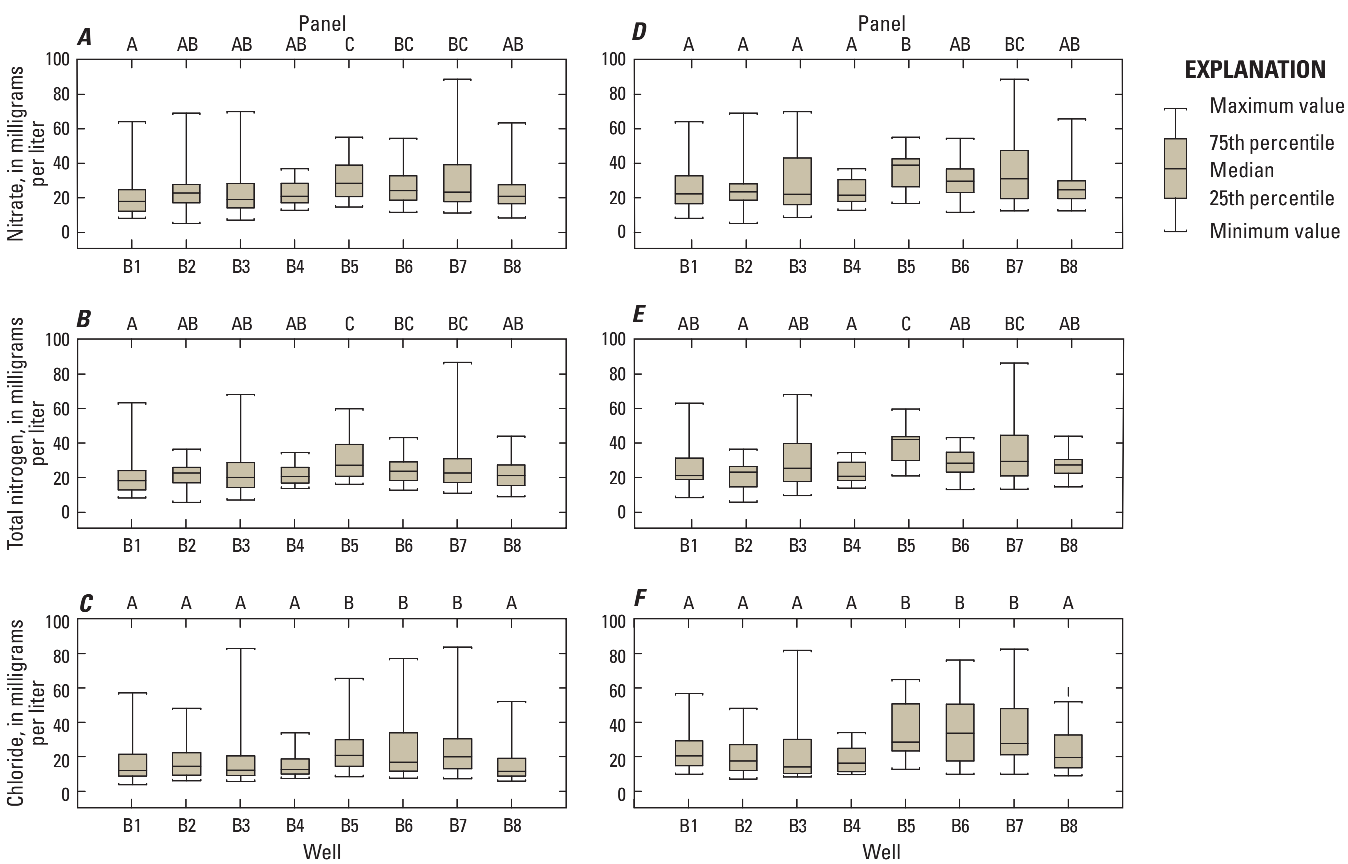


Figure 17. Nitrate, total nitrogen, and chloride concentrations at the water table of site B for all data (*A–C*) and from October to March only (*D–F*). Boxes with dissimilar letters are significantly different (Conover-Iman, p<0.05). Wells B1–B4 received application risk management (ARM) treatment and wells B5–B8 received conventional management (CON) treatment.

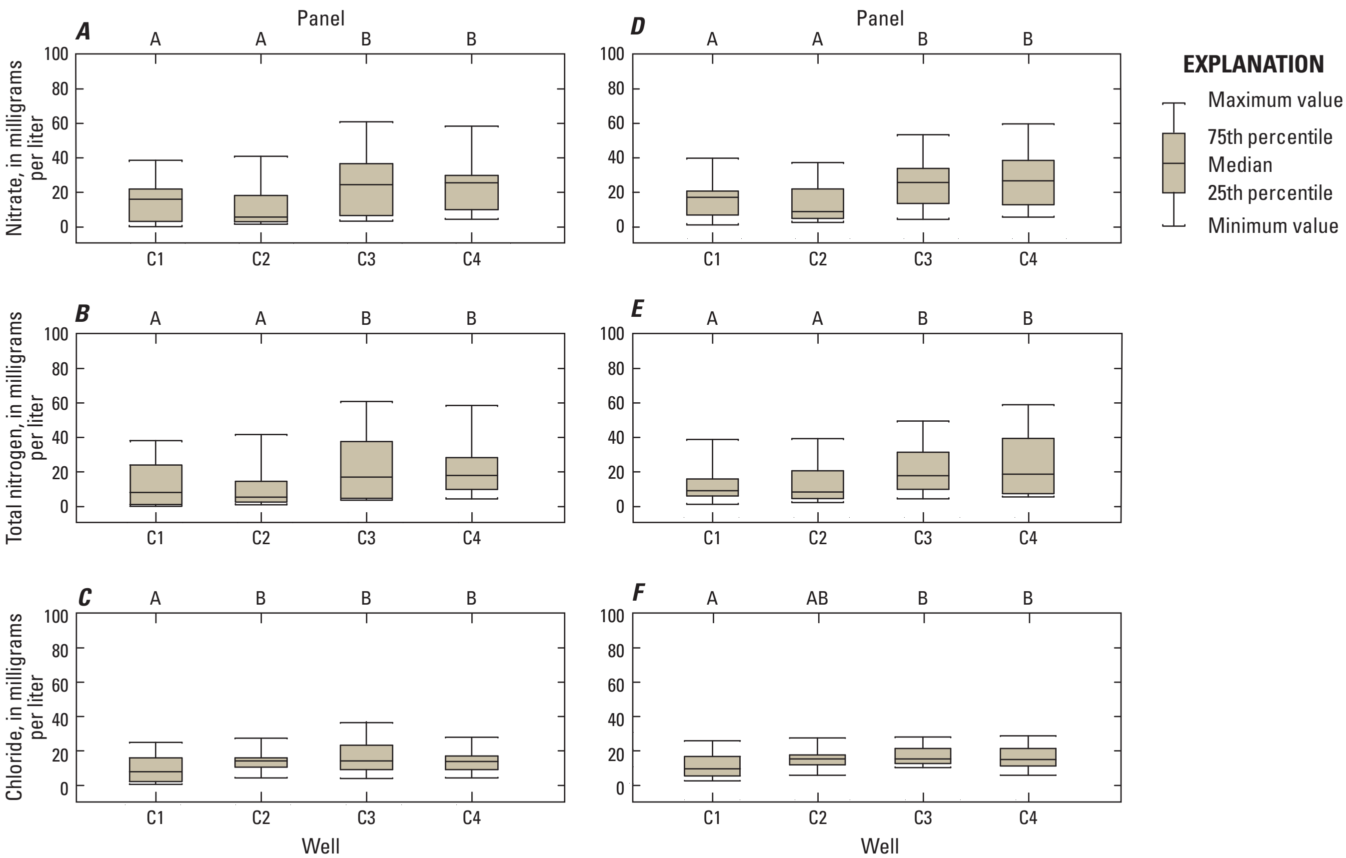


Figure 18. Nitrogen, and chloride concentrations at the water table of site C for all data (*A*–*C*) and only measurements from October to March (*D*–*F*). Boxes with dissimilar letters are significantly different (Conover-Iman, $p<0.05$). Wells C1–C2 received conventional management (CON) treatment and wells B3–B4 received application risk management (ARM) treatment.

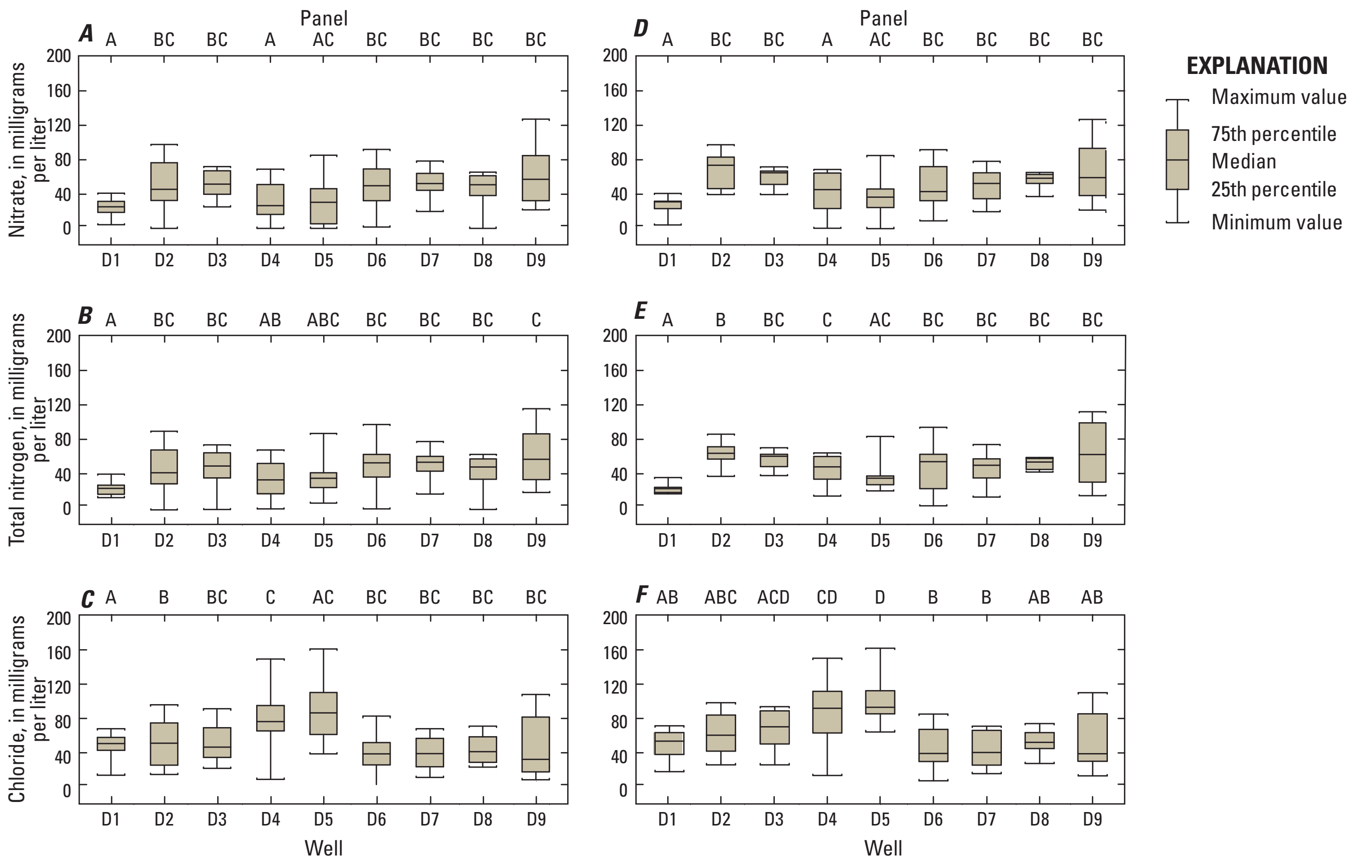


Figure 19. Nitrate, total nitrogen, and chloride concentrations at the water table of site D for all data (*A–C*) and from October to March only (*D–F*), Whatcom County, Washington. Boxes with dissimilar letters are significantly different (Conover-Iman, p<0.05).

Table 8. Annual estimates of nitrogen loading based on an average annual recharge rate of 26–30 inches per year as determined by Cox and Kahle (1999) for the Sumas-Blaine Aquifer and mean water table nitrate concentrations for the defined recharge period, Whatcom County, Washington.

[Loading estimates are shown to be highly sensitive to both recharge amount and water-table concentrations. **Abbreviations:** mg–N/L, milligram nitrogen per liter; ARM, application risk management; CON, conventional]

Recharge period July 1–June 30	Average nitrate (mg–N/L)	Nitrogen loading based on average annual recharge rate (pound per acre per year)		Average nitrate (mg–N/L)	Nitrogen loading based on average annual recharge rate (pound per acre per year)	
		26 inches	30 inches		26 inches	30 inches
	Wells B1–B4 (ARM)			Wells B5–B8 (CON)		
2012–13	18.2	107	123	24.8	146	168
2013–14	26.1	153	177	28.6	168	194
2014–15	24.5	144	166	29.3	172	199
	Wells C1–C2 (CON)			Wells C3–C4 (ARM)		
2012–13	9.1	53	62	15.1	89	102
2013–14	14.1	83	96	26.3	155	178
2014–15	19.5	115	132	31.9	187	216

Seasonal variations in N-loading to groundwater computed from monthly average nitrate concentrations and monthly recharge estimates based on the soil water balance for sites B and C are shown in figure 20. The average summation of monthly groundwater recharge estimates for the July 1–June 30 recharge periods, determined from soil water balance data, were 23.0 in/yr at site B and 27.2 in/yr at site C. These were about 90 percent of the range of annual recharge reported by Cox and Kahle (1999) that were used for annual estimates of loading. The largest part of leaching arrives at the water table during the seasonal period of heavy rainfall typically around December (fig. 20). Nitrate leached to groundwater based on the sum of monthly loading estimates ranged from 86.0 to 196.4 lb/acre for the calculated periods (table 9). The range of nitrate loading to groundwater estimated in this study was similar to the annual 155 lb/acre estimated by Kuipers and others (2014) beneath a raspberry field near the Abbotsford International Airport in Sumas, British Columbia. Similar seasonal patterns of nitrate leaching have been reported for grassland farm plots on freely draining soils receiving monthly applications of animal manures (Smith and others, 2002).

Data on manure application rates were not available. However, annual manure application guidance for high-intensity forage production (5–8 cuttings) at these fields was recommended at 348–448 lb-N/acre using University Extension guidance (Downing and others, 2007). This nitrogen amendment rate is based on the mass of nitrogen expected to be removed by the crop. An additional approximately 20 percent nitrogen is recommended to account for uncontrollable estimated losses due to other processes such as ammonia volatilization during application, nitrogen in the sub-surface root system, and nitrogen immobilized in soil organic matter, bring the total application recommendation from 417 to 537 lb-N/acre depending on expected yield. The estimate of leaching losses to groundwater derived from monthly recharge were 16–37 percent of the recommended soil nitrogen amendment in the manure application guidance which is somewhat larger than leaching losses commonly reported in soil nitrogen budgets in which leaching losses typically range from 10 to 30 percent of total nitrogen inputs (Meisinger and others, 2008).

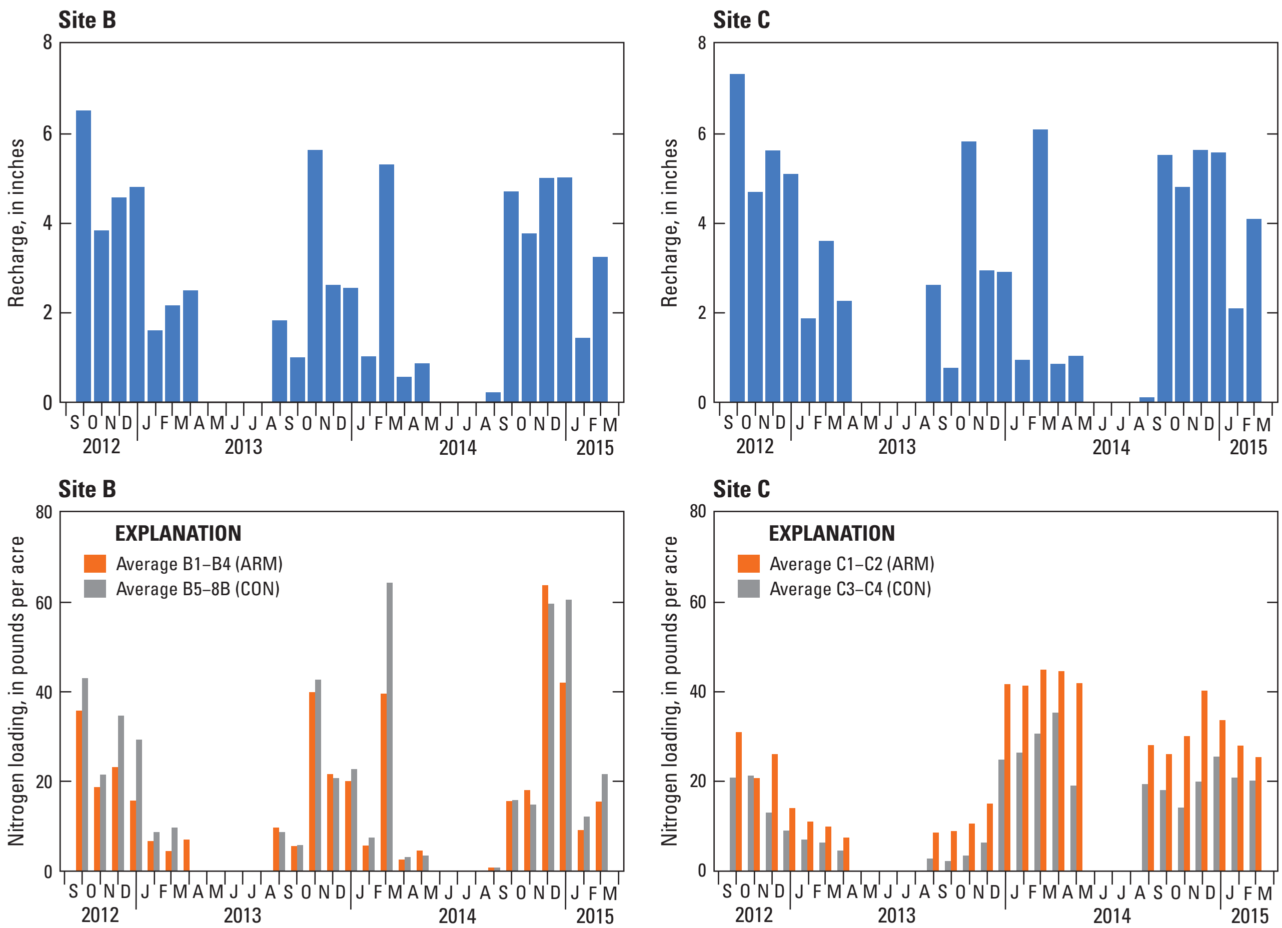


Figure 20. Estimated monthly groundwater recharge and nitrogen loading to groundwater at sites B and C, Whatcom County, Washington. Locations of field sites shown in figure 1. Treatments—ARM, application risk management; CON, conventional management.

Table 9. Annual nitrogen loading rates for field sites B and C calculated from monthly mean nitrogen concentration and monthly recharge estimates summed over the recharge period, Whatcom County, Washington.

[Recharge periods are from July 1 through June 30, except where noted. **Abbreviations:** ARM, application resource management; CON, conventional]

Recharge period	Recharge from water balance (inch per year)	Sum of monthly nitrogen loading (pound per acre per year)	
		Wells B1–B4 (ARM)	Wells B5–B8 (CON)
2012–13	26.0	146.7	111.1
2013–14	21.4	178.6	148.5
2014–15[1]	23.4	185.1	164.5
		Wells C1–C2 (CON)	Wells C3–C4 (ARM)
2012–13[2]	30.5	94.3	139.1
2013–14	24.0	86.0	147.0
2014–15[3]	27.2	124.3	196.4

[1]Calculated for July 2014–March 2015.

[2]Calculated for October 2012–June 2013.

[3]Calculated for July 2014–March 2015.

Nitrogen in soils used by plants or leached to groundwater can originate from sources other than applied manure including mineralization of soil organic matter, wet and dry atmospheric deposition, and from irrigation water. Mineralization of soil organic matter, which was not measured as part of this study, can contribute a large amount of the nitrogen available to plants (Meisinger and others, 2008). The total organic carbon content of agricultural soils in the SBA range from 3 to 9 percent (Goldin, 1992) and the annual 1–2 percent mineralization rates of the soil organic matter could easily generate upwards of 150 lb-N/acre-yr (Kowalenko, 1989). Similar soils, in nearby areas of the Fraser Valley, were shown to have post-harvest mineralization in the autumn that alone generated up to 30 lb-N/acre (Kowalenko, 1987). In the autumn post-harvest period uptake by grasses is limited. Immobilization of nitrate into soil organic matter can be equally large and soil nitrogen budgets often assume that these processes are in equilibrium (Meisinger, 2008). Although this assumption may be useful for long studies, in this study, year-to-year variability in soil nitrogen processes due to weather, slow reaction rates, and the large mass of soil nitrogen (4,700–8,500 lb/acre) make such an assumption untenable. With the onset of heavy rainfall in the autumn, mineralization of nitrate may continue to occur in warm soil temperatures that extend through much of the autumn. Yet, while nutrient uptake by perennial grasses will continue in the autumn, nitrogen uptake will diminish because of the reduced sunlight and cooler air temperatures in November–December. Thus, the effects of mineralization on the rate of nitrate leached to groundwater can be difficult to assess accurately and distinguishing the distribution of nitrate leached to groundwater from mineralized soil organic matter and excess application of manure was not possible.

Wet and dry atmospheric deposition and irrigation water both are additional sources of nitrogen to agricultural soils. Irrigation practices for forage crops in Whatcom County typically apply from 6 to 12 in. of groundwater irrigation per year. For groundwater containing 10 mg-N/L, annual loading from irrigation would be from 14 to 28 lb-N/ acre-year. Atmospheric deposition is estimated to be on the order of 15 lb-N/acre-year (Barry and others, 1993; Zebarth and others, 1999).

During October 2014, average autumn soil nitrate concentrations in 12-in. samples from the study sites provided by the WCD companion study were about 11, 14, and 19 parts per million (ppm) at sites B, C, and D, respectively. The soil nitrate concentrations were generally similar (within 20 percent) between the ARM and CON plots, but the ARM plots tended to have the consistently lower values of the two. Anticipated leachate to groundwater from these measurements would be less than was observed from measurements of nitrate at the water table. Nitrogen removed from the test plots by crop harvest exceeded the anticipated crop yields of 448 lb-N/acre. Taken together with leaching losses of about 150 lb-N/acre, the total mass of N removed from the soil system was on the order of 600 lb-N/acre, which exceeded the targeted nitrogen amendment rate. The contribution of nitrogen to soils from mineralization of soil organic matter, irrigation, and atmospheric deposition need to be accounted for in scheduling manure application as they may have contributed to nitrate leaching in the autumn.

Summary

Three years of groundwater-quality data were collected from shallow wells installed in forage fields receiving seasonal applications of dairy manure as part of a field evaluation of a manure application system aimed at scheduling of manure applications based on site-specific conditions rather than calendar dates. The water-quality samples collected at the water table showed that recently recharged groundwater contained variable, but consistently high, concentrations of nitrate exceeding the EPA MCL of 10 mg-N/L in more than 85 percent of samples. Leaching of nitrogen and chloride were greatest during autumn following the onset of heavy seasonal precipitation indicating that residual nitrate present in the soil column at the end of the growing season can be transported to the groundwater system during the autumn-winter recharge period. Although autumn nitrate concentrations in soil measured after the last cutting of forage grass were in the acceptable range for sites B and C (less than 15 ppm) and slightly greater than for site D (less than 20 ppm), excess nitrate accumulation in the soil column from manure application and (or) soil mineralization still resulted in flushing of nitrate to groundwater. Less easily transported water-quality constituents associated with manure applications, including ammonia, phosphorus, and *E. coli*, were rarely observed in samples from the water table indicating that little if any of these constituents are being transported to groundwater with seasonal groundwater recharge. Ammonia and phosphorus are both positively charged ions that can readily be sorbed to soil particles. However, in the predominantly aerobic unsaturated zone present at these sites, ammonia would likely be oxidized to nitrate, which is readily soluble and available for either plant uptake or transport to groundwater. Application of nitrogen in the latter part of the growing season should be more carefully evaluated on the soil types tested to reduce the potential for residual nitrate to be present in soils at the end of the growing season. Difficulties in estimating application and mineralization rates are well documented, and expectations of 100 percent efficiency in nutrient use on manure applications are unrealistic. Although some loss of nitrogen through denitrification and leaching are to be expected, fertilization rates that are designed to maximize crop yield by over applying nitrogen may not be desirable due to environmental and economic losses (Rice and others, 1995).

Statistical comparison of nitrate and chloride data from the paired test plots was inconclusive regarding the effectiveness of manure application scheduling via the ARM system alone on reducing impacts of nitrate leaching to groundwater under the conditions monitored. Reductions in groundwater nitrates from ARM manure application strategies were inconclusive; concentrations were significantly lower in site B, yet significantly higher in site C when comparing concentrations in groundwater at paired treatment plots. However, the number of wells at site B was also up to 50 percent more than at site C, which may have limited data collection at site C to reduce the in-field variability effect on well data. Nitrates in groundwater significantly varied among individual wells at each site, suggesting leaching of nitrates from soil after manure application is spatially variable at the field scale tested regardless of manure application strategy. The amount of nitrate present in recently recharged groundwater was variable, and the observed differences could not be statistically differentiated. Improvements in nitrogen crediting are needed as estimates of nitrate loading to groundwater during recharge periods ranged from 86.0 to 196.4 lb-N/acre per year, which was 16–37 percent of the recommended nitrogen soil amendment rate.

Acknowledgments

The authors would like to acknowledge and thank the contributions from many individuals who made this study possible. We would like to acknowledge the dairy producers who made accommodations to provide access to install sampling facilities in their forage fields and provided continued access for the duration of the study. Assistance was also provided in sample collection from field staff of the Whatcom Conservation District and U.S. Geological Survey, Ferndale, Washington, field office. We also thank colleague and peer reviewers who provided valuable suggestions that improved this manuscript.

References Cited

Allen, R.G., Walter, I.A., Elliot, R., Howell, T., Itenfisu, D., and Jensen, M., eds., 2005, The ASCE standardized reference evapotranspiration equation: Task Committee on Standardization of Reference Evapotranspiration of the Environmental and Water Resources Institute of the American Society of Civil Engineers, 59 p.

Barry, D.A.J., Goorahoo, D., and Goss, M.J., 1993, Estimation of nitrate concentrations in ground water using whole farm nitrogen budgets: Journal of Environmental Quality, v. 22, no. 4, p. 767–775.

Beckwith, C.P., Cooper, J., Smith, K.A., and Shepherd, M.A., 1998, Nitrate leaching loss following application of organic manures to sandy soils in arable cropping—Effects of application time, manure type, overwinter crop cover and nitrification inhibition: Soil Use and Management, v. 14, no. 3, p. 123–130.

Beven, K., and Germann, P., 1982, Macropores and water flow in soils: Water Resources Research, v. 18, no. 5, p. 1311–1325.

Biggar, J.W., and Nielsen, D.R., 1976, Spatial variability of the leaching characteristic of a field soil: Water Resources Research, v. 12, no. 1, p. 78–84.

Böhlke, J.K., 2002, Groundwater recharge and agricultural contamination: Hydrogeology Journal, v. 10, no. 1, p. 153–179.

Bruckler, L., de Cockborne, A.M., Renault, R., and Claudot, B., 1997, Spatial and temporal variability of nitrate in irrigated salad crops: Irrigation Science, v. 17, no. 2, p. 53–61.

Burkart, M.R., and Stoner, J.D., 2001, Nitrogen in groundwater associated with agricultural systems, chap. 6 of Follett, R.F., and Hatfield, J.L., eds., Nitrogen in the environment—Sources, problems and management: Amsterdam, Netherlands, Elsevier Science, p. 123–145.

Burow, K.R., Nolan, B.T., Rupert, M.G., and Dubrovsky, N.M., 2010, Nitrate in groundwater of the United States, 1991–2003: Environmental Science & Technology, v. 44, no. 13, p. 4988–4997.

Carey, B.M., 2017, Sumas-Blaine aquifer long-term groundwater quality monitoring, 2009–2016: Washington State Department of Ecology, Publication No. 17-03-013.

Carey, B.M., and Cummings, R., 2013, Sumas-Blaine aquifer nitrate contamination summary: Washington State Department of Ecology, Publication No. 12-03-026 (revised February 2013).

Carey, B.M., and Harrison, J.H., 2014, Nitrogen dynamics at a manured grass field overlying the Sumas-Blaine aquifer in Whatcom County: Washington State Department of Ecology, Publication No. 14-03-001.

Carey, B.M., Pitz, C.F., and Harrison, J.H., 2017, Field nitrogen budgets and post-harvest soil nitrate as indicators of N leaching to groundwater in a Pacific Northwest dairy grass field: Nutrient Cycling in Agroecosystems, v. 107, no. 1, p. 107–123, https://doi.org/10.1007/s10705-016-9819-5.

Chang, C., and Entz, T., 1996, Nitrate leaching losses under repeated cattle feedlot manure applications in Southern Alberta: Journal of Environmental Quality, v. 25, no. 1, p. 145–153, https://doi.org/10.2134/jeq1996.00472425002500010019x.

Cherney, D.J.R., Cherney, J.H., and Mikhailova, E.A., 2002, Orchardgrass and tall fescue utilization of nitrogen from dairy manure and commercial fertilizer: Agronomy Journal, v. 94, no. 3, p. 405–411, https://doi.org/10.2134/agronj2002.4050.

Chesnaux, R., and Allen, D.M., 2008, Simulating nitrate leaching profiles in a highly permeable vadose zone: Environmental Modeling and Assessment, v. 13, no. 4, p. 527–539, https://doi.org/10.1007/s10666-007-9116-4.

Chichester, F.W., 1977, Effects of increased fertilizer rates on nitrogen content of runoff and percolate from monolith lysimeters: Journal of Environmental Quality, v. 6, no. 2, p. 211–217, https://doi.org/10.2134/jeq1977.00472425000600020023x.

Clark, C.M., Bell, M.D., Boyd, J.W., Compton, J.E., Davidson, E.A., Davis, C., Fenn, M.E., Geiser, L., Jones, L., and Blett, T.F., 2017, Nitrogen-induced terrestrial eutrophication—Cascading effects and impacts on ecosystem services: Ecosphere, v. 8, no. 7, p. 1–28, https://doi.org/10.1002/ecs2.1877.

Close, M., 2010, Critical review of contaminant transport time through the vadose zone: Christchurch, New Zealand, Institute of Environmental Science and Research, Report prepared for Environment Canterbury, 46 p.

Compton, J.E., Harrison, J.A., Dennis, R.L., Greaver, T.L., Hill, B.H., Jordan, S.J., Walker, H., and Campbell, H.V., 2011, Ecosystem services altered by human changes in the nitrogen cycle—A new perspective for US decision making: Ecology Letters, v. 14, no. 8, p. 804–815, https://doi.org/10.1111/j.1461-0248.2011.01631.x.

Cox, S.E., and Kahle, S.C., 1999, Hydrology, ground-water quality, and sources of nitrate in lowland glacial aquifers of Whatcom County, Washington, and British Columbia, Canada: U.S. Geological Survey Water-Resources Investigations Report 98-4195, https://pubs.er.usgs.gov/publication/wri984195.

Cox, S.E., Huffman, R.L., Olsen, T.D., and Spanjer, A.R., 2016, Concentration of nitrate and other water-quality constituents in groundwater from the water table beneath forage fields receiving seasonal applications of dairy manure, Whatcom County, Washington (2015): U.S. Geological Survey data release, https://dx.doi.org/10.5066/F7D50K3F.

Cox, S.E., Simonds, F.W., Doremus, L.A., Huffman, R.L., and Defawe, R.M., 2005, Ground water/surface water interactions and quality discharging groundwater in streams of the Lower Nooksack River Basin, Whatcom, Washington: U.S. Geological Survey Scientific Investigations Report 2005-5255, p. 45.

Dasika, R.K., 1996, Investigation into the distribution of non-point source nitrate in two unconfined aquifers: University of British Columbia, Vancouver, Canada, PhD dissertation, 365 p.

Dowdell, R.J., and Webster, C.P., 1980, A lysimeter study using nitrogen-15 on the uptake of fertilizer nitrogen by perennial ryegrass swards and losses by leaching: Journal of Soil Science, v. 31, no. 1, p. 65–75. https://doi.org/10.1111/j.1365-2389.1980.tb02065.x

Downing, T., Sullivan, D., Hart, J., and Gamroth, M., 2007, Manure application rates for forage production: Corvallis, Oregon State University Extension Service, EM 8585-E, Oregon, 4 p.

Drost, B.W., 2005, Quality-assurance plan for ground-water activities: U.S Geological Survey Open-File Report 2005-1126, 28 p., http://pubs.usgs.gov/of/2005/1126/pdf/ofr20051126.

Embertson, N., 2016, Protecting Puget Sound watersheds from agricultural pollution using a progressive manure application risk management (ARM) system: Project report prepared by Whatcom Conservation District for the U.S. Environmental Protection Agency, 71 p., accessed August 2018, at https://www.eopugetsound.org/sites/default/files/features/resources/PO-00J08101%20Whatcom%20ARM%20final%20report.pdf

Erickson, D., and Matthews, W., 2002, Effects of land application of dairy manure and wastewater on groundwater quality: Washington State Department of Ecology, https://fortress.wa.gov/ecy/publications/summarypages/0203002.html

Galloway, J.N., Leach, A.M., Bleeker, A., and Erisman, J.W., 2013, A chronology of human understanding of the nitrogen cycle: Philosophical Transactions of the Royal Society of London. Series B, Biological Sciences, v. 368, no. 1621, p. 20130120, https://doi.org/10.1098/rstb.2013.0120.

Gerke, H.H., Germann, P., and Nieber, J., 2010, Preferential and unstable flow—From the pore to the catchment scale: Vadose Zone Journal, v. 9, no. 2, p. 207–212, https://doi.org/10.2136/vzj2010.0059.

Gleeson, T., Alley, W.M., Allen, D.M., Sophocleous, M.A., Zhou, Y., Taniguchi, M., and VanderSteen, J., 2012, Towards sustainable groundwater use—Setting long-term goals, backcasting, and managing adaptively: Groundwater Journal, v. 50, no. 1, p. 19–26, https://doi.org/10.1111/j.1745-6584.2011.00825.x.

Goldin, A., 1992, Soil survey of Whatcom County Area. U.S. Department of Agriculture, Soil Conservation Service, p. 481.

Graham, G., Allen, D.M., and Finkbeiner, B., 2015, Climate controls on nitrate concentration variability in the Abottsford-Sumas aquifer, British Columbia, Canada: Environmental Earth Sciences, v. 73, no. 6, p. 2895–2904, https://doi.org/10.1007/s12665-014-3072-5.

Hahn, G.J., and Meeker, W.Q., 1991, Statistical intervals—A guide for practitioners: New York, John Wiley and Sons, 392 p., https://doi.org/10.1002/9780470316771.

Hepperly, P., Lotter, D., Ulsh, C.Z., Seidel, R., and Reider, C., 2009, Compost, manure, and synthetic fertilizer influences crop yields, soil properties, nitrate leaching and crop nutrient content: Compost Science & Utilization, v. 17, no. 2, p. 117–126.

Hermanson, R.E., Pan, W.L., Perilla, C.A., Stevens, R.G., and Stockle, C.O., 2000, Nitrogen use by crops and the fate of nitrogen in the soil and vadose zone—A literature search: Washington State Department of Ecology Publication No. 00-10-015, variously paginated.

Korom, S.F., 1992, Natural denitrification in the Saturated zone—A review: Water Resources Research, v. 28, no. 6, p. 1657–1668, https://doi.org/10.1029/92WR00252.

Kowalenko, C.G., 1987, The dynamics of inorganic nitrogen in a Fraser Valley soil with and without spring or fall ammonium nitrate application: Canadian Journal of Soil Science, v. 67, no. 2, p. 287–301, https://doi.org/10.4141/cjss87-032.

Kowalenko, C.G., 1989, The fate of applied nitrogen in a Fraser Valley soil using 15N in field microplots: Canadian Journal of Soil Science, v. 69, no. 4, p. 825–833, https://doi.org/10.4141/cjss89-083.

Kuipers, P.J., Ryan, M.C., and Zebarth, B.J., 2014, Estimating nitrate loading from and intensively managed agricultural field to a shallow unconfined aquifer: Water Quality Research Journal of Canada, v. 49, no. 1, p. 10–22, https://doi.org/10.2166/wqrjc.2013.136.

Meisinger, J.J., Calderon, F.J., and Jenkinson, D.S., 2008, Soil nitrogen budgets, *in* Schepers, J.S., and Raun, W.R., eds., Nitrogen in agricultural systems: Madison, Wisconsin, American Society of Agronomy, Crop Science Society of America, Soil Science Society of America, Nitrogen in Agricultural Systems, Agronomy Monograph 49, chap. 13, p. 505–562.

Mueller, D.K., Schertz, T.L., Martin, J.D., and Sandstrom, M.W., 2015, Design, analysis, and interpretation of field quality-control data for water-sampling projects: U.S. Geological Survey, Techniques and Methods, book 4, chap. C4, 54 p., https://doi.org/10.3133/tm4C4.

Myers, D.N., Stoeckel, D.M., Bushon, R.N., Francy, D.S., and Brady, A.M.G., 2007, Fecal indicator bacteria: U.S. Geological Survey Techniques of Water-Resources Investigations, book 9, chap. A7, section 7.1 (version 2.0), http://water.usgs.gov/owq/FieldManual/Chapter7/index.html.

Nimmo, J.R., 2005, Unsaturated zone flow process, chap. 150 of Anderson, M.G., ed., Encyclopedia of hydrologic sciences: New York, John Wiley & Sons, Ltd., chap. 150, https://doi.org/10.1002/0470848944.hsa161.

Nimmo, J.R., 2012, Preferential flow occurs in unsaturated conditions: Hydrological Processes, v. 26, no. 5, p. 786–789, https://doi.org/10.1002/hyp.8380.

Osmond, D.L., Hoag, D.L.K., Luloff, A.E., Meals, D.W., and Neas, K., 2015, Farmers' use of nutrient management—Lessons from watershed case studies: Journal of Environmental Quality, v. 44, no. 2, p. 382–390, https://doi.org/10.2134/jeq2014.02.0091.

Paul, J.W., and Zebarth, B.J., 1997, Denitrification and nitrate leaching during the fall and winter following dairy cattle slurry application: Canadian Journal of Soil Science, v. 77, no. 2, p. 231–240, https://doi.org/10.4141/S96-052.

Poore, J.K., 2005, Winter period application of manure in Washington State: U.S. Department of Agriculture, Natural Resources Conservation Service Technical Note 14, 7 p.

Puckett, L.J., Tesoriero, A.J., and Dubrovsky, N.M., 2011, Nitrogen contamination of surficial aquifers—A Growing Legacy: Environmental Science & Technology, v. 45, no. 9, p. 839–844.

Redding, M., 2008, Nitrate trends in the Central Sumas-Blaine Aquifer: Washington State Department of Ecology, 117 p., Publication No. 08-03-018, https://fortress.wa.gov/ecy/publications/SummaryPages/0803018.html.

Rible, J.M., Nash, P.A., Pratt, P.F., and Lund, L.J., 1976, Sampling the unsaturated zone of irrigated lands for reliable estimates of nitrate concentrations: Soil Science Society of America Journal, v. 40, no. 4, p. 566–570, https://doi.org/10.2136/sssaj1976.03615995004000040031x.

Rice, C.W., Havlin, J.L., and Schepers, J.S., 1995, Rational nitrogen fertilization in intensive cropping systems: Fertilizer Research, v. 42, no. 1–3, p. 89–97, https://doi.org/10.1007/BF00750502.

Robertson, G.P., and Vitousek, P.M., 2009, Nitrogen in agriculture—Balancing the cost of an essential resource: Annual Review of Environment and Resources, v. 34, no. 1, p. 97–125, https://doi.org/10.1146/annurev.environ.032108.105046.

Rupert, M.G., 2008, Decadal-scale changes of nitrate in ground water of the United States, 1988–2004: Journal of Environmental Quality, v. 37, 5_Supplement, p. pS240–S248, https://doi.org/10.2134/jeq2007.0055.

Sebilo, M., Mayer, B., Nicolrdot, B., Pinay, G., and Mariotti, A., 2013, Long-term fate of nitrate fertilizer in agricultural soils: Proceeding of the National Academy of Science of the United States of America v. 110, p. 18185-18189, https://doi.org/10.1073/pnas.1305372110.

Singer, J.W., and Moore, K.J., 2003, Nitrogen removal by orchardgrass and smooth bromegrass and residual soil nitrate: Crop Science, v. 43, no. 4, p. 1420–1426, https://doi.org/10.2135/cropsci2003.1420.

Smith, K.A., Beckwith, C.P., Chalmers, A.G., and Jackson, D.R., 2002, Nitrate leaching following autumn and winter application of animal manures to grassland: Soil Use and Management, v. 18, no. 4, p. 428–434, https://doi.org/10.1111/j.1475-2743.2002.tb00262.x.

Sobota, D.J., Compton, J.E., and Harrison, J.A., 2013, Reactive nitrogen inputs to US lands and waterways—How certain are we about sources and fluxes: Frontiers in Ecology and the Environment, v. 11, no. 2, p. 82–90, https://doi.org/10.1890/110216.

Sullivan, D.M., 2008, Estimating plant-available nitrogen from manure: Oregon State University Extension Service, EM 8954-E, 21 p.

Tiedje, J.M., 1988, Ecology of denitrification and dissimilatory nitrate reduction to ammonium, in Zehnder, A.J.B., ed., Biology of anaerobic microorganisms: New York, John Wiley, p. 179–244, https://efotg.sc.egov.usda.gov/toc.aspx?CatID=16204.

U.S. Department of Agriculture, 2003, Washington agricultural statistics bulletin 2003: U.S. Department of Agriculture, 128 p., accessed August 2017, at www.digitalarchives.wa.gov/do/5E50E66193F7775742AAA7FA8DC170B2.pdf.

U.S. Department of Agriculture, 2016, Washington agricultural statistics 2003: U.S. Department of Agriculture, www.nass.usda.gov/Statistics_by_State/Washington/Publications/Annual_Statistical_Bulletin/2017/WA_annual%20bulletin%202017.pdf.

U.S. Department of Agriculture, 2012, Part 651—Agricultural waste management field handbook: U.S. Department of Agriculture, https://directives.sc.egov.usda.gov/viewerFS.aspx?id=3851.

U.S. Department of Agriculture, 2013, Nutrient Management Code 590—Conservation practice standard: U.S. Department of Agriculture, https://efotg.sc.egov.usda.gov/references/public/WA/590_stnd_021814.pdf https://efotg.sc.egov.usda.gov/toc.aspx?CatID=16204.

U.S. Department of Agriculture, 2014a, Nutrient Management Code 590—Winter period application of manure in Washington State: U.S. Department of Agriculture, Natural Resources Conservation Service Technical Note 14, 7 p., https://efotg.sc.egov.usda.gov/references/public/WA/WA_TN14_021214.pdf.

U.S. Department of Agriculture, 2014b, Nutrient Management Code 590 budget worksheet: U.S. Department of Agriculture, accessed November 16, 2017, https://efotg.sc.egov.usda.gov/references/public/WA/590_WAAutoBudget_022014.xlsm.

U.S. Department of Agriculture, 2017, Natural Resources Conservation Service conservation practice standard—Nutrient management—Code 590: U.S. Department of Agriculture, accessed November 16, 2017, at https://www.nrcs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb1046433.pdf.

U.S. Environmental Protection Agency, 2011, Reactive nitrogen in the United States—An analysis of inputs, flows, consequences, and management options: U.S. Environmental Protection Agency, EPA-SAB-11-013, p. 139.

U.S. Geological Survey, variously dated, National field manual for the collection of water-quality data: U.S. Geological Survey Techniques of Water-Resources Investigations, book 9, chaps. A1–A9, accessed September 10, 2012, at http://pubs.er.usgs.gov/publication/twri09.

Vitousek, P., Naylor, R., Crews, T., David, M.B., Drinkwater, L.E., Holland, E., Johnes, P.J., Katzenberger, J., Martinelli, L.A., Matson, P.A., Nziguheba, G., Ojima, D., Palm, C.A., Robertson, G.P., Sanchez, P.A., Townsend, A.R., and Zhang, F.S., 2009, Nutrient imbalances in agricultural development: Science, v. 324, no. 5934, p. 1519–1520, https://doi.org/10.1126/science.1170261.

Vaccaro, J.J., Hanse, A.J., and Jones, M.A., 1998, Hydrogeologic framework of the Puget Sound aquifer system: U.S. Geological Survey Professional Paper, v. 1425D, 77 p.

Van Es, H.M., Sogbedji, J.M., and Schindelbeck, R.R., 2006, Effect of manure application timing, crop, and soil type on nitrate leaching: Journal of Environmental Quality, v. 35, no. 2, p. 670–679, https://doi.org/10.2134/jeq2005.0143.

Vieira, S.R., Nielsen, D.R., and Biggar, J.W., 1981, Spatial variability of field measured infiltration rate: Soil Science Society of America Journal, v. 45, no. 6, p. 1040–1048, https://doi.org/10.2136/sssaj1981.03615995004500060007x.

Viets, F.G., 1965, The plants need for and use of nitrogen, chap. 14 of Bartholomew, W.V., and Clark, F.E., eds., Soil nitrogen, Agronomy Monograph 10, 1965: Madison, Wisconsin, American Society of Agronomy, Crop Science Society of America, Soil Science Society of America, p. 503–549.

Wagner, R.J., Kimbrough, R.A., and Turney, G.L., 2007, Quality-assurance plan for water-quality activities in the U.S. Geological Survey Washington Water Science Center: U.S. Geological Survey Open-File Report 2007-1307, 47 p.

Ward, M.H., deKok, T.M., Levallois, P., Brender, J., Gulis, G., Nolan, B.T., and VanDerslice, J., 2005, Workgroup report—Drinking-water nitrate and health—Recent findings and research needs: Environmental Health Perspectives, v. 113, no. 11, p. 1607–1614, https://doi.org/10.1289/ehp.8043.

Washington State University, 2018, AgWeatherNet current conditions map: Washington State University, http://weather.wsu.edu/.

Western Regional Climate Center, 2018, Cooperative climatological data summaries: Western Regional Climate Center, https://wrcc.dri.edu/Climate/west_coop_summaries.php.

Whatcom Conservation District, 2015, Application risk management (ARM): Whatcom Conservation District, http://www.whatcomcd.org/arm.

Yungen, J.A., Jackson, T.L., and McGuire, W.S., 1977, Seasonal responses to perennial forage grasses to nitrogen applications: Oregon State University Agricultural Experiment Station Bulletin 626, 11 p.

Zebarth, B.J., Ryan, M.C., Graham, G., Forge, T.A., and Neilsen, D., 2015, Groundwater monitoring to support development of BMPs for groundwater protection—The Abbotsford-Sumas Aquifer Case Study: Ground Water Monitoring and Remediation, v. 35, no. 1, p. 82–96.

Publishing support provided by the U.S. Geological Survey
Science Publishing Network, Tacoma Publishing Service Center

For more information concerning the research in this report, contact the
Director, Washington Water Science Center
U.S. Geological Survey
934 Broadway, Suite 300
Tacoma, Washington 98402
https://wa.water.usgs.gov

Cox and others—**Nutrients at the Water Table beneath Forage Fields Receiving Seasonal Applications of Manure, Washington, 2011–15**—Scientific Investigations Report 2018–5124

ISSN 2328-0328 (online)
https://doi.org/10.3133/sir20185124