NITROGEN AND PESTICIDE CONTAMINATION OF GROUND WATER IN WATER RESOURCES INVENTORY AREA 1

Project Report

Submitted to

Ground water Quality Technical Committee of
WRIA 1 Watershed Management Project

by

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ABSTRACT

NITROGEN AND PESTICIDE CONTAMINATION OF GROUND WATER IN WATER RESOURCES INVENTORY AREA 1

by

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The objectives of the work presented are to perform a preliminary analysis of available ground water quality data related to nitrogen and pesticides in water resources inventory area 1 (WRIA 1) to assess the historical trends and current status of nitrogen and pesticides in ground water and provide recommendations related to future modeling, data gathering, and potential future concerns related to existing land use activities. The work described here is based on ground water quality data obtained directly from the ambient water quality database of the Washington Department of Ecology, and other databases of Washington Department of Health, Whatcom Department of Health, and the US Geological Survey.

WRIA 1 is in northwest Washington State, and covers more than 1,250 square miles. WRIA 1 is bounded on the north mostly by the Canadian border, but also includes the boundaries of the Sumas River drainage that extend into British Columbia, Canada. WRIA 1 includes most of Whatcom County east of Mount Baker. WRIA 1 is bounded on the south mostly by the Skagit County border, but it also includes portions of Skagit County containing a part of South Fork of Nooksack River. It contains more than 1,000 miles of rivers and streams. The main tributary of the area is Nooksack River, draining more than two-thirds of the area. The western two-thirds of the area is mostly flatland, but the eastern third is mountainous. The study area has seven major cities in the US portion, i.e., Bellingham, Blaine, Everson, Ferndale, Lynden, Nooksack, and Sumas; all located in the western part of the study area where heavy agricultural practices are present.

Ground water is a major source of drinking water in much of the area. The primary source of ground water is the Sumas-Blaine water table aquifer; also known as the Sumas-Blaine aquifer. The depth to the water table of this aquifer is shallow, typically less than 10 feet from the land surface. The other surficial aquifers are the Upper Valley surficial aquifer and the discontinuous surficial aquifers. Surficial aquifers are not present in some parts of the area.

There have been concerns in recent times of contamination of ground water in WRIA 1 by nitrogen, most notably nitrates. This study compiled a composite ground water quality database for WRIA 1 with data from previously mentioned agencies. There were 3,831 ground water wells contributing 9,842 measurements of nitrate concentration from 1990 to 2000. On a yearly basis, the number of wells with nitrate concentration data ranged from 214 to 747 contributing 494 to 1,322 data points annually.

In order to assess the anthropogenic effects on ground water quality, the nitrate data were analyzed by classifying the water quality measurements into four concentration ranges: 0-1 mg/l, 1-3 mg/l, 3-10 mg/l, and >10 mg/l, in order of increasing contamination. At sub-basin level, the number of wells per sub-basin ranged from 0 to 555. The highest percentage of wells in any sub-basin exceeding the maximum contaminant level (MCL) of nitrate at least once during 1990 to 2000 was 35.8%. This observation was recorded in Bertrand/Fishtrap/Kamm sub-basin, which straddles the US-Canadian border. The annual mean nitrate concentration in this area has been rising steadily since 1990. The maximum nitrate concentration recorded in the early 1990s was from the Barrett Lake sub-basin with a value of 260 mg/L. In the latter part of the 1990s, Bertrand Creek sub-basin recorded concentrations as high as 39 mg/L in 1999 and 20 mg/L in 2000. Sumas River sub-basin recorded nitrate concentrations as high as 26 mg/L in 1998. In general, the annual maximum, mean, and median [NO3+NO2] concentrations have been increasing in several other basins in the northern part of WRIA 1 indicating ground water quality concerns. Available data on ammonia and organic nitrogen were limited. The results also showed that baseline concentrations of nitrate, ammonia, and organic nitrogen are 1, 0.1, and less than 0.5 mg/l, respectively.

The results indicated that the percentage of sampled wells with high nitrate concentration increased with time. This observation together with other results showed that nitrate is a concern in ground water and will remain to be a concern due to heavy agricultural activities in most parts of WRIA 1, shallow ground water depths, and high permeability of the main aquifer. Hence, continuous field monitoring across WRIA 1, especially in areas currently affected by nitrogen, should be carried out. In addition, a suitable modeling framework should be developed to assist in management decision-making relevant to future ground water development, effects of such development on existing water quality, and agricultural practices using nitrogen-based fertilizer and other chemicals.

Pesticides have been major contaminants in ground water of WRIA 1 in the past based on previous studies. The existing ground water quality data indicate some reduction of pesticide concentrations in certain parts of WRIA 1. However, there is a continuing concern of pesticide contamination in some sub-watersheds of WRIA 1. A careful look at the existing data shows that these data have been collected from areas with heavy agricultural practices and known prior contamination. Therefore, the available data do not necessarily indicate a good representation of ground water quality across WRIA 1.

Analysis of existing data of 12 pesticides indicated that two previously banned pesticides from public use, ethylene dibromide (EDB) and 1,2-dichloropropane (1,2-DCP), are of serious concern in the Fishtrap Creek and Bertrand Creek sub-basins and pose a threat to ground water quality in Schneider, Kamm, Scott, and Johnson sub-watersheds. The available data showed that the annual maximum concentration of EDB decreased across WRIA 1. However, the concentration of EDB in the Bertrand Creek sub-watershed was as high as 0.186 ppb in 1999 compared to the MCL of 0.05 ppb. The behavior of 1,2-DCP was similar and the concentrations were as high as 11 to 19 ppb in the Bertrand Creek and Fishtrap Creek sub-watersheds in 1997 when the MCL is 5 ppb.

Due to the high permeability of the Sumas aquifer, shallow water table, and resistance to degradation in soil and ground water, pesticide transport by advection and dispersion with some natural attenuation is a possibility in the affected sub-watersheds. On the other hand, the high concentrations still remaining after nearly 15 years after the ban can also suggest that local application sites have residual pesticides that are undergoing slow natural attenuation without much transport. Although both of these possibilities are viable in these sub-watersheds, the available data are not adequate, in terms of monitoring frequency and spatial distribution of monitoring locations, to rigorously analyze these possibilities.

In any event, the threat to ground water quality due to pesticides remains a concern, especially for agricultural areas of WRIA 1. Due to the low to moderate runoff characteristics, high permeability, and shallow water table in the major agricultural areas, ground water is vulnerable to pesticide contamination. Therefore, the existing pesticide contamination will remain to be a major concern in the future as well.

Based on the findings from this study of nitrogen and pesticides, the key recommendations are to (a) implement a long-term field monitoring scheme across WRIA 1; the objective here is to gather a consistent set of long-term data from spatially distributed locations representing both heavily contaminated areas as well as other areas of WRIA 1 with minimal potential for contamination, (b) gather information on natural attenuation and sorption characteristics of pesticides through soil sampling at selected locations of affected sub-watersheds, (c) develop suitable modeling tools to assess the effectiveness of various management alternatives, and (d) re-assessment of ground water quality after a set of consistent long-term ground water quality data is available.

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1. INTRODUCTION

Watershed management involves understanding a number of factors, such as surface water flow and quality, ground water flow and quality, in-stream flow, fish habitat, and climate. A proper watershed management plan for Water Resources Inventory Area 1 (WRIA 1) can be implemented only after analyzing surface water and ground water flow and quality, climate, in-stream flow, and fish habitat. Ground water plays a vital role in sustainable watershed management. Understanding the ground water flow and quality issues in a watershed is among the basic steps towards implementing a proper watershed management plan.

The purpose of the WRIA 1 watershed management project is to develop an integrated water management plan for WRIA 1, considering all interconnected physical processes contributing to water balance. Utah Water Research Laboratory (UWRL) of Utah State University (USU) was given the task of developing necessary analyses and tools to support the development of the watershed management plan and the proposed goals will be accomplished through a phased approach. The project includes six major areas of interest to water and environmental management and sustainability. These six integrated areas and the corresponding principal investigators from USU are given below:

Surface water hydrology	Drs. David Tarboton and Mac McKee
Ground water hydrology	Dr. Mariush Kemblowski
Ground water quality	Dr. Jagath Kaluarachchi
Surface water quality	Dr. David Stevens
Instream flows and fish habitat	Dr. Thom Hardy
Database management and decision-support system	Dr. Robert Pack

The work described in this report is related to the Phase II work on nitrogen and pesticide in ground water. The specific objectives of Phase II ground water quality are as follows:

- Develop a comprehensive ground water quality database of WRIA 1.
- Assess the historical trends of pesticide and nitrogen in ground water.
- Determine the current status of nitrogen and pesticide contamination of ground water.
- Provide recommendations related to future monitoring requirements, data adequacy and gaps, modeling needs, and management interventions needed to preserve ground water quality.

Phase III of this work will continue to analyze other ground water pollutants such as organics, heavy metals, and inorganics such as chlorides, dissolved oxygen, pH, etc. together with detailed modeling of nitrogen across WRIA 1.

The work described in this report is based on the ground water quality database prepared by USU, and a MS Access copy of the database is available from the principal investigator. The database was developed using data from four major agencies in WRIA 1, and these are Washington Department of Health, Washington Department of Ecology, Whatcom County Health Department, and US Geological Survey (USGS). The information in the database includes descriptions of each monitoring well and time series of more than 30 ground water quality parameters from the 1980's to 2000. Since the data availability prior to 1980 is limited, the subsequent analysis was performed using data collected after 1990. It should be noted that the data available for year 2000 are limited as the database preparation commenced in June, 2000. Also, some field monitoring in 2000, for example on pesticides, were conducted to fill in the data gaps only. Therefore, year 2000 data are not complete and should not be considered as a true representation of ground water quality for this year.

As the database was prepared and used in this work, quality control checks were performed as much as possible to ensure the data are correct and representative of the location and the time of measurement. However, as most data points have been circulated among many different databases over the past decade or so, the USU team cannot guarantee the validity and accuracy of the entire database. Therefore, the information, results, and discussion presented in this report will be subject to the limitations of the database.

The work described in this report is related to the preliminary analysis of pesticide and nitrogen contamination of ground water in WRIA 1 with the aim of extending the findings to an integrated framework through future tasks. Where necessary, the report contains information related to ground water hydrology, surface water hydrology, and surface water quality for completeness sake. However, the reader is advised to refer to other reports pertaining to work of other areas (as indicated earlier) to obtain a broader view of the WRIA 1 watershed management project and other related technical details.

1.1 Physical Description of WRIA 1

WRIA 1 is located in the northwest corner of Washington State (see Figure 1.1). The study area includes the drainage area of the Nooksack River and its tributaries, including portions of Skagit County, which are drained by south fork of Nooksack River. WRIA 1 has seven towns in the US portion of the basin which are located within the western half of the study area. These towns are Bellingham, Blaine, Everson, Ferndale, Lynden, Nooksack, and Sumas. The study area also includes the Sumas-Blaine aquifer, which extends across the international border into Canada. In addition, the study area includes several coastal drainages that drain into the marine waters along the coastline of Whatcom County. The western-most portion of WRIA 1 is generally flat where the elevation mostly remains below 100 feet MSL for about 20 miles inland. The changes in

elevation toward the east then become sharp. Numerous rugged mountains with high peaks are found in the eastern portion of the study area, with Mount Baker at 10,775 feet above MSL and Mount Shuksan at 9,127 feet above MSL. Surface water sources originate in the northern Cascades. The Nooksack River and its tributaries drain WRIA 1 for the most part with water flowing westerly through the Nooksack River to Bellingham Bay. A lesser amount of water flows northerly into British Columbia through the Sumas River, emptying into the Fraser River. British Columbia contributes surface water to the lower part of the Nooksack River through three south-flowing tributaries that cut through the Lynden Terrace.

Figure 1.1 Location of WRIA 1 in the State of Washington.

1.1.1 Nitrate and Pesticide Contamination

High concentrations of nitrate have been detected in the surficial aquifers of the WRIA 1 since the early 1970's. Historical ground water quality data showed nitrate contamination in WRIA 1 is widespread. Nitrate concentrations higher than the US Environmental Protection Agency (US EPA) prescribed maximum contaminant level (MCL) of 10 mg/l have been detected in several wells in the area. Previous studies have shown (see Cox, Kahle, 1999 for details) that the main sources of nitrate contamination have been inorganic fertilizer applications, the widespread practice of land applications of dairy and poultry manure, and the use of domestic septic tank systems in unincorporated areas of WRIA 1.

The main areas with high concentrations of nitrate have been identified as the northeast portion of the Sumas-Blaine Aquifer in the vicinity of Judson Lake and the Bertrand Creek drainage, and the area south of Lynden (Erickson, 1998).

Pesticide contamination in WRIA 1 is more of historical significance than of immediate importance. The contamination of ground water due to pesticides is primarily due to the use of soil fumigants. The principal aquifers that support agriculture are shallow unconfined aquifers which make the aquifers in WRIA 1 more vulnerable to pesticide contamination.

1.1.2 Land Use Practices

The land use practices in WRIA 1 have been more or less the same for many years. Figure 1.2 shows the extent of land use in Whatcom County. The sub-watersheds drained by North Fork, Middle Fork, and South Fork are mostly covered by forests. However, glaciers such as Mount Baker cover a minor part of the above-mentioned areas. Heavy agricultural practices are followed in the area supported by the Sumas-Blaine Aquifer, which is located in the western part of the study area. Figures 1.3 through 1.7 show the land use patterns in WRIA 1 for a selected number of years. The figures clearly indicate a high concentration of agricultural practices and forest cover in the western and eastern parts of WRIA 1, respectively. Agriculture and forestry have traditionally been important to the economy of WRIA 1. As the valleys were cleared for agriculture, the lumber industry became less dominant, although still important. Dairy farming and fishing have long been the primary land-use activities in the area. While fishing has declined in recent years, the area has remained a major base for the Alaskan fishing industry.

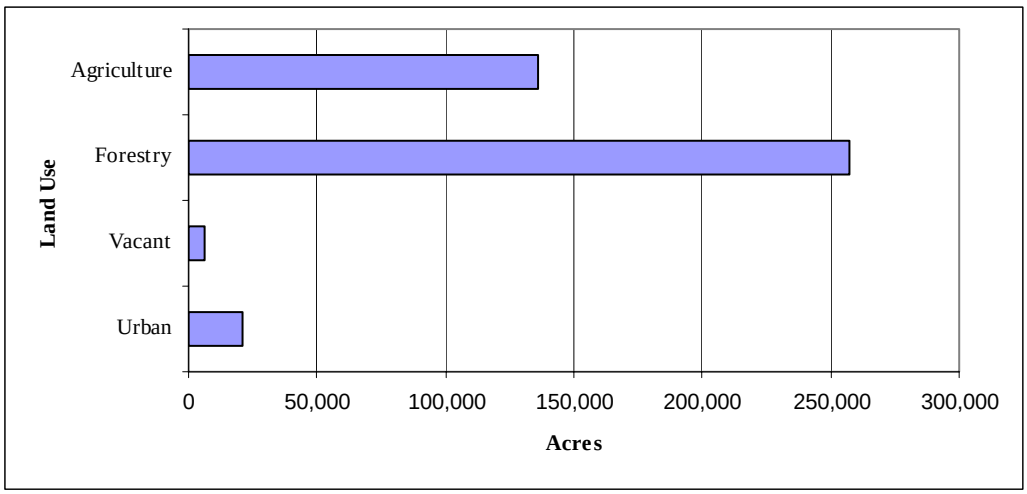


Figure 1.2. Land use distribution in Whatcom County excluding tribal and federal lands (Nelson and others, 1985).

Land use in WRIA 1 can be grouped into three categories and these are:

- Forest and cleared area in the eastern half of the study area
- Agricultural areas dominating the western half
- Urban and industrial areas along the I-5 corridor, western coastal areas, and other population centers

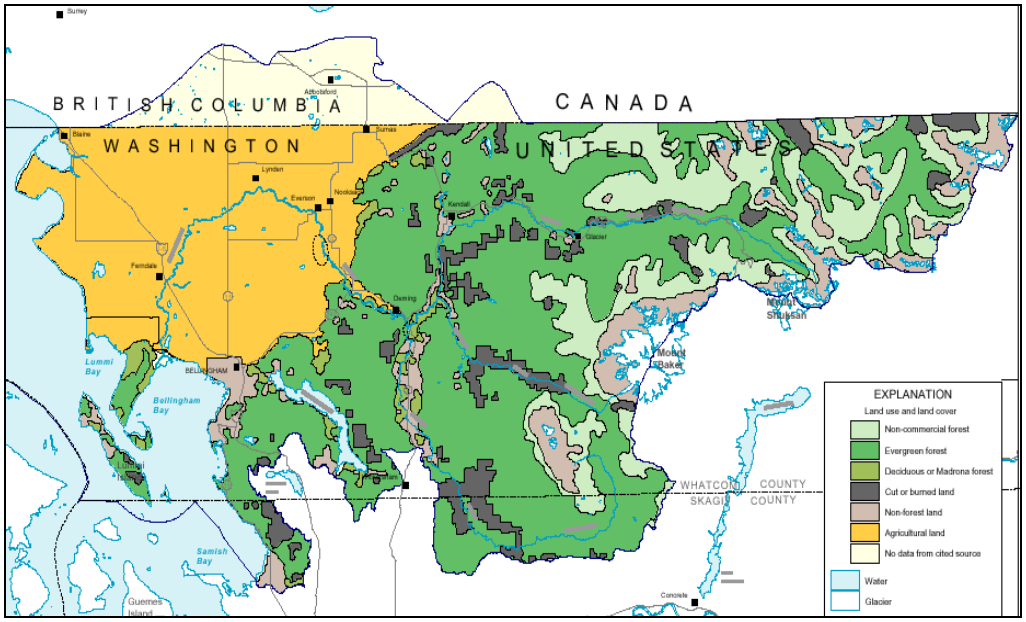


Figure 1.3. Land use and land cover of WRIA 1 in 1936 (http://wa.water.usgs.gov/wria1/maps.html).

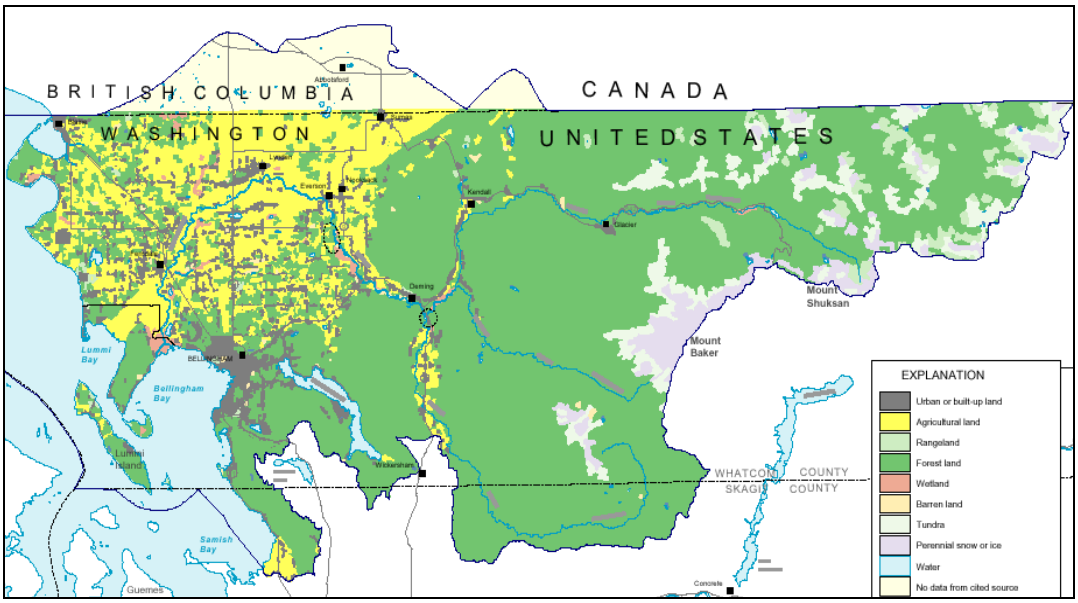


Figure 1.4. Land use and land cover of WRIA 1 from 1973 through 1976 (http://wa.water.usgs.gov/wria1/maps.html).

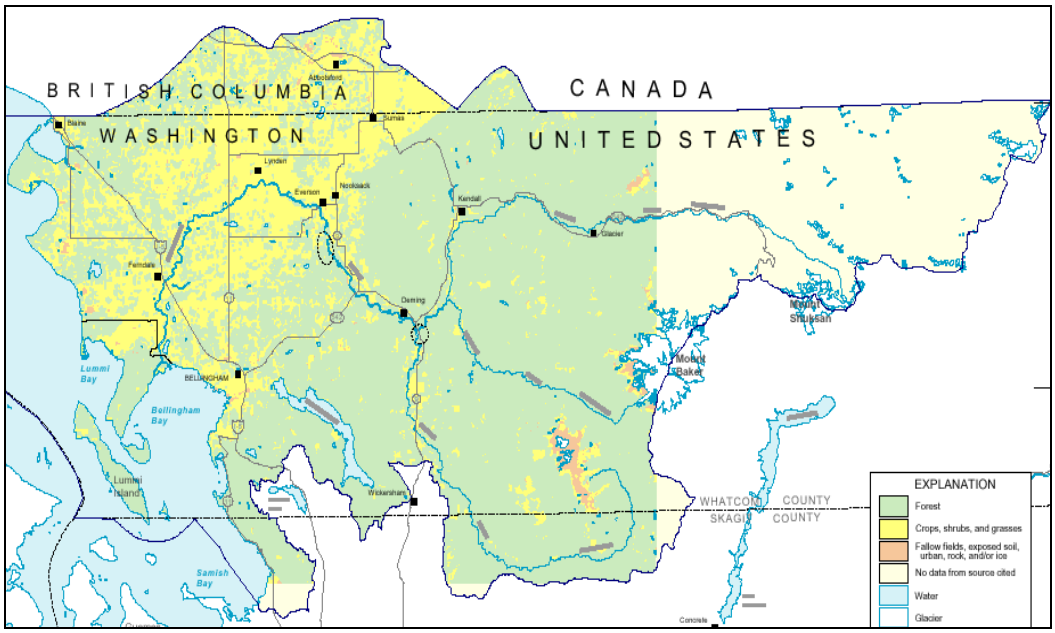


Figure 1.5. Land use and land cover of WRIA 1 in 1991 (http://wa.water.usgs.gov/wria1/maps.html).

Figure 1.6. Land use and land cover in the Lynden-Everson-Nooksack-Sumas (LENS) area of WRIA 1 in 1992 (http://wa.water.usgs.gov/wria1/maps.html).

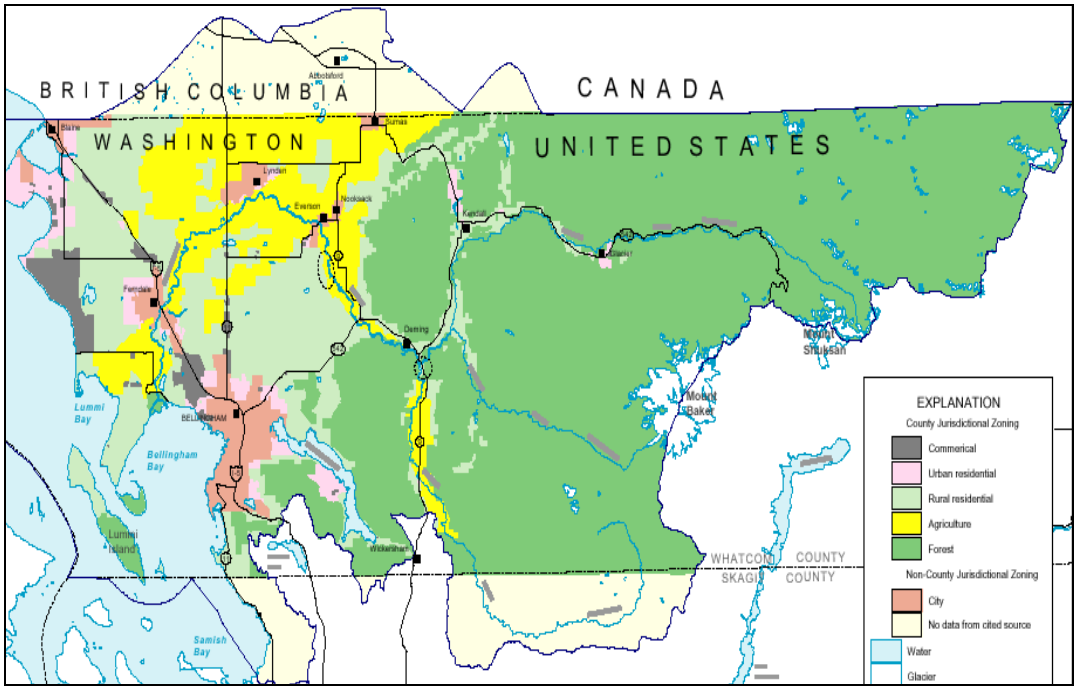


Figure 1.7. Projected future land use in Whatcom County (http://wa.water.usgs.gov/wria1/maps.html).

1.1.3 Agricultural Practices

Agricultural practices are concentrated in the western part of WRIA 1, particularly in the Lynden-Everson-Nooksack-Sumas (LENS) area. Whatcom County, which covers a major portion of WRIA 1, ranks sixth out of 39 counties in the state for agricultural production (USDA, 1997). Whatcom County is one of the top producers of dairy products and is among the top 0.5% in US for dairy production (USDA, 1997). In the Canadian portion of WRIA 1, berry cultivation, and dairy and poultry industries are common. Also, berries form a major part of crop production in the Canadian portion of WRIA 1. It may be concluded that the agricultural practices are more or less the same on both sides of the international boundary.

Dairy production is a major practice in the study area. The dairy industry is among the prominent industries in the study area. Table 1.1 shows the extent of dairy production in Whatcom County.

Major crops grown in WRIA 1 include raspberries, strawberries, blueberries, corn, potatoes, and peas. Raspberries, blueberries and nursery stock are the dominant cash crops. Most of these crops are irrigated using rills or sprinklers. Table 1.2 shows the average harvest during 1989-1998 for each crop. The table clearly shows a significant increase in the average harvest of berries.

Table 1.1. Annual number of livestock in Whatcom County from 1989 to 1998 (http://www.nass.usda.gov/wa/annual98/livstk98.htm).

Livestock	1989	1991	1992	1994	1996	1998
Cattle & Calves	110,000	123,000	124,000	124,000	114,500	115,000
Beef Cows	6,400	6,900	7,000	6,500	5,500	NA
Milk Cows	54,400	59,300	60,000	65,100	67,600	64,900
Hogs & Pigs	1,800	1,500	600	100	100	300
Sheep & Lambs	500	400	400	600	500	NA
Chickens	2,798,000	1,870,000	2,580,000	3,100,000	2,700,000	NA
Mink (breeding)	6,300	5,700	5,000	5,600	3,750	4,700
Mink (pelts)	22,500	20,500	21,100	17,650	17,050	20,900

Table 1.2. Average crop harvest (in acres) in Whatcom County from 1989 to 1998 (http://www.nass.usda.gov/wa/annual98/conten98.htm).

Crop	1989	1991	1992	1994	1996	1998
Silage Corn	9,000	9,400	9,300	10,000	13,500	17,600
Sweet Corn	650	900	1,000	1,300	900	0
Potatoes	1,200	1,300	1,300	1,600	1,600	2,400
Alfalfa Hay	2,600	1,700	1,200	3,000	3,600	800
Other Hay	30,600	28,000	26,000	22,300	21,800	16,000
Blueberries	320	350	480	520	480	450
Red Raspberries	2,210	3,400	3,500	3,800	4,200	6,000
Strawberries	400	250	350	450	420	390
Carrots	600	200	200	200	250	390
Green Peas	5,000	2,600	2,900	2,400	1,800	700
Winter Wheat	800	1,000	500	1,900	1,900	0

1.1.4 Agricultural Chemical Usage

Fertilizers and pesticides are used heavily in WRIA 1. Usage of fertilizers is high in the LENS area where intensive agriculture has been carried out regularly. Figure 1.8 shows the use of agricultural chemicals in Whatcom County for over three years. Pesticides are used largely in the western part of the study area where agricultural activities are widespread. Soil fumigants are the most used out of all the pesticides. Soil fumigation for nematodes is common prior to planting major cash crops such as raspberries and seed potatoes. WRIA 1, being among the top producers of berries and seed potatoes in the country, consumes large amounts of soil fumigants. Ethylene dibromide (EDB) and 1,2-dichloropropane were widely used throughout the area before these chemicals were banned from agricultural use for suspected carcinogenic properties. Some other pesticides that were used before they were banned are dibromochloropropane (DBCP) and dinoseb. DBCP is a soil fumigant and was banned in 1985. Dinoseb was used as a herbicide and desiccant on berries, seed potatoes, and legumes in the area before it was banned for most uses in 1987. Metribuzin (lexone, sencor) and metolachlor have replaced dinoseb use on seed potatoes while metribuzin may be used as a replacement for dinoseb on peas (Erickson and Norton, 1990).

Other pesticides that may be considered as widely used are atrazine, and simazine. US EPA has listed the use of dacthal and oxamyl on berries in Whatcom County and currently dacthal is no longer registered to use in the US.

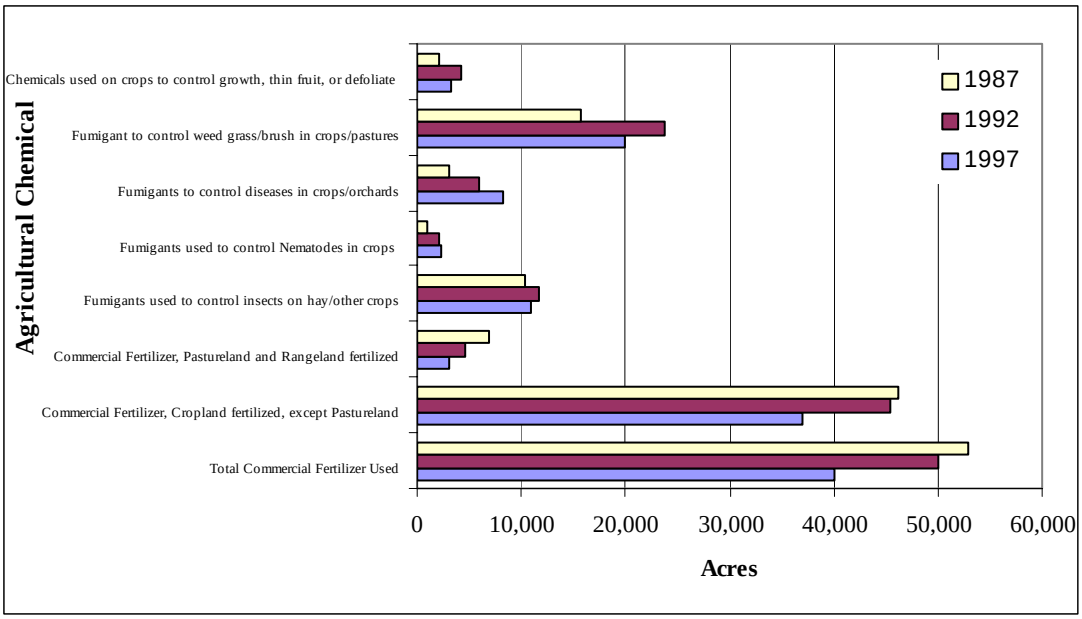


Figure 1.8. Agricultural chemicals used in Whatcom County (http://govinfo.library.orst.edu/cgi-bin/ag-list?10-073.wac).

1.2 Hydrogeologic Features of WRIA 1

1.2.1 Aquifer System

The ground water hydrology of WRIA 1 with respect to hydrogeology, ground water flow, and historical trends of water availability and demand have been addressed in the ground water hydrology part of this project. For detailed information related to ground water quantity, the readers are encouraged to refer to appropriate project reports of Dr. Marius Kemblowski of USU. The information discussed here on ground water hydrology is a summary provided to give a brief overview of WRIA 1.

The aquifer systems of WRIA 1 can be classified into two types; i.e., surficial aquifers and non-surficial aquifers. The principal surficial aquifers, i.e., the uppermost, saturated zone, typically an unconfined water-table condition, are grouped into three aquifer units (Tooley and Erickson, 1996):

- Sumas-Blaine aquifer
- Upper Valley aquifers
- Discontinuous surficial aquifers

A large portion of WRIA 1 is characterized as non-surficial aquifers. Figure 1.9 shows the principal aquifer systems of WRIA 1.

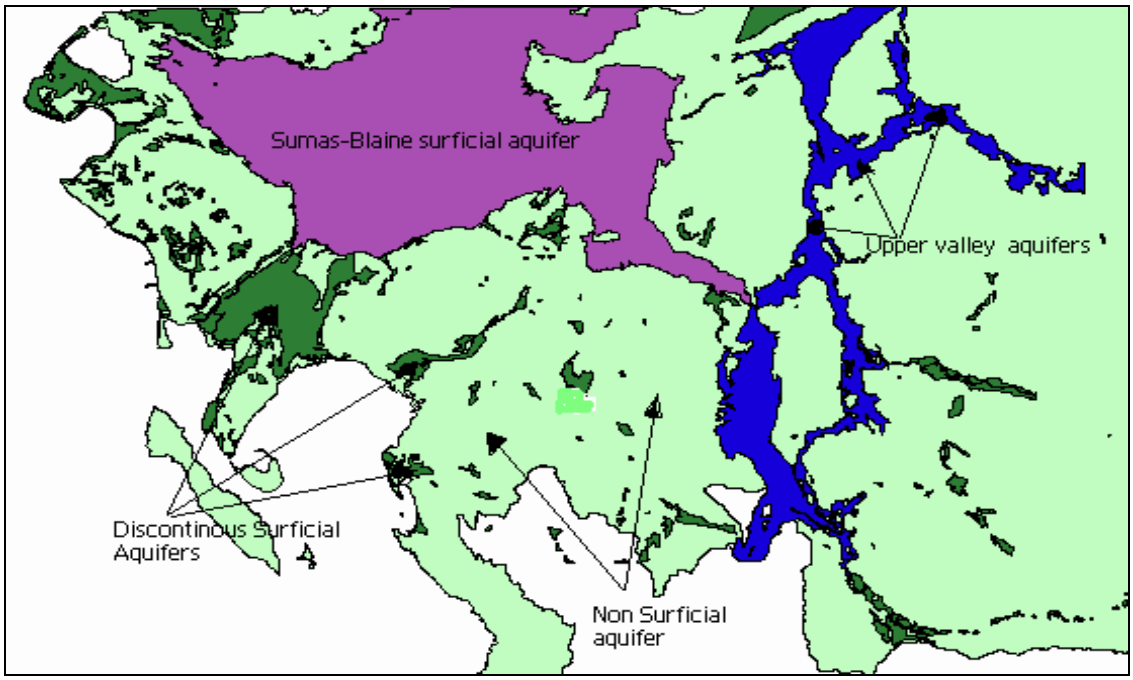


Figure 1.9. Map showing the aquifer systems of WRIA 1.

Sumas-Blaine Aquifer

The principal aquifer in WRIA 1 is the Sumas-Blaine aquifer. This aquifer underlies the flat glacial outwash plain between the towns of Sumas, Blaine, Ferndale, and the Nooksack River, and occupies about 150 square miles. It consists of mostly sand and gravel glacial outwash deposits and alluvial gravel, sand, silt, and clay deposits of the Nooksack and Sumas Rivers (Tooley and Erickson, 1996). The water table is typically less than 10 feet (Morgan, 1999). The vertical extent of the Sumas-Blaine aquifer ranges from 75 feet thick near Sumas in the eastern edge to less than 25 feet near Blaine at western edge.

Discontinuous Surficial Aquifers

There are many discontinuous surficial aquifers spread throughout WRIA 1, both within and outside WRIA 1. Many of these aquifers are located to the west and southwest of the Sumas-Blaine aquifer, but there are also several smaller aquifers around Lake Whatcom and in the upper valleys. These aquifers consist of many geologic deposits such as beach, glacial-fluvial terrace deposits, modern alluvial and floodplain deposits, isolated outwash terraces, and marine terrace deposits (Tooley and Erickson, 1996). The largest of these aquifers are south of Ferndale, east of Blaine, across the bay southwest of Blaine, and east of Sumas. These aquifers are usually thin and not major sources of water. The lateral boundaries are based solely on parent associations of soils, due to lack of sufficient well data (Morgan, 1999).

Upper Valley Aquifers

The Upper Valley aquifers are associated with North, Middle, and South Forks of Nooksack River. These aquifers consist of inter-layered mixtures of gravel, sand, silt and clay and occupy the river valley bottoms. These aquifers are limited in extent by the surrounding bedrock (Tooley and Erickson, 1996).

Non-Surficial Aquifers

Large portions of the watershed management project study area are characterized as areas with no surficial aquifers present (personal communication with Ground water Quantity Group, Utah State University). These areas are located mainly in the south (except small areas around Lake Whatcom), eastern uplands (except along the river valleys of the North, Middle, and South Forks of Nooksack), and the western coastal areas (except for the Nooksack delta area and pockets of land around Blaine). The central and northern areas, of course, are dominated by the Sumas and Sumas-Blaine surficial aquifer system. The generalized surficial geology of the study area describes the western, coastal and central lowlands as alluvial, terrace, glacial, and other sedimentary deposits; the mountainous eastern lands as sedimentary are meta sedimentary rocks; and the area around the highest peaks of the study area (Mount Baker) as intrusive rocks of granite and intermediate composition.

The aquifer near the US-Canada boundary, except for limited portions under confined conditions below local ice deposits near Sumas, is generally an unconfined water table aquifer.

1.2.2 Surface Water Drainage

Nooksack River is the major drainage in WRIA 1 where it flows from east to west. The major tributaries of the Nooksack River are North Fork, South Fork, and the Middle Fork. Other major tributaries include the Bertrand Creek and Fishtrap Creek. Except for some areas in the north which drain into the Sumas River and some areas that drain into Drayton Harbor, a substantial portion of WRIA 1 drains into Nooksack River. Figure 1.10 shows the major sub-basins in the area. Except for the coastal sub-basins, Sumas River sub-basin, Little Campbell sub-basin, and Fraser sub-basin, all other sub-basins drain into Nooksack River.

Figure 1.10 depicts the various sub-basins in WRIA 1. In addition, WRIA 1 consists of many sub-watersheds. Every sub-watershed is drained by a stream, which in most of the cases joins the Nooksack River. Figure 1.11 shows the various sub-watersheds that form WRIA 1.

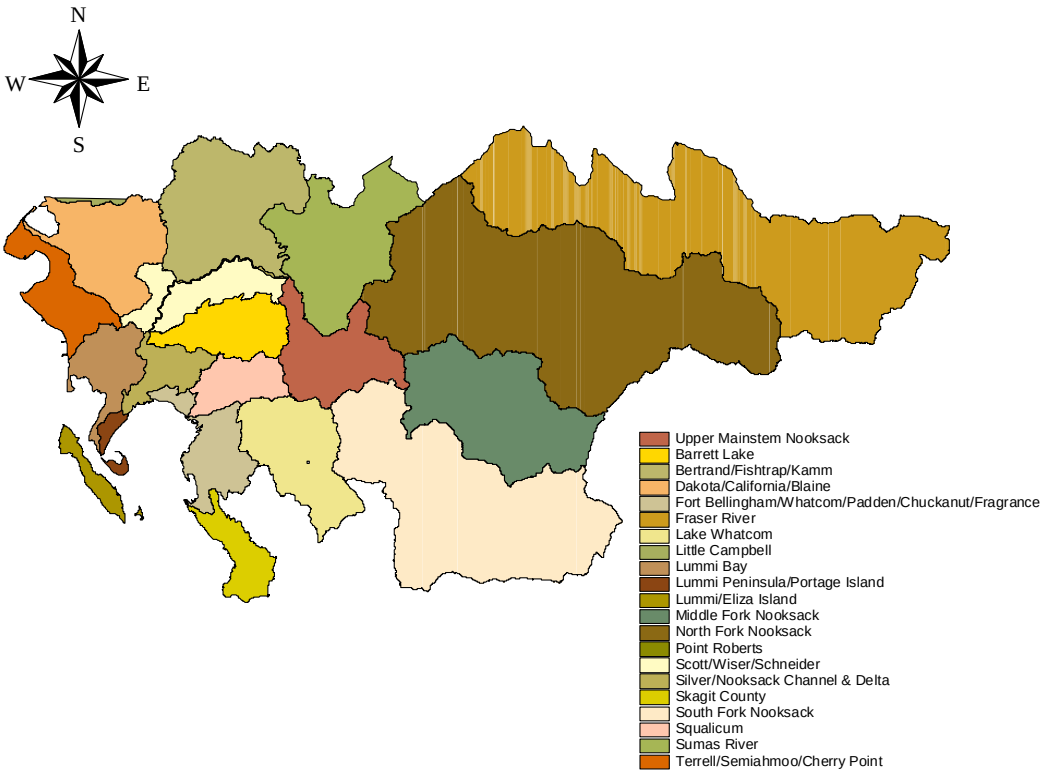


Figure 1.10. Different sub-basins of WRIA 1 used in the present analysis to define the regional ground water quality.

In the subsequent analysis, the regional ground water quality was assessed based on the sub-basin for nitrogen and sub-watershed for pesticides characterization shown in Figures 1.10 and 1.11. Although ground water flow is independent of surface water drainage pathways, this approach was used in the analysis, as these sub-basins or sub-watersheds are the spatial aggregations used in other tasks of the watershed management project and in agreement with the watershed delineation program of WRIA 1.

Recharge to the Sumas-Blaine aquifer is derived from precipitation that falls directly in the upland north of Lynden and in Canada. The high rainfall and relatively shallow water table in the study area cause a portion of precipitation to runoff via surface water drainage ditches. The network of ditches developed to relocate natural streams or drain wetlands in the Lynden area generally follows the local orthogonal road pattern and ultimately discharges into Bertrand Creek, Fishtrap Creek, and Nooksack River, or to Johnson Creek and Sumas River.

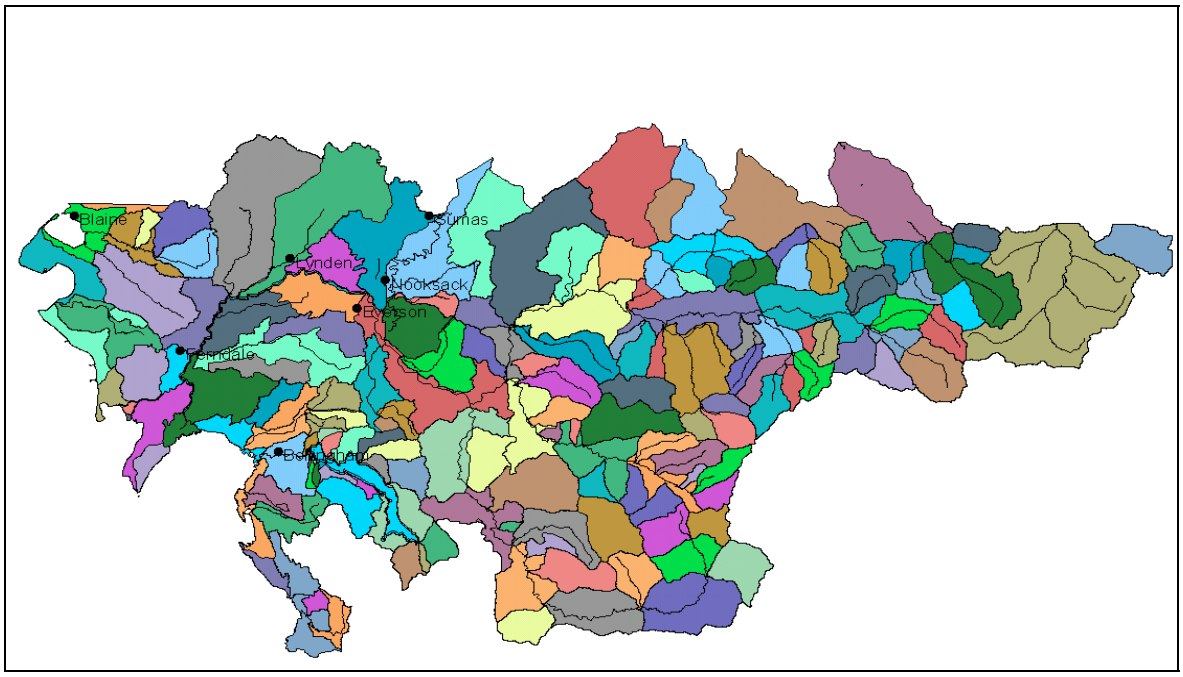


Figure 1.11. Different drainages of WRIA 1.

1.2.3 Ground water Flow

The flow in a ground water system includes the movement of water within and between individual hydrogeologic units and the movement of water into and out of the system. The general movement of ground water in WRIA 1 is from the areas of recharge, typically found in the uplands, to areas of discharge, typically found at lower altitudes along rivers and stream channels (Cox and Kahle, 1999). Most ground water enters the shallow Sumas-Blaine aquifer then travels laterally. Lesser amounts of ground water travel vertically to the deeper Everson-Vashon unit. Ground water flow is mostly lateral at higher depths, toward the Nooksack River, where the flow then becomes mostly upward (Cox and Kahle, 1999). The regional ground water flow direction is from the uplands to the south toward the Nooksack River. The ground water flow path can range in scale from local flow paths that are generally short and shallow to regional flow paths that cover great distances and travel deep within the aquifer system.

Flow conditions are mostly unconfined except in some areas where confined conditions exist. The ground water flow is from the east to west in the eastern part of the aquifers. However, near the Canada-US border, the water predominantly flows from north to south except in a smaller portion drained by the Sumas River where the flow is from south to north. Figure 1.12 shows the ground water flow pattern in the western part of the study area. It can be observed from Figure 1.12 that, in general, the ground water flow direction is towards the draining rivers and/or streams.

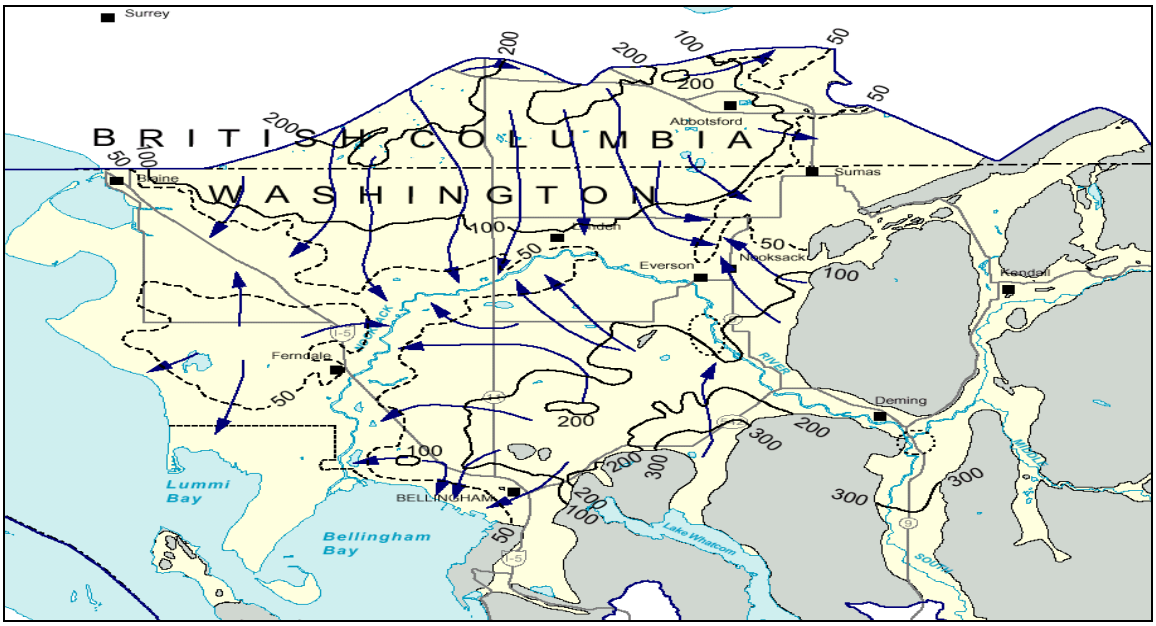


Figure 1.12. Regional ground water flow pattern in WRIA 1.

1.2.4 Geology

The geological framework controls the occurrence and movement of ground water. Newcomb et al. (1949) divided western Whatcom County into two physiographic regions, the lowlands and the uplands. The lowlands of western Whatcom County consist largely of the Nooksack River flood plain, Custer Trough leading northwest to Drayton Harbor and Birch Bay, Sumas River Trough leading northward to Canada and the Fraser River drainage, and the Lynden Terrace.

The uplands are composed of four low plateau areas: (1) a small peninsular area southwest of Blaine called the Birch point upland, (2) boundary upland which extends ten miles eastward from Blaine and across the Canadian border, (3) Mountain View upland west of Ferndale, and (4) King Mountain upland which extends northward from Bellingham to the Nooksack River Valley. According to Newcomb et al. (1949), it may be difficult to obtain ground water in the highest part of the boundary upland, but this may be due to the lack of exploratory drilling. Certain Parts of the Mountain View upland, especially near the western side, parts of the boundary upland, and some places in the alluvial bottom lands in the Everson area lack adequate ground water supply.

The predominant rocks in the eastern highland area include the Paleozoic sediments and volcanic flows, which were metamorphosed during mid-Mesozoic time. These dense, compact rocks are primarily impermeable to either retention or transmission of ground water except through occasional joints and fractures. They permit precipitation to be quickly drained off to surface streams. It is estimated that approximately 70% of total precipitation falling in the mountains will reach the gauging station at Deming (USGS,

1960). These pre-tertiary metamorphic rocks are also exposed in a few places in the foothills, in particular in the Sumas area. A few areas of tertiary igneous rock occur in this region and they are of little importance to water resources of the Nooksack Basin. These areas include the Twin Sisters Mountain and Mount Shuksan at the headwaters of the North Fork Nooksack River.

Sumas-Blaine aquifer consists mainly of sand and gravel glacial outwash deposits and alluvial gravel, sand, silt, and clay deposits of the Nooksack and Sumas Rivers (Tooley and Erickson, 1996). The thickness of the deposits ranges from over 75 feet thick near Sumas to less than 25 feet near Blaine (Tooley and Erickson, 1996). A thick sequence of sandstones, shales, conglomerates of continental type and fresh-water deposits form the bedrock beneath the unconsolidated Pleistocene deposit throughout the lowlands of the Nooksack Basin. The streams flowing from the melting ice front deposited recessional outwash consisting of sand, gravel and other finer material on the broad Nooksack River flood plain area (USGS, 1960), which consists of Holocene alluvial deposits of the Nooksack River (Easterbrook, 1976).

Cox and Kahle, (1999) covered a considerable portion of the Sumas-Blaine aquifer in their work. Four principal hydrogeologic units delineated in the Lynden-Everson-Nooksack-Sumas (LENS) area are, in order of increasing geologic age, the Sumas Aquifer, the Everson-Vashon semiconfining unit, the Vashon semiconfining unit, and the bedrocksemi-confining unit (Cox and Kahle, 1999). The Sumas unit is generally 40 to 80 feet thick, but can be more than 200 feet thick in the northeastern part of the LENS study area. This unit is the thinnest along the Nooksack River channel south of Lynden where it is about 15 feet thick. The Everson-Vashon unit is mostly composed of fine-grained material with scattered lenses of coarse-grained material. The productive zone of this unit, in the south-central part of the LENS study area, is believed to be about a 30-foot interlayer of Deming Sand (Cox and Kahle, 1999). The thickness of the Everson-Vashon semiconfining unit is largely unknown because of few fully-penetrating wells, but according to available drilling records, typical thickness is 100 to 200 feet (Cox and Kahle, 1999). Thickness of the Vashon semiconfining unit is unknown, but probably does not exceed 200 feet (Cox and Kahle, 1999). It consists of small bands of poorly sorted Vashon drift and gravel that are exposed along the eastern part of the LENS area. The bedrock semi-confining unit consists of sandstone, mudstone, conglomerate, and coal. Ground water is likely to occur under unconfined conditions where the bedrock is exposed at or near land surface; and where the bedrock is covered by a significant thickness of glaciomarine drift or till, ground water is likely to be confined.

Cox and Kahle (1999) characterized the vertical extent of the geologic formations in the LENS study area for ten specific cross sections across the district. The California Creek drainage is in the western part of the study area and not contained within the LENS study area. The terrace deposits in this area (Qt) are relatively thin, up to 15 feet, and generally unsaturated (Didricksen, 1997). Peat deposits (Qp) range from a few to about 35 feet thickness. The thickness of the Sumas outwash unit (Qso) is unknown; regionally, it may exceed 50 feet (Didricksen, 1997). The Blaine city well (40/1E-4JI), which penetrates to

560 feet below sea level in the Pleistocene materials, does not reach tertiary rocks (USGS, 1960).

A well has penetrated 746 feet of sediments without reaching bedrock near Blaine and in west of Ferndale, bedrock was encountered beneath 320 feet of Pleistocene sediments. A well reached the bedrock at 615 feet in north of Ferndale (Easterbrook, 1973). At Cherry Point (between Blaine and Birch Bay along the coast) on the Strait of Georgia, wells have penetrated 300 feet of clay. In an area about two miles northwest of Lynden, the outwash deposits range in thickness from about 40 to 50 feet and are underlain by silt and clay deposits with low permeability (Erickson, 1992).

Figure 1.13 shows the various soil hydrologic groups in the area to classify the aquifers by infiltration/runoff conditions. There is no specific pattern in WRIA 1 with reference to soil hydrologic groups. Figure 1.14 shows the maximum hydraulic conductivity distribution in the area. It can be observed that the area, in general, has a maximum hydraulic conductivity of 0.6 to 2 inches per hour except along the rivers and in major portions of the Sumas-Blaine aquifer where high conductivity exists. Figure 1.15 shows the minimum hydraulic conductivity distribution in the area. At this point, it may be of interest to notice the existence of maximum and minimum hydraulic conductivity for a given location. The maximum hydraulic conductivity at any location, in general, indicates the permeability in the unconfined portion of the aquifer, wherein the contamination problem exists and the minimum permeability refers to the permeability in the confining layers of the aquifer.

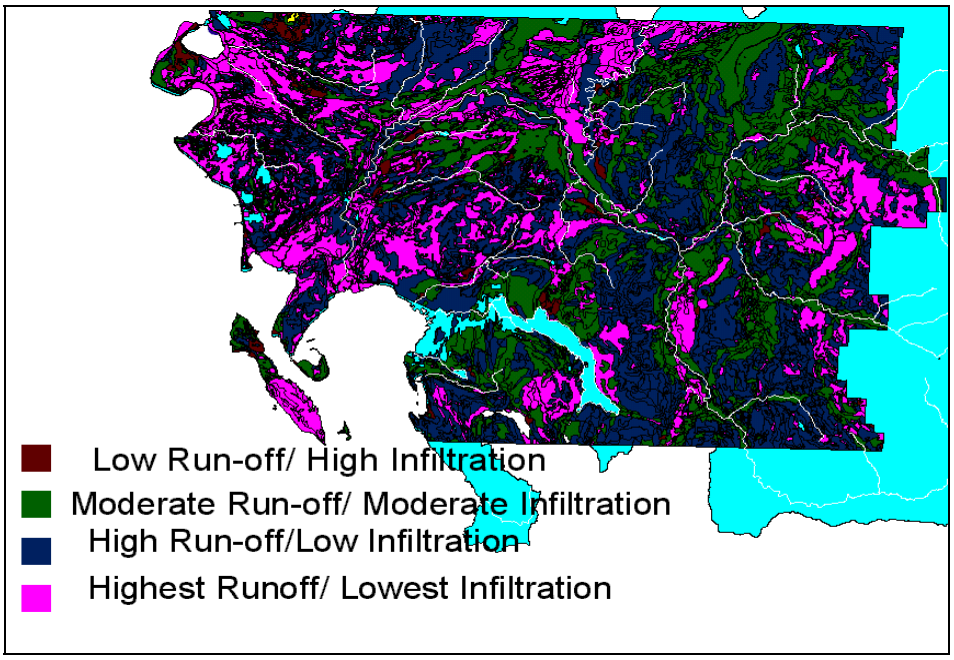


Figure 1.13. Map showing different soil hydrologic groups in the western part of WRIA 1 (from information obtained from Soil Survey of Whatcom County, Washington).

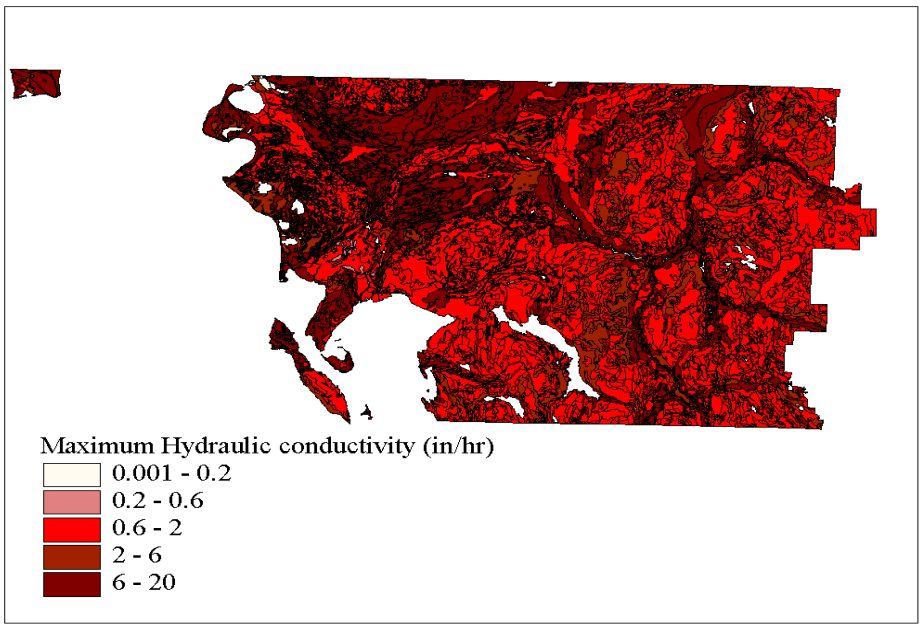


Figure 1.14. Map showing the maximum hydraulic conductivity distribution in the western part of WRIA 1.

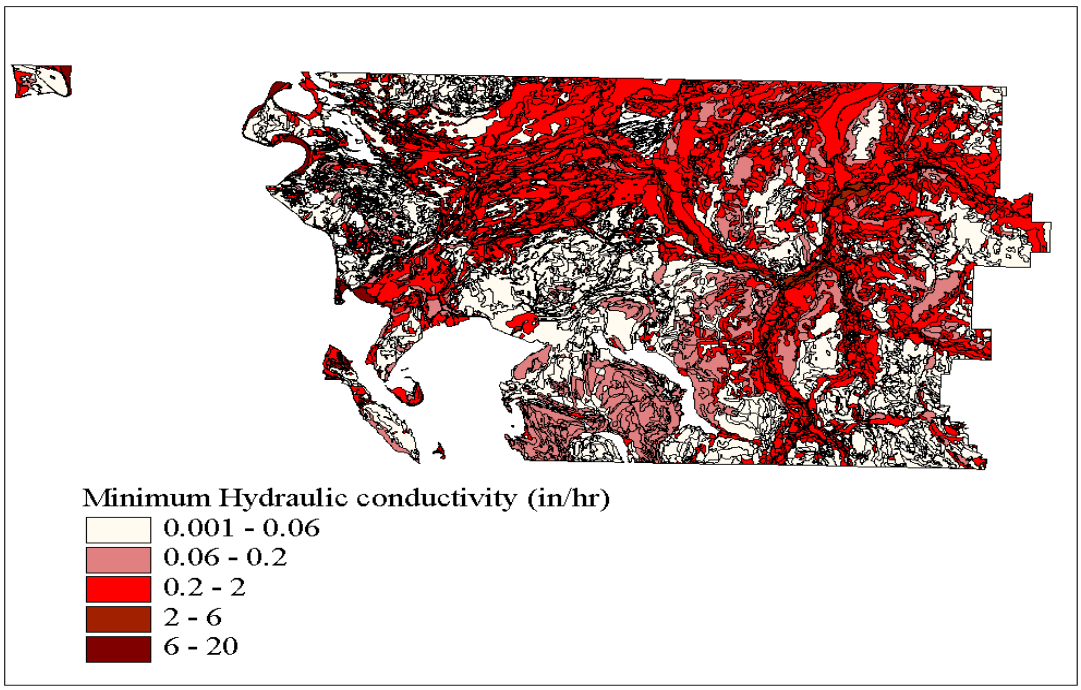


Figure 1.15. Map showing the minimum hydraulic conductivity distribution in the western part of WRIA 1.

2. NITROGEN IN GROUND WATER

2.1 Introduction

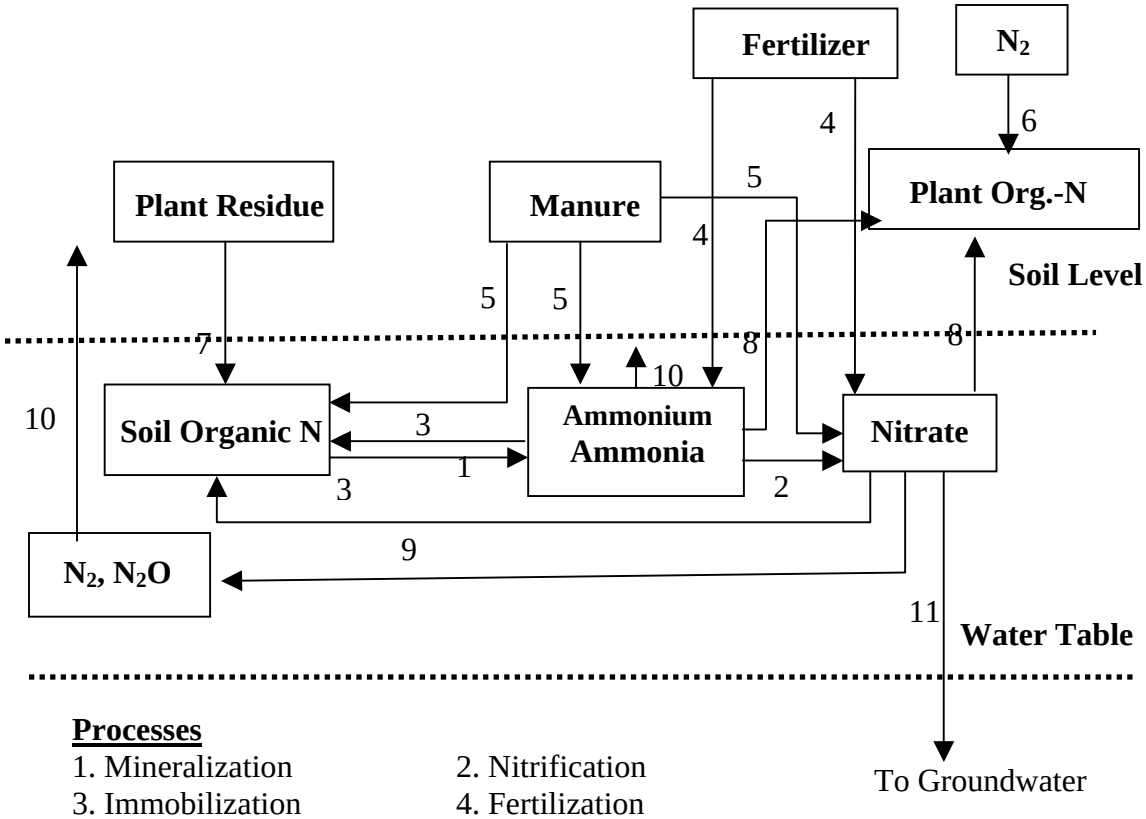
Nitrogen is one of the most essential chemical elements for life on earth. Nitrogen occurs naturally in many different forms in the environment. Nitrogen occurs in various chemical forms in air, soil, and water. There are many chemical and biological processes which transform nitrogen from one form to another in the different environmental media. Nitrogen is an essential nutrient for healthy plant growth and high crop yields. Nitrogen is found in living organisms, in proteins, amino acids, plant matter, fertilizers, human and animal waste products, etc. The growth of all organisms depends on the availability of nutrients, and none is more important than nitrogen, which is required in large amounts as an essential component of proteins, nucleic acids and other cellular constituents.

There is an abundant supply of nitrogen in the earth's atmosphere - nearly 79% in the form of N_2 gas. However, N_2 is unavailable for use by most organisms because there is a triple bond between the two nitrogen atoms, making the molecule almost inert. In order for nitrogen to be used for growth it must be "fixed" (combined) in the form of ammonium (NH_4) or nitrate (NO_3) ions. The weathering of rocks releases these ions so slowly that the process has a negligible effect on the availability of fixed nitrogen. So, nitrogen is often the limiting factor for growth and biomass production in all environments. For this report, the focus will be on nitrate, nitrite, ammonia, and organic nitrogen in ground water.

Nitrate is the most commonly encountered form of nitrogen in streams and ground water. Nitrate is a colorless, odorless, and tasteless compound, and can therefore only be detected using chemical detection equipment. Dissolved ammonia is a less common form of nitrogen. Organic nitrogen is a term used to refer to nitrogen containing organic compounds usually found in soil organic matter.

Even though there are many chemical forms of nitrogen available in soil (see Figure 2.1), most plants can use only nitrate and ammonium. The nitrate ion has a negative charge, while the ammonium has a positive electrical charge. Most soil particles in temperate regions carry a net negative charge and, therefore, repel negative ions. Nitrates are highly soluble in water. Any excess nitrates in soil, above plant needs, dissolve readily in rain or irrigation water percolating through the soil and are carried to ground water. Nitrogen is constantly being transformed from one form to another through the nitrogen cycle. The major processes involved in the nitrogen cycle as they affect leaching of nitrates to ground water are depicted in Figure 2.1.

The key processes of the nitrogen cycle are described below:



Processes

1. Mineralization	2. Nitrification
3. Immobilization	4. Fertilization
5. Manure Application	6. N Fixation
7. Crop Residue	8. Plant Uptake
9. Denitrification	10. Volatilization
11. Leaching	

Figure 2.1. Nitrogen transformation pathways related to fate and transport in ground water.

Nitrification

Classically, nitrification is referred to the conversion of NH_4 to NO_3 via NO_2. The bacteria that carry out this process are the nitrifying bacteria; they are aerobic and chemoautotrophic, and belong to genera with the prefix *nitroso* (-NH_4 to NO_2) and *nitro-* (NO_2 to NO_3). A more detailed taxonomy reveals that the most common genera are *nitrosomoas* and *nitrobacter*.

Nitrosomonas and *nitrobacter* have wider growth ranges than the others and are much more numerous in soils and water. The others tend to inhabit specialized habitats. The growth rates are often slow - generation times of 20 to 40 hours are common in culture and are undoubtedly slower in the natural environment. The process involves the removal of electrons from a hydrated nitrite ion. The reactions of *nitrobacter* are inhibited by small quantities of ammonia gas. This causes ammonia in soils leading to the accumulation of nitrite since only *nitrobacter* is inhibited. Nitrite is continually formed

by *nitrosomonas* but is not utilized by *nitrobacter* when it is inhibited. This nitrite can accumulate to levels toxic to plants.

Denitrification

Denitrification is the only reductive mechanism through which nitrate is reduced to nitrogen gas via N_2O. Nitrate is replacing oxygen as the electron acceptor in the process when oxygen is completely exhausted. Only facultative anaerobic bacteria carry out this process and these bacteria are relatively common soils. Denitrification is one of the major sources of loss of N from soils. Typically, N_2O will be produced early in the reaction and nitrogen will be predominant later. At pH less than 6, N_2O is present, but at higher pH levels, N_2O is produced first followed by nitrogen gas. A pH of about 5.5 is the lower limit of activity. The bacteria can exist and grow aerobically, but only produce N_2O and nitrogen under anaerobic conditions. The optimum temperature for denitrification is usually about 25 °C.

Nitrogen Fixation

Some bacteria can convert N_2 into ammonia by the process termed nitrogen fixation. Three processes are responsible for nitrogen fixation in the biosphere and these are (a) atmospheric fixation by lightning, (b) biological fixation by certain microbes - alone or in a symbiotic relationship with plants, and (c) industrial fixation.

(a) Atmospheric Fixation

The enormous energy of lightning breaks nitrogen molecules and enables their atoms to combine with oxygen in the air forming nitrogen oxides. Nitrogen oxides dissolve in rain, forming nitrates that are carried to the earth. Atmospheric nitrogen fixation probably contributes to about 5-8% of the total nitrogen in the atmosphere.

(b) Biological Fixation

The ability to fix nitrogen is found only in certain bacteria. Some live in a symbiotic relationship with plants of the legume family (e.g., soybeans, alfalfa). Some establish symbiotic relationships with plants other than legumes (e.g., alders). Some nitrogen-fixing bacteria live free in the soil. Nitrogen-fixing cyan bacteria are essential to maintaining the fertility of semi-aquatic environments such as rice fields. Although the first stable product of the process is ammonia, this is quickly incorporated into protein and other organic nitrogen compounds.

Mineralization

Mineralization is the conversion of organic nitrogen into inorganic forms when it re-enters the biogeochemical cycle via decomposition. Decomposers, found in the upper soil layer, chemically modify the nitrogen found in organic matter from ammonia (NH_3) to ammonium salts (NH_4) which is called ammonification. A wide variety of fungi and bacteria can produce ammonification.

Immobilization

Immobilization is the process opposite of mineralization where soil organisms convert ammonium and nitrate in the soil to organic forms of nitrogen.

2.1.1 Health Effects of Nitrogen in Ground water

Ground water is a major source of drinking water for many people across the United States, including inhabitants of WRIA 1. Ingesting high doses of nitrate can induce a health condition known as methemoglobinemia (blue baby syndrome) in humans and animals which can result in death or brain damage.

Nitrate is converted by bacteria to nitrite in the digestive system of humans and animals. Nitrite absorbed into the bloodstream combines with hemoglobin to form methemoglobin, which reduces the oxygen-carrying capacity of blood. Most adults have the ability to convert methemoglobin back to oxyhemoglobin. Infants, especially under six months of age, do not have this capability and are, therefore, very susceptible to methemoglobinemia. This condition occurs mainly because they lack an enzyme for converting methemoglobin to hemoglobin. Also, their stomachs are less acidic and, therefore, more favorable for the multiplication of bacteria which convert nitrate to nitrite. Adults with certain gastrointestinal system disorders may also be at risk when exposed to high concentrations of nitrates. There have also been some reports of birth defects when pregnant women were exposed to water contaminated with high nitrate levels. It has also been reported that nitrate can react with amines or amides in the body to form carcinogenic substances (Jasa and others, 1998; Self and Waskon, 1998).

Younger animals are affected by nitrates in the same way humans are affected. Therefore it is recommended not to allow domestic animals to consume water containing more than 100 mg/l of nitrate (as nitrogen). However, toxicity depends on the rate at which nitrate is consumed. Symptoms of nitrate poisoning include abdominal pain, diarrhea, muscular weakness or poor coordination, and blood with chocolate-brown color (Self and Waskon, 1998). Animals such as cattle and sheep and horses are more prone to risk than chicken and swine because the systems of the latter rapidly remove nitrates through the urine.

In addition to the health hazards of drinking water polluted with nitrates, one other reason the level of nitrates is important is because it is often an indicator of other ground water pollutants. If the source of nitrates in a particular aquifer is animal wastes or effluent from septic tanks, then bacteria, viruses, and protozoa can most likely to be present. If the source is inorganic fertilizers, this may indicate the probable presence of other agro-chemicals (Nugent and others, 1993). Because it is such a common contaminant of ground water, it has been suggested as an indicator of overall ground water quality (US EPA, 1996).

2.1.2 Nitrate Contamination of Ground water

High concentrations of nitrate have been detected in surficial aquifers of WRIA 1 since the early 1970's. Historical ground water quality data show that nitrate contamination in WRIA 1 is widespread. Nitrate concentrations higher than the maximum contaminant level (MCL) defined by the US EPA of 10 mg/l (as nitrogen) have been detected in several wells in the area. Previous studies have shown that the main sources of nitrate contamination are inorganic fertilizer applications, the widespread practice of land

applications of dairy and poultry manure, and the use of domestic septic tank systems in unincorporated areas of WRIA 1.

The major areas with high concentrations of nitrate are the northeast portion of the Sumas-Blaine aquifer in the vicinity of Judson Lake and the Bertrand Creek drainage, and the area south of Lynden (Erickson, 1998).

2.2 Sources of Nitrogen in Ground water

Ground water is recharged by percolation of water from direct precipitation, streams, lakes, rivers, wetlands, and other surface water bodies. Thus chemicals in the recharging water are transported into ground water and subsequent transport in aquifers cause widespread pollution of ground water. There are natural and anthropogenic sources of nitrogen species to ground water.

2.2.1 Natural Sources of Nitrate

The common sources of nitrogen in percolating rainwater are:

- Decaying plant and animal organic matter on or in soil
- Lightning during rainstorms
- Nitrogen-fixing microorganisms in soils, such as blue-green algae
- Leguminous plants, such as peas, clover and alder

2.2.2 Anthropogenic Sources of Nitrate

The anthropogenic sources of nitrogen to ground water include:

- Inorganic fertilizer application to farms and lawns
- Manure application to farms and fields
- Animal feedlots
- Municipal wastewater effluent
- Domestic septic tank systems and drainfield sites

In WRIA 1, domestic septic tank systems for the disposal of domestic waste are common in unincorporated areas. Inorganic nitrogen fertilizers are used in the production of many crops in the area. Applications of barnyard manure to land are common also (Cox and Kahle, 1999).

2.2.3 Influence of Canadian Land Use Activities

The main water table aquifer in WRIA 1, the Sumas-Blaine aquifer, extends into Canada, and is also known as the Sumas-Blaine aquifer. Since this aquifer is unconfined and composed largely of permeable materials such as sand and gravel, and the ground water

level is shallow, contaminants from various land-use activities can easily reach ground water.

The soils overlying the Canadian portion of this aquifer have moderate to high permeabilities (Luttmerding, 1981). The main land use activity over the aquifer in Canada is agriculture; mainly raspberry and poultry production (Hii and others, 1999). Fertilizer and animal manure applications on raspberry farms have been identified as the major sources of nitrates to the aquifer. Poultry production is the main source of the animal manure (Hii and others, 1999).

The Sumas-Blaine aquifer (the USA portion of the aquifer) is recharged primarily by direct infiltration of precipitation. However, since it is also hydraulically connected to the Canadian portion of the aquifer and the general ground water flow direction is from Canada towards the US, it also receives some recharge by throughflow from the Sumas-Blaine aquifer (Erickson, 1998). Therefore, land use activities on the Canadian portion of the aquifer contribute to the nitrate levels in the US portion of the aquifer.

2.3 Nitrogen Contamination of Ground water in WRIA 1

The maximum allowable nitrate concentration in drinking water is 10 mg/l (as nitrogen). Land use activities lead to increases in concentrations of chemicals in ground water. Contaminant levels in the underlying ground water are indicators of the influence of surface activities. Nitrogen is found in ground water in several different forms. The nitrogen species considered in this study are nitrate, nitrite, ammonia, and organic nitrogen.

The data used in this study of WRIA 1 were from four main agencies, i.e., US Geological Survey (USGS), Whatcom County Department of Health, Washington State Department of Health, and Washington State Department of Ecology. All available data were assembled into one composite database. The total number of wells in the database contributing data for all chemicals found in ground water is 4,247.

Some of the data collection agencies analyzed nitrate concentrations as the sum of nitrate and nitrite ([NO3+NO2]). For consistency, in cases where agencies reported separate concentrations for nitrate and nitrite, these were combined into [NO3+NO2] values. This approach did not affect the quality of the analysis as Liebscher and others (1992) note that typically more than 99% of the [NO3+NO2] value in ground water is in nitrate form. Therefore in this report, nitrate is represented as [NO3+NO2]. The ammonia and organic nitrogen data are for total concentrations measured in ground water.

2.3.1 Nitrate Contamination

There were a total of 3,831 ground water wells contributing 9,842 measurements of nitrate concentration from 1990 to 2000. On a yearly basis, the number of wells with nitrate concentration data ranged from 214 to 747 contributing 494 to 1,322 data points annually. Statistics of the nitrate data for the period of 1990 to 2000 are shown in Tables 2.1 through 2.3. Annual statistics for all the available nitrate data for 1990 to 2000 are

shown in Table 2.1. Statistics of the secondary data, such as maximum and median nitrate concentrations at each well in the study area, are shown in Tables 2.2 and 2.3, respectively.

To describe and assess probable anthropogenic effects on ground water quality, nitrate concentrations over WRIA 1 were classified into four groups, based on the work of Cox and Kahle (1999), Berndt and others (1998), and Madison and Brunett (1985). The four ground water nitrate concentration ranges used for the study area are:

- 0 – 1 mg/l : most likely natural background concentrations
- 1 – 3 mg/l : indicates possible human influence
- 3 – 10 mg/l : concentrations due to probable human influence
- ≥ 10 mg/l : MCL exceeded as a result of human activities

Baseline concentrations for [NO3+NO2] and the other nitrogen species for WRIA 1 are discussed in Section 2.4.

Trend of Total Nitrate Concentration Across WRIA 1

It should be noted that monitoring of ground water in WRIA 1 does not necessarily follow a well-developed monitoring plan. The selection of existing monitoring locations for previous studies is based on a specific purpose deemed important at the local level in prior years and may not be the optimal locations for a WRIA 1-wide study. Also these locations may have a bias on areas previously detected with high nitrogen concentrations. In such cases, some descriptive statistics such as percentage of wells exceeding MCL may be biased towards areas with frequent monitoring. Nevertheless, the statistics are important to obtain the overall trend of nitrogen contamination in WRIA 1.

All available [NO3+NO2] data for each year, from 1990 to 2000, were grouped into one of the four concentration ranges described above. The proportions of data in each of the concentration categories are shown in Table 2.1 and Figure 2.2. The proportion of data exceeding the MCL decreased from 1990 to 1993, but increased from 1993 to 1998. The results showed a decreasing trend again from 1998 to 2000, but the average proportion of data exceeding the MCL during 1998 to 2000 was greater than for the period 1990 to 1997. The results show that 11.7% of the data exceeded MCL in 1998 and this is the highest percentage from all years. The actual percentages and other statistics of all the [NO3+NO2] data are shown in Table 2.1. It can also be seen from Table 2.1 and Figure 2.2 that the trend of the percentage of data in the 0 to 1 mg/l range (which indicates little contamination) was opposite to that of data exceeding MCL.

Table 2.1. Summary statistics of all [NO3+NO2] data for WRIA 1 from 1990 to 2000.

Statistic	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	1990-2000
No. of data points	713	871	599	777	855	1068	812	1283	1322	1048	494	9842
No. of non-zero data	561	613	336	526	556	805	602	1065	973	805	397	7239
% of non-zero data	78.7	70.4	56.1	67.7	65.0	75.4	74.1	83.0	73.6	76.8	80.4	73.6
No. of data>=MCL	75	83	26	20	66	86	79	121	155	61	45	817
% of data>MCL	10.5	9.5	4.3	2.6	7.7	8.1	9.7	9.4	11.7	5.8	9.1	8.3
99th percentile (mg/l)	18.9	19.3	15.0	11.6	15.2	18.8	14.8	21.6	17.3	17.0	18.2	17.6
95th percentile (mg/l)	12	13	8.4	9.4	11.9	12	12.18	13.4	13.21	12.1	15.1	12.18
75th percentile (mg/l)	3.7	4	1.3	1.8	5.6	5.8	4.9	4.9	6.4	6.2	6.41	5.4
Mean concentration (mg/l)	2.9	3.1	1.5	1.7	3.2	3.5	2.9	3.4	4.0	4.0	4.5	3.2
Median concentration (mg/l)	0.7	0.3	0.2	0.3	0.8	1.6	1.0	1.3	2.0	3.5	4.2	1.0
Standard deviation	4.5	9.9	3.0	2.9	4.4	4.4	3.9	5.0	4.8	4.2	4.6	5.1
Min. concentration (mg/l)												
Min. non-zero conc. (mg/l)	0.100	0.010	0.100	0.100	0.077	0.050	0.060	0.010	0.002	0.032	0.200	0.002
Max. concentration (mg/l)	32	260	18.4	16	29	0	17	53	56	39	19.7	260
Range (mg/l)	0-32	0-260	0-18.4	0-16	0-29	0	0-17	0-53	0-56	0-39	0-19.7	0-260
No. in range 0-1 mg/l	380	489	361	507	430	444	394	557	538	384	165	4649
No. in range 1-3 mg/l	112	132	161	120	117	166	150	266	195	107	37	1563
No. in range 3-10 mg/l	146	167	51	130	242	372	189	339	433	496	247	2812
No. in range >=10 mg/l	75	83	26	20	66	86	79	121	155	61	45	817
% data in range < 1 mg/l	53.3	56.1	60.3	65.3	50.3	41.6	48.5	43.4	40.7	36.6	33.4	47.2
% data in range 1- 3 mg/l	15.7	15.2	26.9	15.4	13.7	15.5	18.5	20.7	14.8	10.2	7.5	15.9
% data in range 3- 10 mg/l	20.5	19.2	8.5	16.7	28.3	34.8	23.3	26.4	32.8	47.3	50.0	28.6
% data in range >=10 mg/l	10.5	9.5	4.3	2.6	7.7	8.1	9.7	9.4	11.7	5.8	9.1	8.3

Table 2.2. Summary statistics of maximum [NO3+NO2] concentration in WRIA 1 from 1990 to 2000.

Statistic	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	1990-2000
No. of wells	504	625	463	462	497	526	434	735	747	385	214	3831
No. of non-zero data wells	363	398	224	297	282	283	259	554	497	234	133	2098
% of non-zero data wells	72.0	63.7	48.4	64.3	56.7	53.8	59.7	75.4	66.5	60.8	62.1	54.8
No. of wells >=MCL	61	43	22	18	24	20	18	79	99	19	18	305
% of wells >MCL	12.1	6.9	4.8	3.9	4.8	3.8	4.1	10.7	13.3	4.9	8.4	8.0
99th percentile (mg/l)	21.0	20.0	15.4	11.8	25.0	19.7	15.3	24.6	20.8	17.3	19.7	21.6
95th percentile (mg/l)	14.0	13.0	9.9	9.6	9.9	8.7	8.1	15.0	14.9	9.8	12.3	12.7
75th percentile (mg/l)	4.5	3.6	1.5	2.4	2.4	1.9	1.8	4.3	5.5	3.5	4.5	3.0
Mean concentration (mg/l)	3.2	3.0	1.7	1.9	2.4	2.0	1.7	3.5	3.6	2.6	3.0	2.7
Median concentration (mg/l)	0.9	0.3	0.0	0.5	0.3	0.3	0.5	1.1	1.0	0.9	0.9	0.3
Standard deviation	4.8	11.3	3.2	3.1	4.5	3.9	3.1	5.8	5.4	4.2	4.5	6.4
Min. concentration (mg/l)	below detection limits (BDL)											
Min. non-zero conc. (mg/l)	0.10	0.01	0.10	0.10	0.08	0.05	0.06	0.01	0.04	0.20	0.20	0.01
Max. concentration (mg/l)	32.0	260.0	18.4	16.0	29.0	39.5	17.0	53.0	56.0	39.0	19.7	260.0
Range (mg/l)	0-32	0-260	0-18	0-18	0-29	0-40	0-17	0-53	0-56	0-39	0-20	0-260
No. of wells in range 0-1 mg/l	255	353	270	278	290	317	258	331	349	200	109	2221
No. of wells in range 1-3 mg/l	79	102	126	87	95	107	108	173	145	70	36	649
No. of wells in range 3-10 mg/l	109	127	45	79	88	82	50	152	154	96	51	656
No. of wells in range >=10 mg/l	61	43	22	18	24	20	18	79	99	19	18	305
% wells in range < 1 mg/l	50.6	56.5	58.3	60.2	58.4	60.3	59.4	45.0	46.7	51.9	50.9	58.0
% wells in range 1- 3 mg/l	15.7	16.3	27.2	18.8	19.1	20.3	24.9	23.5	19.4	18.2	16.8	16.9
% wells in range 3- 10 mg/l	21.6	20.3	9.7	17.1	17.7	15.6	11.5	20.7	20.6	24.9	23.8	17.1
% wells in range >=10 mg/l	12.1	6.9	4.8	3.9	4.8	3.8	4.1	10.7	13.3	4.9	8.4	8.0

Table 2.3. Summary statistics of median [NO3+NO2] concentration from all data in WRIA 1 for each year from 1990 to 2000.

Statistic	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	1990-2000
No. of wells	504	625	463	462	497	526	434	735	747	385	214	3831
No. of non-zero data wells	362	396	224	293	282	283	258	572	529	256	132	2125
% of non-zero data wells	71.8	63.4	48.4	63.4	56.7	53.8	59.4	77.8	70.8	66.5	61.7	55.5
No. of wells >=MCL	49	37	10	9	23	16	16	76	92	13	9	222
% of wells >MCL	9.7	5.9	2.2	1.9	4.6	3.0	3.7	10.3	12.3	3.4	4.2	5.8
99th percentile (mg/l)	20.0	20.0	11.2	11.4	19.0	12.0	12.1	24.6	19.6	15.4	15.1	18.9
95th percentile (mg/l)	12.0	11.4	7.2	7.8	9.4	8.6	7.8	14.8	14.1	9.4	8.8	10.9
75th percentile (mg/l)	3.4	3.2	1.2	1.2	2.0	1.7	1.4	3.8	5.2	2.2	4.4	2.2
Mean concentration (mg/l)	2.8	2.8	1.2	1.4	2.1	1.6	1.5	3.2	3.4	2.1	2.4	2.2
Median concentration (mg/l)	0.7	0.3	0.0	0.2	0.3	0.3	0.5	1.0	0.9	0.5	0.8	0.2
Standard deviation	4.6	11.2	2.5	2.5	4.0	3.2	2.6	5.7	5.2	3.8	3.4	4.7
Min. concentration (mg/l)						BDL						
Min. non-zero conc. (mg/l)	0.10	0.01	0.10	0.10	0.05	0.05	0.05	0.01	0.04	0.10	0.20	0.01
Max. concentration (mg/l)	32.0	260.0	16.0	12.0	26.2	33.9	12.1	53.0	56.0	39.0	15.1	130.0
Range (mg/l)	0-32	0-260	0-16	0-12	0-26.2	0-34	0-12	0-53	0-56	0-39	0-15	0-130
No. of wells in range 0-1 mg/l	284	366	294	314	309	329	285	377	415	258	115	2408
No. of wells in range 1-3 mg/l	82	100	118	87	88	109	89	158	126	65	36	625
No. of wells in range 3-10 mg/l	89	122	41	52	77	72	44	142	149	75	54	614
No. of wells in range >=10 mg/l	49	37	10	9	23	16	16	76	92	13	9	222
% wells in range < 1 mg/l	56.3	58.6	63.5	68.0	62.2	62.5	65.7	51.3	55.6	67.0	53.7	62.9
% wells in range 1- 3 mg/l	16.3	16.0	25.5	18.8	17.7	20.7	20.5	21.5	16.9	16.9	16.8	16.3
% wells in range 3- 10 mg/l	17.7	19.5	8.9	11.3	15.5	13.7	10.1	19.3	19.9	19.5	25.2	16.0
% wells in range >=10 mg/l	9.7	5.9	2.2	1.9	4.6	3.0	3.7	10.3	12.3	3.4	4.2	5.8

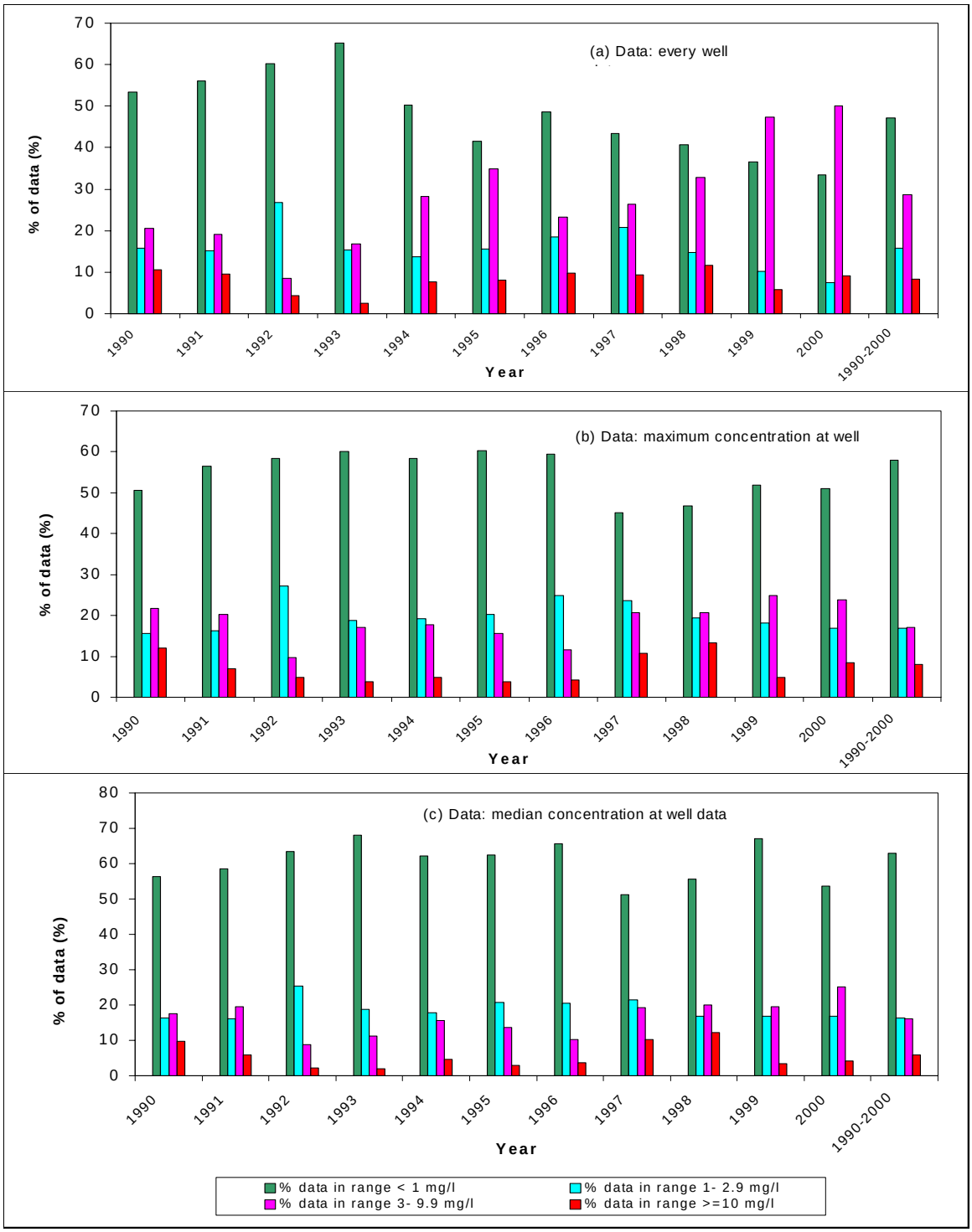


Figure 2.2. Historical trends of percentage of [NO3+NO2] data in various concentration ranges using (a) all data, (b) maximum, and (c) median concentration.

Since the wells in the study area did not have equal numbers of [NO3+NO2] water quality samples each year, the analysis was repeated using the percentages of wells instead of percentages of all data. The annual maximum and annual median [NO3+NO2] concentrations at each well were used in the analysis. Charts of the relative proportions of wells with annual maximum and annual median concentrations in the four concentration ranges are shown in Figure 2.2 and the actual percentages are shown in Tables 2.2 and 2.3. The trends for the maximum and median concentrations were similar to those for all the data. The highest percentage in the category "greater than MCL" was still in 1998.

It can also be seen from Figure 2.2 that the proportions of data as well as the proportions of annual maximum and annual median concentrations in the range 3-10 mg/l have increased steadily from 1992 to 2000, with slight decreases between 1994 and 1996. This range, though below the MCL, indicates the influence of anthropogenic effects of ground water quality in the area. Therefore, the increasing pattern of [NO3+NO2] concentration in this range is of concern.

The percentages of all data, and the maximum and median [NO3+NO2] concentrations in the range 0-1 mg/l (which indicates the least [NO3+NO2] contamination) increased from 1990 to 1993, and decreased from 1993-1994. While the percentages of all data in this range decreased from 1996 to 2000, the percentages for the maximum concentrations increased during the same period. The percentages for the median well concentrations data also showed a general increasing trend for this period.

The mean, median, 75th percentile and 95th percentile concentrations of all [NO3+NO2] data decreased from 1990 to 1992, but increased from 1992 to 1998 and decreased again from 1998 to 2000 (see Figure 2.3). The 95th percentile concentration was higher than the MCL in all years except for 1992 and 1993. The 75th percentile was always below the MCL, indicating that at least 75 percent of the [NO3+NO2] concentrations was below the MCL from 1990 to 2000. The corresponding plots for the maximum and median wells data were similar, but fewer wells had the 95th percentile below the MCL (see Figure 2.3 (b) and (c)). In these cases too, the 75th percentile was always below the MCL, indicating that at least 75 percent of the wells in WRIA 1 had maximum and median [NO3+NO2] concentrations below the MCL. However, it is obvious that there was a general trend of increasing [NO3+NO2] concentrations starting from 1992 or 1993.

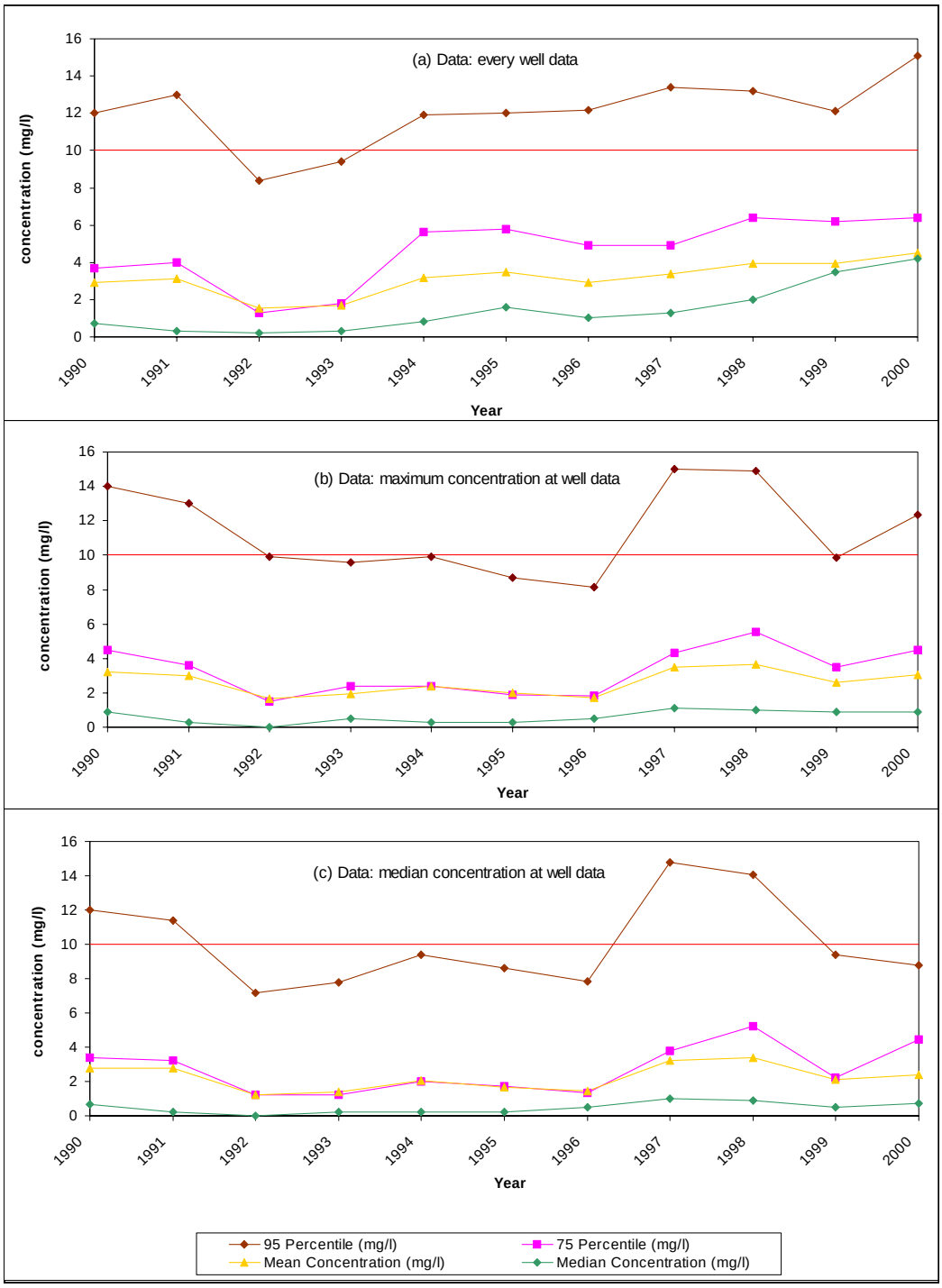


Figure 2.3. Historical trends of mean, median, 75th percentile, and 95th percentile [NO3+NO2] concentrations, using (a) all data, (b) maximum concentration, and (c) median concentration data. The red horizontal line indicates the MCL.

Trends in Individual Sub-Basins

The annual mean, median and maximum [NO3+NO2] concentrations in the various sub-basins in WRIA 1 from 1990 to 2000 are summarized in Tables 2.4 through 2.6. Plots of the mean and median concentrations are shown in Figures 2.4 and 2.5. It can be seen from the plots that none of the sub-basins had mean or median [NO3+NO2] concentrations above the MCL during the period. A few of the sub-basins though, stand out because of their higher average mean and median concentrations. The sub-basins with the higher mean and median [NO3+NO2] concentrations are:

- Bertrand/Fishtrap/Kamm
- Silver/Nooksack Channel and Delta
- Barrett Lake
- Scott/Wiser/Schneider
- Sumas River

There have been fluctuations in the levels of [NO3+NO2] concentrations in the sub-basins, but in general, the trend has been an increasing pattern in most of the sub-basins. Figure 2.6 shows a classification of sub-basins of WRIA 1 using the maximum [NO3+NO2] concentration level for the period 1990 to 2000. The yearly maximum [NO3+NO2] concentration classifications are shown in Appendix A.

It should be noted that if at least one observation during a given year falls within a given concentration group, then the sub-basin corresponding to that location is defined as represented by that concentration group. It is true that this definition is rather conservative and too spatially extensive, but provides a mechanism to define regional-scale ground water quality in the absence of detailed field data.

The yearly classifications by median concentrations are shown in Appendix B. Most of the sub-basins had median nitrate concentrations less than MCL in most years, except for the Bertrand/Fishtrap/Kamm sub-basin where even the annual median nitrate concentration exceeded the MCL in 1995, 1996, and 1999.

Using maximum nitrate concentration levels, the sub-basins in WRIA 1 can be classified by two different methods:

Classification A - Using the maximum [NO3+NO2] concentration from each well between 1990 and 2000 across a given sub-basin, the group assignment given below was carried out for each sub-basin. The four concentration groups are as follows:

Group 1	-	$\geq$ 10 mg/l
Group 2	-	3 – 10 mg/l
Group 3	-	1 – 3 mg/l
Group 4	-	< 1 mg/l

Classification B - The maximum annual [NO3+NO2] concentration of each sub-basin in each year between 1990 and 2000 was first computed. Second, these values were then assigned to different groups described in Classification A. This assignment provided the distribution of annual maximum nitrate concentration to different groups such that their occurrence is known.

The results of the grouping of the sub-basins using these two classifications are shown in Table 2.7. The number of years in the period 1990 to 2000 that each sub-basin had maximum concentrations in the various groups is plotted in Figure 2.7. Two sub-basins, Barrett Lake and Bertrand/Fishtrap/Kamm, had wells with maximum nitrate concentration exceeding the MCL in every one of the 11 years in the period. They were followed closely by Scott/Wiser/Schneider sub-basin, exceeding MCL in 10 out of the 11 years. These three sub-basins fall in Group 1 by both methods of classification. There were no data available for the Fraser River sub-basin. Classification A is useful in identifying those sub-basins, which have experienced [NO3+NO2] concentration levels in excess of the MCL. These sub-basins, falling into Group 1 of Classification A (as shown in Figure 2.6 and Table 2.7, are:

- Terrell/Semiahmoo/Cherry Point
- Lummi/Eliza Island
- Lummi Bay
- Silver/Nooksack Channel & Delta
- Anderson/Smith/Deming to Everson
- Dakota/California/Blaine
- Barrett Lake
- Sumas River
- Scott/Wiser/Schneider
- Bertrand/Fishtrap/Kamm

Table 2.4. Annual mean [NO3+NO2] concentration (mg/L) of each sub-basin in WRIA 1 from 1990 to 2000.

Sub-Basin	**1990**	**1991**	**1992**	**1993**	**1994**	**1995**	**1996**	**1997**	**1998**	**1999**	**2000**	**1990-2000**
Skagit County	0.53	0.20	0.10	1.64	0.37	0.50	0.10	0.00	0.55	0	0	0.63
Lummi/Eliza Island	0.34		0.10	0.59	0.18	0.68	1.81		1.00	0.86	1.00	0.76
Lummi Peninsula/Portage Island	0.20	4.26	0.00	0.45	0.13	0.97	0.18	1.00	0.25	1.57	-	0.70
Lummi Bay	0.39	0.76	0.65	0.41	0.89		0.84	0.59	1.12	0.74	0.94	0.75
South Fork Nooksack	1.10	0	0	0.83	0.50	0.72	0.57	0.45	0.90	0.90	-	0.55
Silver/Nooksack Channel & Delta	5.29	3.44	1.03	1.31	1.88	2.01	1.16	2.57	1.73	2.40	0.17	2.21
Fort Bellingham/Whatcom/Padden /Chuckanut/Fragrance	1.90	0.04	0.59	1.38	0.07	0.65	0	0.11	0.58	0.11	0.50	0.50
Squalicum	0.06	0.88	0.20	0.12	0.10		0.49	0.14	0.27	0.31	0	0.29
Anderson/Smith/Deming to Everson	1.41	0.96	1.31	0.62	1.53	0.83	1.16	1.68	0.69	0.66	0.67	1.10
	2.15	4.48	1.67	1.60	1.54	3.23	1.62	3.26	3.34	3.53	4.15	2.91
Terrell/Semiahmoo/Cherry Point	0.83	0.84	0.59	0.10	0.42	3.85	0.47	1.03	0.35	0.76	0.95	0.99
Sumas River	2.43	1.88	1.78	2.85	5.04	4.79		3.89	5.64	5.89	5.26	4.62
Scott/Wiser/Schneider	4.04	4.63	2.53	1.96	3.96	3.63	3.15	4.57	3.14	3.81	3.18	3.64
North Fork Nooksack	0.76	0.44	0.41	0.64	0.54	0.49	0.56	0.61	0.73	0.74	0.52	0.60
Dakota/California/Blaine	0.70	2.11	1.63	0.91	1.56	0.64	0.79	1.81	3.14	0.93	1.96	1.59
Bertrand/Fishtrap/Kamm	5.64	6.00	6.02	4.77	7.98	9.85	9.66	8.40	8.02	10.00	9.02	7.75
Point Roberts		0.15		0		0	0.90	1.61				0.77
Lake Whatcom	0.55	0.26	0.32	0.54	0.80	0.40	0.72	0.55	0.12	0.54	0.67	0.50
Middle Fork Nooksack	0.80	0.32	1.40	0.40	0.61	0	0.98		0.78	BDL	0.40	0.61
Little Campbell						0.60						0.60
Fraser River												

Table 2.5. Annual median [NO3+NO2] concentration (mg/L) of each sub-basin in WRIA 1 from 1990 to 2000.

Sub-Basin	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	1990-2000
Skagit County	0.20	0.20	0.10	0.26	0.10	0.50	0.10	0	0	0	0	0.100
Lummi/Eliza Island	0.40	0.00	0.00	0.00	0.00	0.55	0.55	1.00	0.50	0.40	0	0.100
Lummi Peninsula/Portage Island	0.20	4.26	0.00	0.45	0.00	0.08	0.00	1.00	0.25	2.04		0.000
Lummi Bay	0.20	0.20	0.20	0.11	0.20	0.20	0.50	0.50	0.50	0.50	0.00	0.200
South Fork Nooksack	1.10	0.00	0.00	0.80	0.50	0.65	0.58	0.40	1.20	0.30		0.000
Silver/Nooksack Channel & Delta	3.20	3.17	0.00	0.70	1.50	1.50	0.20	3.40	0.69	0.56	0.00	0.700
Fort Bellingham/Whatcom/Padden /Chuckanut/Fragrance	2.30	0.00	0.00	1.20	0.00	0.00	0.00	0.00	0.56	0.00	0.50	0.000
Squalicum	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.000
Anderson/Smith/Deming to Everson	0.30	0.20	0.00	0.20	0.00	0.60	1.00	0.92	0.20	0.00	0.30	0.200
Barrett Lake	0.20	1.10	0.60	0.30	0.30	1.00	0.50	1.53	2.80	2.70	4.20	1.000
Terrell/Semiahmoo/Cherry Point	0.40	0.10	0.20	0.00	0.00	1.00	0.00	0.20	0.00	0.50	0.00	0.100
Sumas River	1.70	0.01	1.32	0.30	5.40	5.10	5.00	3.80	5.70	5.80	5.60	5.000
Scott/Wiser/Schneider	2.20	3.00	1.01	0.60	3.40	1.00	2.00	3.55	1.76	3.80	2.12	1.960
North Fork Nooksack	0.20	0.20	0.30	0.35	0.25	0.50	0.50	0.69	0.70	0.72	0.60	0.500
Dakota/California/Blaine	0.20	0.20	1.00	0.20	0.00	0.20	0.50	0.39	0.59	0.00	0.74	0.200
Bertrand/Fishtrap/Kamm	4.40	5.30	6.90	3.20	9.30	10.90	11.60	9.06	8.77	10.70	7.10	7.900
Point Roberts		0.15		0.00		0.00	0.90	1.90				0.200
Lake Whatcom	0.20	0.00	0.00	0.10	0.10	0.00	0.00	0.50	0.00	0.00	0.00	0.000
Middle Fork Nooksack	0.80	0.30	0.65	0.40	0.55	0.00	1.20		0.80	0.00	0.50	0.515
Little Campbell						0.60						0.600
Fraser River												

Table 2.6. Annual maximum [NO3+NO2] concentration (mg/L) of each sub-basin in WRIA 1 from 1990 to 2000.

Sub-Basin	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	1990-2000
Skagit County	1.20	0.20	0.10	8.10	1.00	0.50	0.10	0.00	2.10	0.00	0.00	8.10
Lummi/Eliza Island	0.50	2.54	1.07	5.90	2.40	3.20	14.30	8.00	3.90	3.30	5.00	14.30
Lummi Peninsula/Portage Island	0.20	4.26	0.00	0.60	0.50	5.20	0.50	1.00	0.50	2.17		5.20
Lummi Bay	2.20	15.18	8.40	6.10	7.50	6.70	6.10	3.87	3.70	7.50	8.50	15.18
South Fork Nooksack	1.30	0.00	0.00	2.40	1.50	1.60	1.35	1.00	1.49	3.70		3.70
Silver/Nooksack Channel & Delta	15.00	14.00	9.94	7.40	6.20	7.90	5.30	6.30	7.30	9.80	0.50	15.00
Fort Bellingham/Whatcom/Padden /Chuckanut/Fragrance	3.00	0.60	1.60	3.10	0.30	5.20	0.00	1.37	1.20	0.54	0.50	5.20
Squalicum	1.00	8.77	2.23	1.40	1.40	4.20	7.90	1.80	2.00	3.00	0.00	8.77
Anderson/Smith/Deming to Everson	15.00	15.03	18.40	2.80	8.80	5.00	3.20	17.50	4.10	3.00	3.30	18.40
Barrett Lake	18.00	260.00	14.90	11.40	12.70	13.00	10.00	53.00	15.00	17.50	10.84	260.00
Terrell/Semiahmoo/Cherry Point	2.30	4.80	2.15	1.00	2.80	24.70	3.80	6.97	1.90	7.50	14.93	24.70
Sumas River	15.00	16.00	8.10	15.40	26.20	8.10	8.10	42.00	25.60	12.00	12.33	42.00
Scott/Wiser/Schneider	22.40	19.00	17.20	16.00	10.00	39.50	17.00	32.30	16.60	12.15	8.50	39.50
North Fork Nooksack	3.50	3.40	1.60	2.70	1.90	1.80	2.40	1.50	2.60	2.70	1.00	3.50
Dakota/California/Blaine	6.70	34.90	16.00	7.80	29.00	3.90	4.90	17.80	56.00	9.85	9.60	56.00
Bertrand/Fishtrap/Kamm	32.00	43.00	16.02	11.80	25.00	19.70	15.30	34.20	27.50	39.00	19.70	43.00
Point Roberts		0.20		0.00		0.00	1.80	1.92				1.92
Lake Whatcom	2.20	2.50	1.50	1.80	3.90	2.00	2.70	1.60	1.00	4.90	3.46	4.90
Middle Fork Nooksack	0.80	0.80	4.30	0.80	1.80	0.00	1.20		2.00	0.00	0.70	4.30
Little Campbell						0.60						0.60
Fraser River												

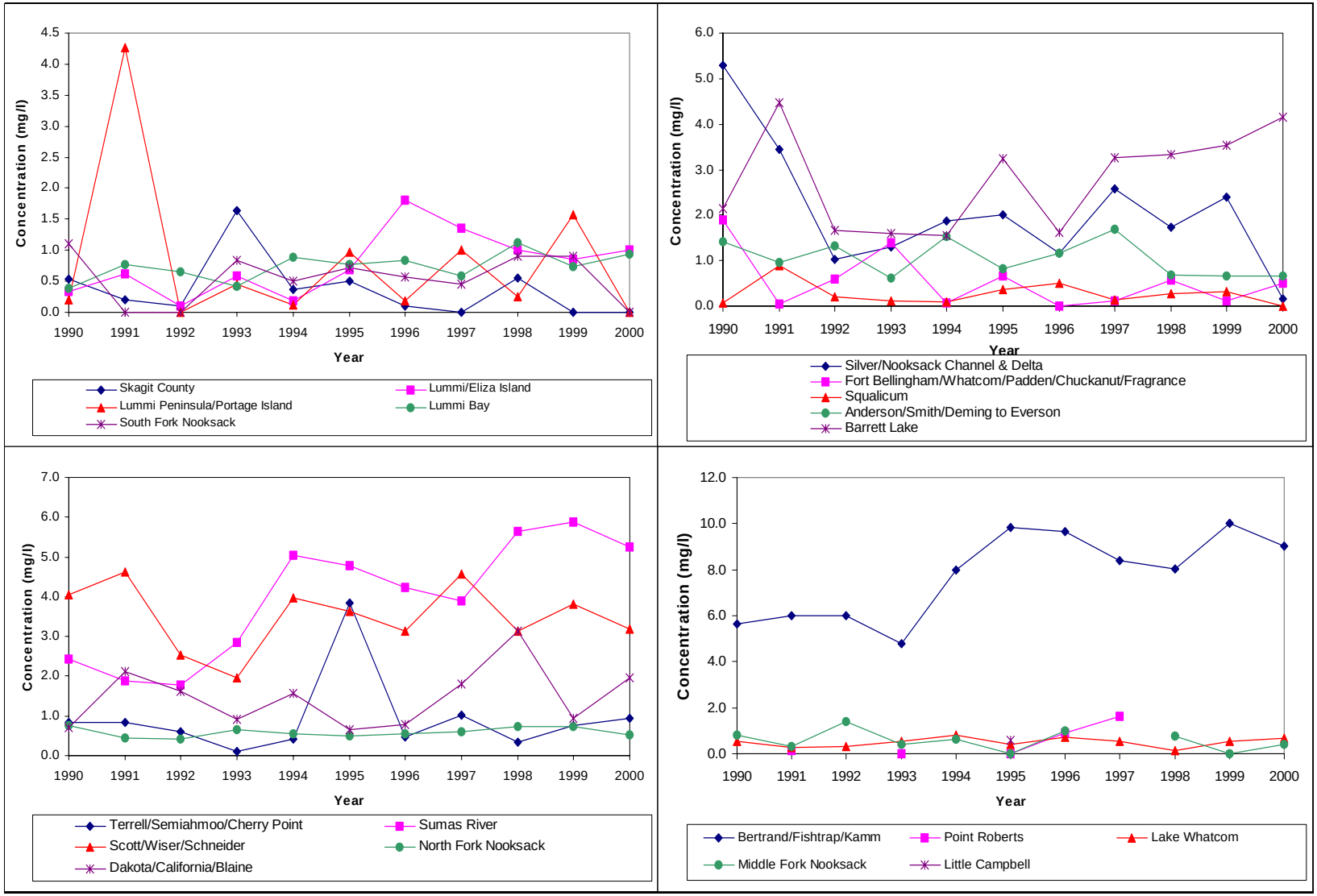


Figure 2.4. Time series of annual mean [NO3+NO2] concentration in different sub-basins.

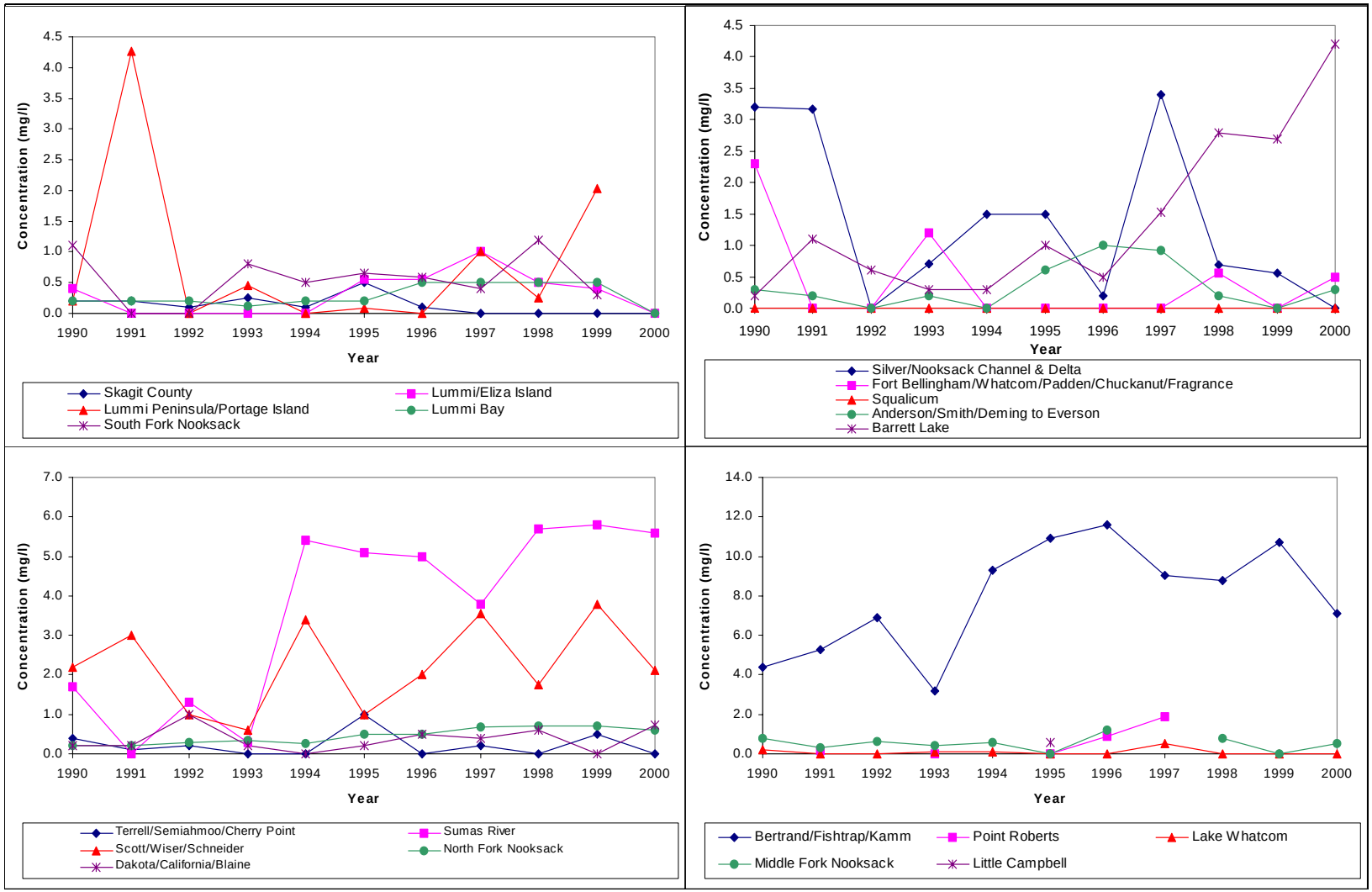
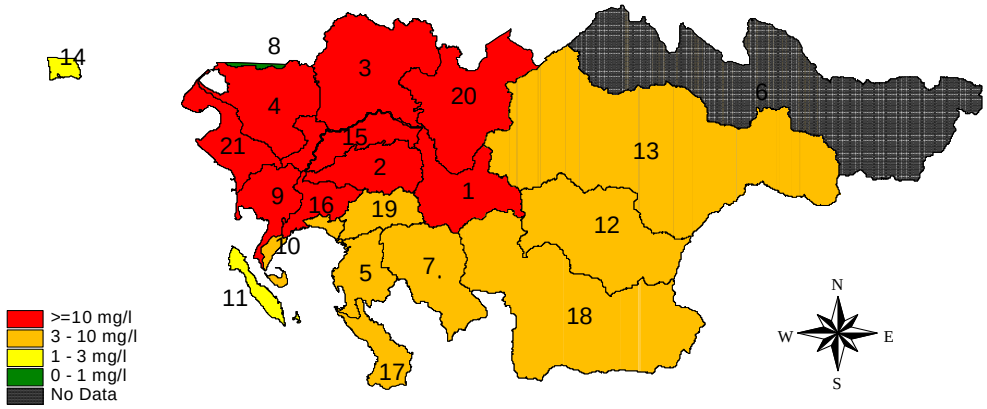


Figure 2.5. Time series of annual median [NO3+NO2] concentration in different sub-basins.



1- Upper Mainstem Nooksack
2- Barrett Lake
3- Bertrand/Fishtrap/Kamm
4- Dakota/California/Blaine
5- Fort Bellingham/Whatcom/Padden/Chuckanut/Fragrance
6- Fraser River
7- Lake Whatcom
8- Little Campbell
9- Lummi Bay
10- Lummi Peninsula/Portage Island
11- Lummi/Eliza Island
12- Middle Fork Nooksack
13- North Fork Nooksack
14 -Point Roberts
15 -Scott/Wiser/Schneider
16 -Silver/Nooksack Channel & Delta
17- Skagit County
18- South Fork Nooksack
19- Squalicum
20- Sumas River
21- Terrell/Semiahmoo/Cherry Point

Figure 2.6. Distribution of maximum nitrate concentration levels in different sub-basins from 1990 to 2000.

Table 2.7. Classification of sub-basins in WRIA 1 by the maximum [NO3+NO2] concentration. Classification A is based on the maximum concentration between 1990 and 2000; Classification B is based on the number of years that the annual nitrate maximum concentration was in range during 1990 to 2000.

			Number of Years in Each Concentration Range				
	Nitrate Level Group		≥10	3-10	1-3	0-1	
Sub-Basin	**Class.-A**	**Class.-B**	**mg/l**	**mg/l**	**mg/l**	**mg/l**	**No data**
Terrell/Semiahmoo/Cherry Point	1	3	2	4	5	0	0
Lummi/Eliza Island	1	2	1	6	3	1	0
Lummi Bay	1	2	1	9	1	0	0
Silver/Nooksack Channel & Delta	1	2	2	8	0	1	0
Anderson/Smith/Deming to Everson	1	2	4	6	1	0	0
Dakota/California/Blaine	1	2	5	6	0	0	0
Barrett Lake	1	1	11	0	0	0	0
Sumas River	1	1	8	3	0	0	0
Scott/Wiser/Schneider	1	1	10	1	0	0	0
Bertrand/Fishtrap/Kamm	1	1	11	0	0	0	0
Skagit County	2	4	0	1	3	7	0
Lummi Peninsula/Portage Island	2	4	0	2	2	6	1
Fort Bellingham/Whatcom/Padden/ Chuckanut/Fragrance	2	4	0	3	3	5	0
Middle Fork Nooksack	2	4	0	1	3	6	1
South Fork Nooksack	2	3	0	1	7	2	1
Squalicum	2	3	0	4	6	1	0
North Fork Nooksack	2	3	0	2	9	0	0
Lake Whatcom	2	3	0	3	8	0	0
Point Roberts	3	1	0	0	2	3	6
Little Campbell	4	1	0	0	0	1	10
Fraser River	N/A	N/A					11

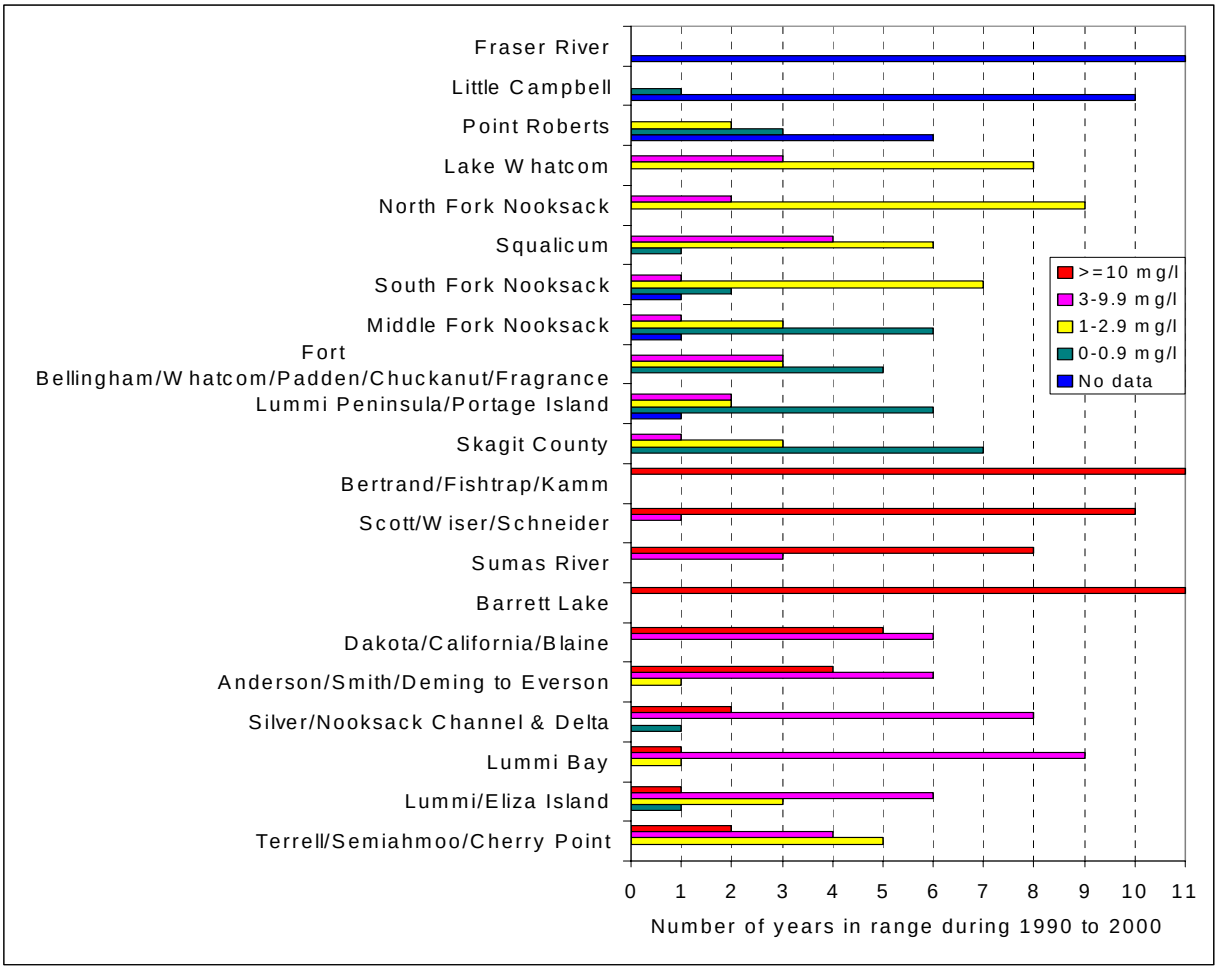


Figure 2.7. Sub-basins and number of years that the annual maximum [NO3+NO2] concentration was in the given range during 1990 to 2000.

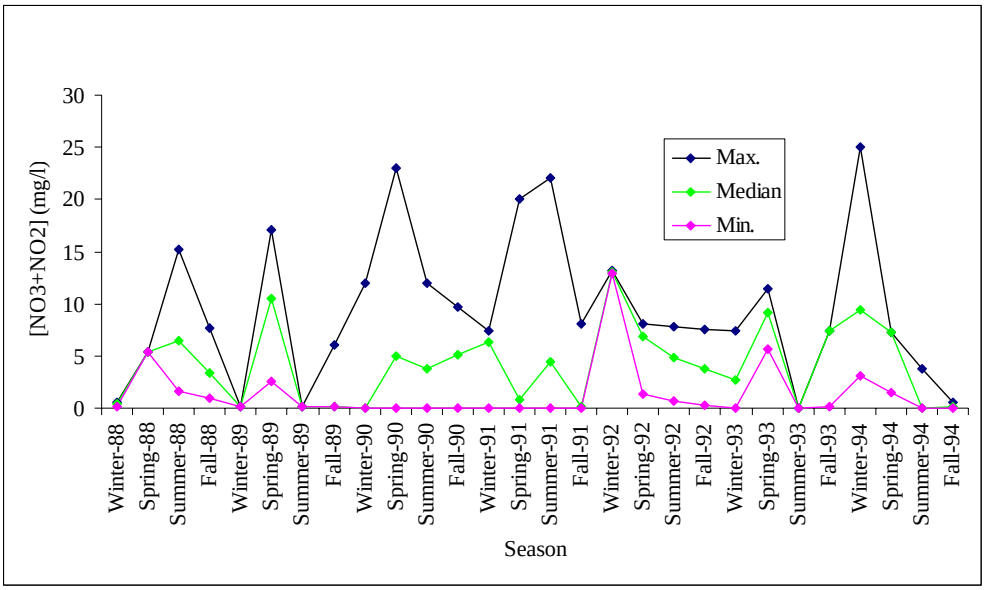
Table 2.8. Number and percentage of wells with all [NO3+NO2] concentration data in various ranges in sub-basins of WRIA 1 for the period 1990-2000.

Sub-Basin	Total No. of Wells	Number of Wells >=10 mg/l	3-10 mg/l	1-3 mg/l	0-1 mg/l	Percentage of Wells ≥10 mg/l	3-10 mg/l	1-3 mg/l	0-1 mg/l
Skagit County	3	0	1	1	1	0.0	33.3	33.3	33.3
Lummi/Eliza Island	84	1	8	5	70	1.2	9.5	6.0	83.3
Lummi Peninsula/Portage Island	31	0	4	5	22	0.0	12.9	16.1	71.0
Lummi Bay	183	1	27	18	137	0.5	14.8	9.8	74.9
South Fork Nooksack	62	0	1	17	44	0.0	1.6	27.4	71.0
Silver/Nooksack Channel & Delta	119	3	35	24	57	2.5	29.4	20.2	47.9
Fort Bellingham/Whatcom/ Padden/Chuckanut/Fragrance	79	0	6	10	63	0.0	7.6	12.7	79.7
Squalicum	227	0	7	11	209	0.0	3.1	4.8	92.1
Anderson/Smith/Deming to Everson	274	6	23	52	193	2.2	8.4	19.0	70.4
Barrett Lake	501	37	86	91	287	7.4	17.2	18.2	57.3
Terrell/Semiahmoo/Cherry Point	130	2	13	10	105	1.5	10.0	7.7	80.8
Sumas River	248	33	46	59	110	13.3	18.5	23.8	44.4
Scott/Wiser/Schneider	386	35	146	70	135	9.1	37.8	18.1	35.0
North Fork Nooksack	185	0	4	48	133	0.0	2.2	25.9	71.9
Dakota/California/Blaine	555	20	81	104	350	3.6	14.6	18.7	63.1
Bertrand/Fishtrap/Kamm	467	167	143	40	117	35.8	30.6	8.6	25.1
Point Roberts	10	0	0	10	0	0.0	0.0	100.0	0.0
Lake Whatcom	138	0	4	32	102	0.0	2.9	23.2	73.9
Middle Fork Nooksack	39	0	1	7	31	0.0	2.6	17.9	79.5
Little Campbell	1	0	0	0	1	0.0	0.0	0.0	100.0
Fraser River	0	0	0	0	0	0.0	0.0	0.0	0.0

Seasonal Variability of Nitrate

Since bulk of the nitrogen contamination occurs due to agricultural activities, it is of interest to assess the impact of ground water quality in aquifers due to seasonal changes. However due to lack of sufficient data across entire WRIA 1, seasonal variability analysis was restricted to two watersheds with most data and high degree of nitrogen contamination and these are Fishtrap and Bertrand Creek watersheds.

Figure 2.8 shows the seasonal variability of nitrate concentration across Bertrand Creek watershed from 1988 to 2000 giving the minimum, maximum and median. Due to the scatter of data, median concentration provides more meaningful insight to the variability than the mean. In addition, Table 2.9 shows the occurrence of highest to lowest nitrate concentrations by the season per each year. It should be noted that for years where there are no ranking in Table 2.9 refers to years where there were no data. The results plotted in Figure 2.8 for those years are computed using the averages obtained from the year before and after that particular year. The results show that the maximum nitrate concentration consistently remained much higher than the MCL of 10 mg/L during the entire period and was as close to 30 mg/l in most years. In addition, the maximum nitrate concentration occurs typically close to the spring and the minimum nitrate concentration occurs close to the summer.



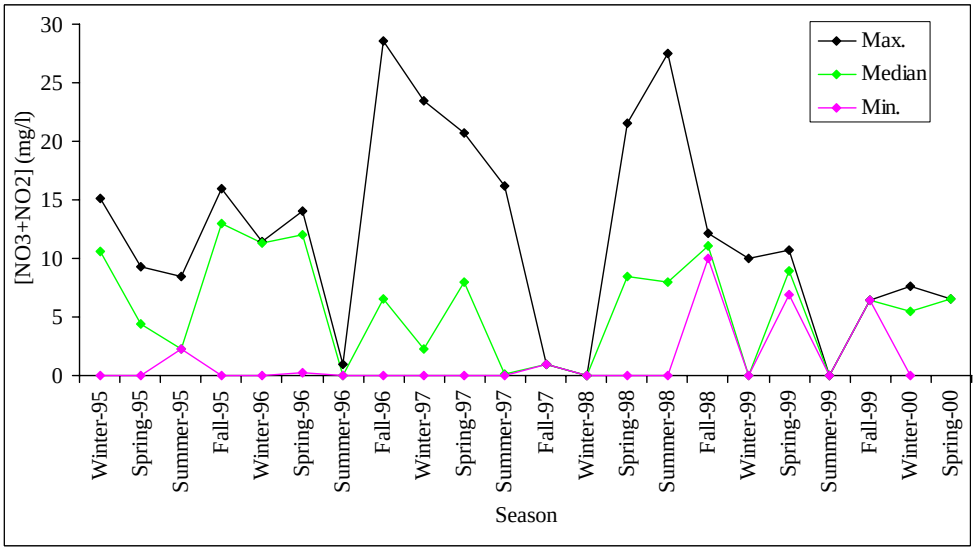


Figure 2.8. Seasonal variability of nitrate concentration in the Bertrand Creek watershed.

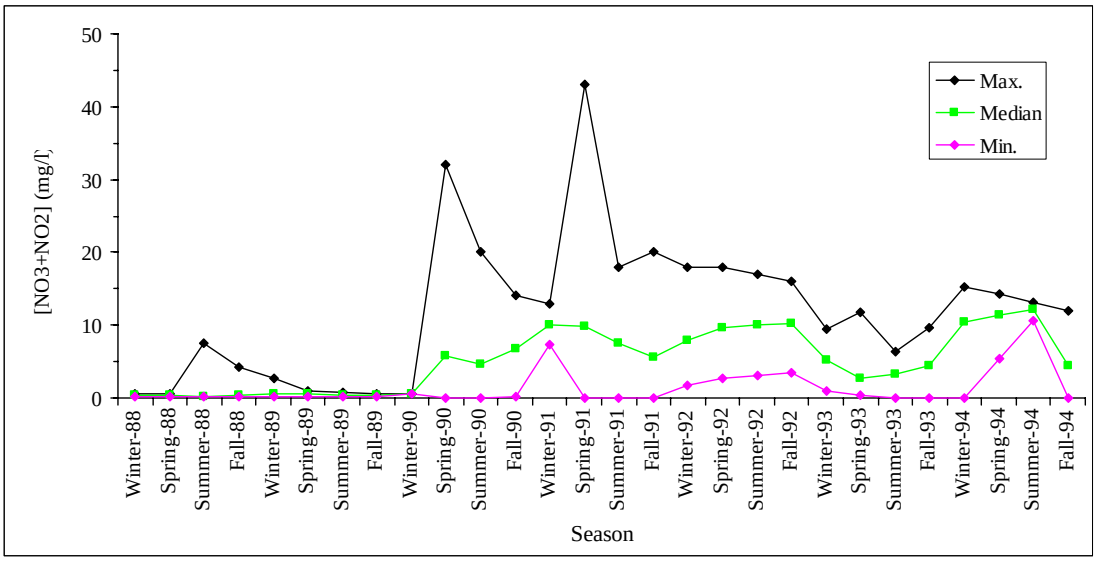
Table 2.9. Occurrence of highest to lowest nitrate concentration in Bertrand Creek watershed by season for each year. Highest concentration refers to 1 and lowest to 4.

Year	1	2	3	4
1988	-	-	-	-
1989	-	-	-	-
1990	-	-	-	-
1991	Summer	Spring	Fall	Winter
1992	-	-	-	-
1993	Spring	Fall+Winter	Fall+Winter	Summer
1994	Winter	Spring	Summer	Fall
1995	Fall	Winter	Spring	Summer
1996	Fall	Spring	Winter	Summer
1997	Winter	Spring	Summer	Fall
1998	Summer	Spring	Fall	Winter
1999	Spring	Winter	Fall	Summer

In addition, the seasonal Kendall test was conducted using the available data (Gibbons, 1994) where the Mann-Kendall statistics were computed. The normal approximation computed for Bertrand Creek watershed was 3.54 at a probability of 0.0002 indicating the rejection of null-hypothesis of no seasonal trend. Hence, the data shows that there is a seasonal trend for nitrate concentration in the Bertrand Creek watershed.

A similar analysis was conducted for Fishtrap Creek watershed using data from 1988 to 2000. The results are shown in Figure 2.9 and Table 2.11 for seasonal variability and also for occurrence of lowest to highest nitrate concentration. The results clearly show that maximum nitrate concentration in the Fishtrap Creek watershed remained high and sometimes more than 30 mg/L. The worst nitrate concentration was observed in 1990-1991 where the maximum concentration reached more than 40 mg/L. In addition, the highest occurrence of nitrate was in the spring season as in the case with Bertrand Creek watershed. The normal approximation of the Mann-Kendall statistics was 2.84 at a probability of 0.002, again indicating a seasonal trend to nitrate concentration data.

In addition to the seasonal variability, nitrate distribution by vertical depth can provide the influence of geological stratification on vertical migration of contaminants. Unfortunately, there were no sufficient data to make any useful observations or conclusions related to vertical distribution of nitrate. In general, the limited data between close-by monitoring wells showed that the nitrate concentration reducing substantially after the initial 40 to 50 feet depth indicating that contamination is somewhat restricted to shallow ground water.



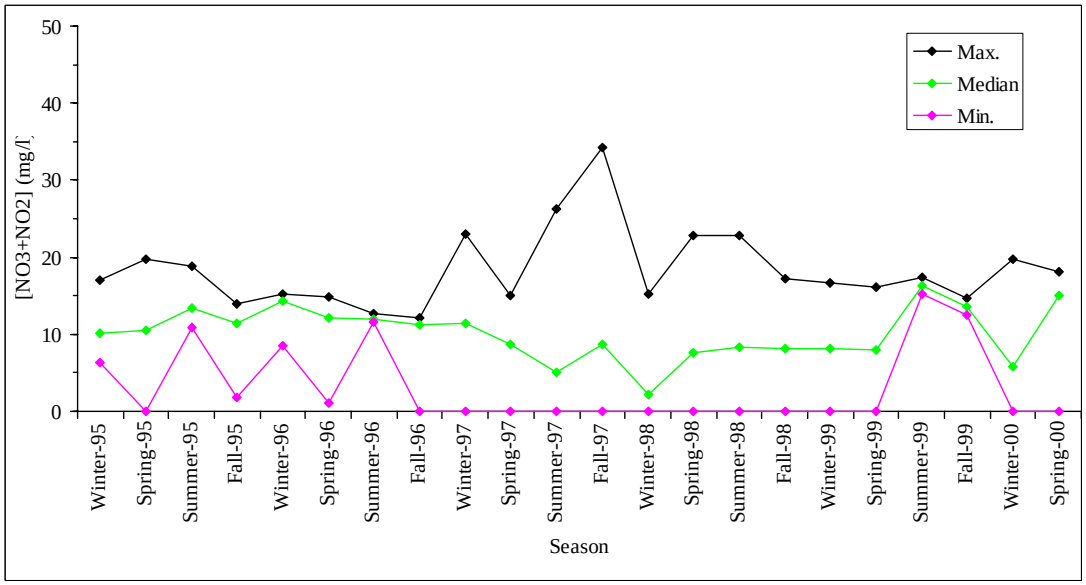


Figure 2.9. Seasonal variability of nitrate concentration in the Fishtrap Creek watershed.

Table 2.10. Occurrence of highest to lowest nitrate concentration in Fishtrap Creek watershed by season for each year. Highest concentration refers to 1 and lowest to 4.

Year	1	2	3	4
1988	Summer	Fall	Winter	Spring
1989	Winter	Spring	Summer	Fall
1990	Spring	Summer	Fall	Winter
1991	Spring	Summer	Fall	Winter
1992	-	-	-	-
1993	Spring	Fall	Winter	Summer
1994	Winter	Spring	Summer	Fall
1995	Spring	Summer	Winter	Fall
1996	Winter	Spring	Summer	Fall
1997	Fall	Summer	Winter	Spring
1998	Spring	Summer	Fall	Winter
1999	Summer	Winter	Spring	Fall

Trends of Nitrate in Individual Wells

A plot of the spatial variation of the maximum [NO3+NO2] concentrations at individual ground water wells in WRIA 1 for the period 1990 to 2000 is shown in Figure 2.10. It is clear from the figure that the maximum concentration of nitrates over WRIA 1 is not uniform. The areas with the highest levels of maximum nitrate concentration are around Everson and Ferndale and above the curve formed by connecting the cities Sumas-Nooksack-Everson-Ferndale-Blaine and around the Canadian border in this area. The areas exceeding the MCL of 10 mg/l are almost, without exception, over the main surficial aquifer, i.e., Sumas-Blaine surficial aquifer, also known as the Sumas-Blaine aquifer.

Plots of the annual maximum [NO3+NO2] concentrations from 1990 to 2000 are shown in Appendix C. Plots of the annual mean [NO3+NO2] concentrations are shown in Appendix D. The number of wells sampled is not the same each year. These plots show that the density of the wells exceeding the MCL each year does not appear to be even, but it is seen from these plots that the density of highly contaminated wells in each year is centered about Lynden and the Sumas-Blaine surficial aquifer around the Canadian border. Most of the highly polluted wells are in the Bertrand/Fishtrap/Kamm and the Scott/Wiser/Schneider sub-basins. The years 1997 and

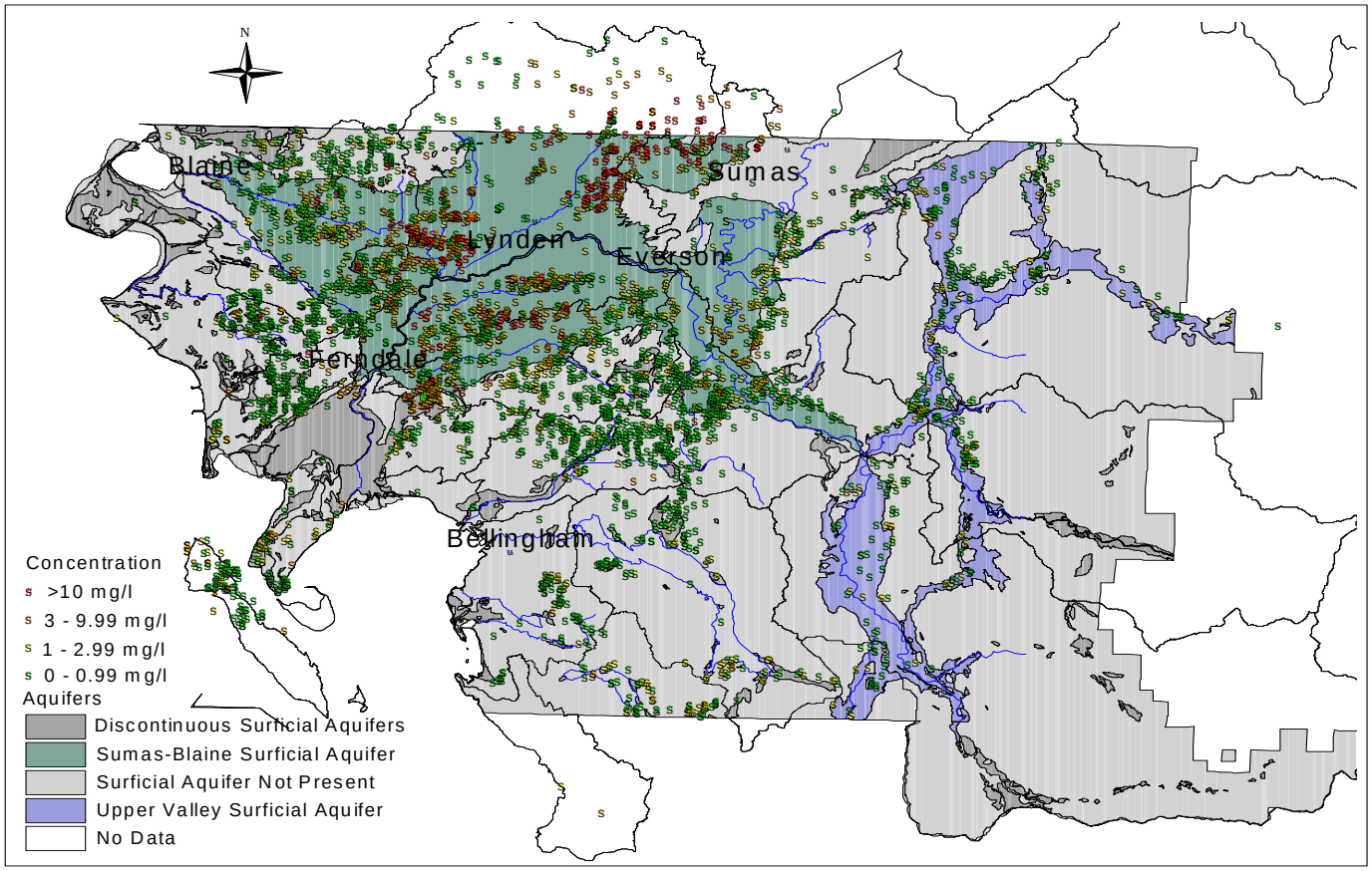


Figure 2.10. Areal distribution of maximum [NO3+NO2] concentration in WRIA 1 from 1990 to 2000.

1998 appear to have the highest density of wells with [NO3+NO2] concentrations exceeding MCL. However, it should be noted that these two years also had the highest number of ground water wells with [NO3+NO2] data (see Tables 2.2 and 2.3). The wells with the lowest [NO3+NO2] concentrations are mostly over the areas where non-surficial aquifers are present. Many of the wells over the Upper Valley surficial aquifer in the eastern part of WRIA 1 also had low maximum [NO3+NO2] concentrations.

Plots of the mean [NO3+NO2] concentration in Appendix D appear to follow the same general spatial trend as that for the maximum concentrations. Time series plots of the wells in each sub-basin with the highest number of time series data are shown in Appendix G. The time series mostly exhibit substantial temporal variation in [NO3+NO2] concentrations. However, since the data collection frequency was not regular, a more rigorous time series analysis could not be performed at this time. Even though the nature of the fluctuations could not be quantified precisely, it is clear from the qualitative behavior of these plots that in general, most of these wells started recording nitrate levels higher than the MCL around 1992.

Summary of Nitrate-Related Contamination

A composite ground water quality database for WRIA 1 was compiled with historical data from four main agencies, i.e., USGS, Whatcom County Department of Health, Washington Department of Health, and Washington Department of Ecology. The total number of ground water wells in the database was 4,247. The number of wells in WRIA 1 with [NO3+NO2] ground water concentration data each year ranged from 214 to 747. Based on available data, the maximum [NO3+NO2] concentration varied from 0 to 260 mg/l between 1990 and 2000. The mean concentration varied from 1.5 to 4.5 mg/l.

In order to assess the anthropogenic effects on ground water quality, the [NO3+NO2] data were analyzed by classifying the well measurements into four concentration ranges - 0-1 mg/l, 1-3 mg/l, 3-10 mg/l, and >10 mg/l in order of increasing contamination due to anthropogenic effects. The probable background or baseline concentration of [NO3+NO2] in ground water in WRIA 1 was estimated to be about 1 mg/l. In general, the [NO3+NO2] concentrations in the ranges above background concentrations have an increasing trend between the years 1990 to 2000. The worst years, in terms of the percentages of wells with [NO3+NO2] concentrations above the MCL of 10 mg/l, were 1990 and 1998 with 12.1% and 13.3%, respectively. The most affected sub-basins in WRIA 1, in terms of number of years that the maximum [NO3+NO2] concentration exceeded the MCL, were Barrett Lake and Bertrand/Fishtrap/Kamm sub-basins, which had nitrate concentrations in excess of MCL in each year during the period.

The percentage of wells with [NO3+NO2] concentrations exceeding the MCL during 1990 to 2000 varied from 0 to 35.8 mg/l, the highest density being found in the Bertrand/Fishtrap/Kamm sub-basin. This observation was followed by the Sumas River and Scott/Wiser/Schneider basins with 13.3% and 9.1%, respectively. The Sumas-Blaine surficial aquifer is the main affected aquifer in seriously contaminated areas. The main source of nitrate contamination must, therefore, be the land use activities in the area.

2.3.2 Ammonia Contamination

There were 251 wells in WRIA 1 with total ammonia concentration. These wells provided 486 data points of ammonia as nitrogen concentration values between 1990 and 2000 for this study. Ammonia concentrations varied from 0 to 63 mg/l during this period (see Table 2.11). The maximum concentration of 63 mg/l appears to be unusually high and validity of this reading in terms of quality control and quality assurance of the data needs to be verified from the original sources. This high measurement was recorded in Bertrand/Fishtrap/Kamm by the USGS. The little information provided in the original databases suggests that this measurement may be due to the manure storage lagoons present near the measurement location.

When this controversial data point was excluded, the maximum ammonia concentration reduced to 2.2 mg/l. The 99th percentile concentration for ammonia was 2.2 mg/l. The median ammonia concentration varied from 0 to 0.04 mg/l. Other statistics of ammonia concentration data across WRIA 1 are summarized in Table 2.11. The time series of the median, 75th percentile and 95th percentile concentrations are shown in Figure 2.9. The 95th percentile increased from 1990 to 1991, but then decreased in 1992. There were no data available between 1993 and 1995. From 1996 to 1998, the 95th percentile increased and dropped sharply again in 1999. The available data, generally, show that the median, 75th, and 95th percentiles decreased between 1990 and 2000.

The trend of ammonia concentration in various ranges is shown in Figure 2.12. Spatial plots of the annual variation of concentration at various wells are shown in Appendix E. It is noted that the areas with high ammonia concentrations do not correspond with the areas of high [NO3+NO2] concentration. This behavior can be explained by noting that, in general, under oxygenated conditions prevailing in surficial aquifers, nitrate is the more favored nitrogen species and available ammonia is nitrified to nitrate. The reverse is true under anaerobic conditions where ammonia is the favored nitrogen species. The median and maximum ammonia concentrations are summarized by sub-basins in Tables 2.12 and 2.13. Excluding the abnormally high values, it is noted that the median and maximum ammonia concentrations for the sub-basins with data, were less than 1 mg/l.

Table 2.11. Summary statistics of ammonia data across WRIA 1 from 1990 to 2000.

Statistic	1990	1991	1992	1996	1997	1998	1999	1990-2000
No. data points	59	312	10	40		30	2	486
No. of non-zero data	50	205	1	8	13	18	2	297
	84.7	65.7	10.0	20.0	39.4	60.0	100.0	61.1
99th percentile (mg/l)	0.804	2.178	0.182	0.090	0.238	0.076	0.012	1.49
95th percentile (mg/l)	0.421	0.710	0.110	0.052	0.137	0.063		0.55
75th percentile (mg/l)	0.050	0.123	0	0	0.050	0.034	0.012	0.05
Mean concentration (mg/l)	0.100	0.573	0.020	0.010	0.033	0.022	0.011	0.39
Median concentration (mg/l)	0.040	0.020	0	0	0	0.022	0.011	0.02
Standard deviation	0.187	4.799	0.063	0.023	0.060	0.022	0.001	
Min. concentration (mg/l)	BDL	BDL	BDL	BDL	BDL	BDL	0.010	BDL
Min. non-zero conc. (mg/l)	0.010		0.200	0.020	0.022	0.021	0.010	0.01
Max. concentration (mg/l)	0.810	63.000	0.200	0.090	0.255	0.079	0.012	63.0
Range (mg/l)	0.800	62.990	0.000	0.070	0.233	0.058	0.002	62.99
No. in range 0 - 0.02 mg/l	10	137	9	32	20	12	2	222
No. in range 0.02 - 0.06 mg/l	37	84	0	6	5		0	148
No. in range 0.06 - 0.1 mg/l	2	10	0	2	6	2	0	22
No. in range ≥ 0.1 mg/l		81	1	0	2	0	0	94
% data in range < 0.02 mg/l	16.9	43.9	90.0	80.0	60.6	40.0	100.0	45.7
% data in range 0.02 - 0.06 mg/l	62.7	26.9	0.0	15.0	15.2	53.3	0.0	30.5
% data in range 0.06 - 0.1 mg/l	3.4	3.2	0.0	5.0	18.2	6.7	0.0	4.5
% data in range ≥ 0.1 mg/l	16.9	26.0		0.0	6.1		0.0	19.3

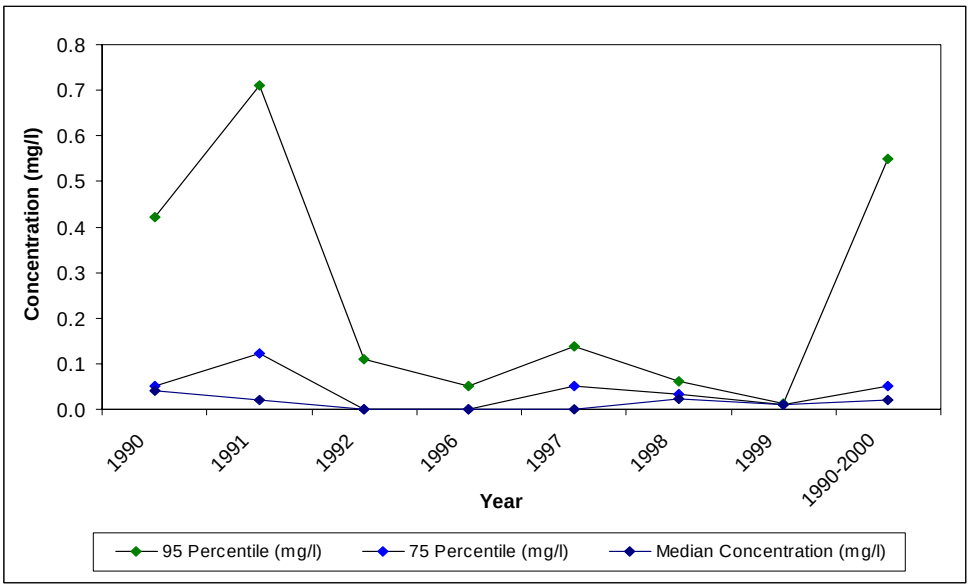


Figure 2.11. Historical trends of median, 75th percentile, and 95th percentile of ammonia concentration in WRIA 1.

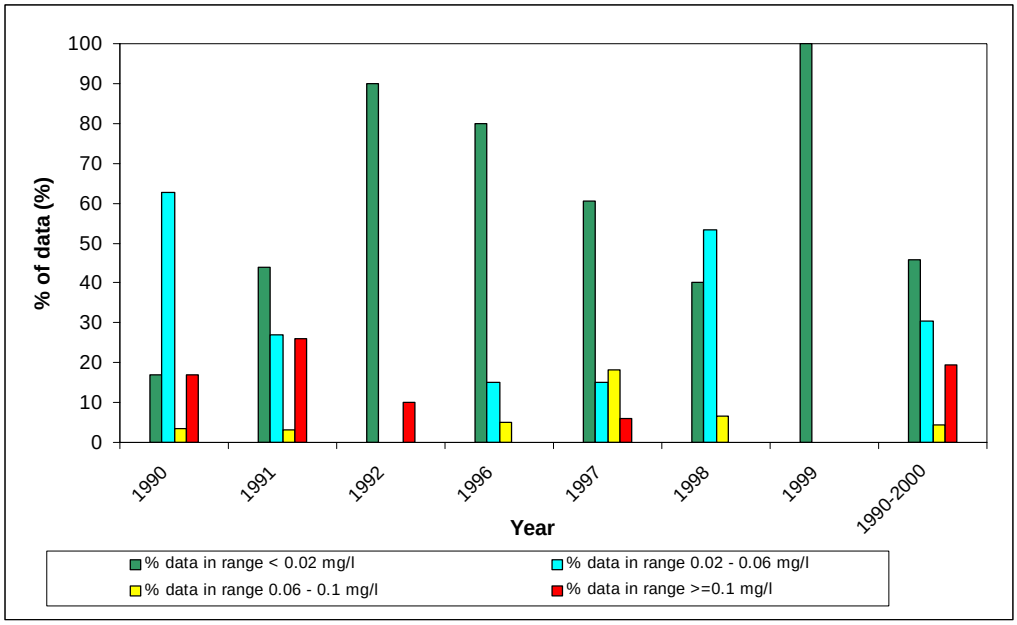


Figure 2.12. Historical trends of percentage of ammonia concentration data in various concentration ranges in WRIA 1.

Table 2.12. Annual median ammonia concentration (mg/l) in different sub-basins of WRIA 1.

Sub-Basin	1990	1991	1992	1996	1997	1998	1999	1990-2000
Skagit County	-		-	-	-	-	-	-
Lummi/Eliza Island	-	-	-	-	-	-	-	-
Lummi Peninsula/Portage Island	-	-	-	-	-	-	-	-
Lummi Bay	-	-	-	0	-	-	-	0
South Fork Nooksack	-	-	-		-	-	-	-
Silver/Nooksack Channel & Delta	-	0.005	-	0	-	-	-	0
Fort Bellingham/Whatcom/Padden /Chuckanut/Fragrance	-	-	-	0	-	-	-	0
Squalicum	-	0.26	-	-	-	-	-	0.26
Anderson/Smith/Deming to Everson	-	0.14	-	-		-	-	0.14
Barrett Lake	0.115	0.025	-	0	-	-	-	0.04
	-	-	-	-	0	-	-	0
Sumas River	0.055	0.04	0.1	-	0	-	-	0.04
Scott/Wiser/Schneider	0.025	0.01	-	0	-	-	-	0.015
North Fork Nooksack	-	-	-	-	-	-	-	-
Dakota/California/Blaine	-	0.02	-	0.04	0		-	0.02
Bertrand/Fishtrap/Kamm	0.03	0.02	0	0.02	0	0.022	0.011	0.02
Point Roberts	-	-	-	0	-	-	-	0
Lake Whatcom	-	-	-	-	-	-	-	-
Middle Fork Nooksack	-	-	-	-	-	-	-	
Little Campbell	-	-	-	-	-	-	-	-
Fraser River	-		-	-	-	-	-	-

Table 2.13. Annual maximum ammonia concentrations (mg/l) in different sub-basins of WRIA 1.

Sub-Basin	1990	1991	1992	1993	1997	1998	1999	1990-2000
Skagit County	-	-	-	-	-	-	-	
Lummi/Eliza Island	-	-	-	-	-	-	-	-
Lummi Peninsula/Portage Island	-	-		-	-	-	-	-
Lummi Bay	-	-	-	0	-	-	-	0
South Fork Nooksack	-	-	-	-	-	-	-	-
Silver/Nooksack Channel & Delta	-	0.01	-	0	-	-	-	0.01
Fort Bellingham/Whatcom/Padden /Chuckanut/Fragrance	-	-	-	0		-	-	0
Squalicum	-	0.62	-	-	-	-	-	0.62
	-		-	-	-	-	-	0.4
Barrett Lake	0.81	2	-	0	-	-	-	2
Terrell/Semiahmoo/Cherry Point	-	-	-	-	0	-	-	0
Sumas River	0.25	0.7	0.2	-	0	-	-	0.7
Scott/Wiser/Schneider	0.05	0.25	-		-	-	-	0.25
North Fork Nooksack	-	-	-	-	-	-	-	-
Dakota/California/Blaine	-	0.04	-	0.05	0	-	-	0.05
Bertrand/Fishtrap/Kamm	0.32	63	0	0.02	0.255	0.079	0.012	63
Point Roberts	-	-	-	0	-		-	0
Lake Whatcom	-	-	-	-	-	-	-	-
Middle Fork Nooksack	-	-	-	-	-	-	-	-
Little Campbell	-	-	-	-	-	-	-	-
Fraser River	-	-	-	-	-	-	-	

2.3.3 Organic Nitrogen Contamination

Data for organic nitrogen concentrations were available for only four years - 1990, 1991, 1997, and 1998. Most of the available data were for 1991. There were only three data points available for 1997 and one for 1998. These data are summarized in Table 2.14 and plotted in Figures 2.13 and 2.14. It should be noted that there were few data points available in selected years between 1990 and 2000 and only 1990, 1991, and 1997 data are shown in the first three columns of Table 2.14. Concentrations for the four years of available data varied from 0.0 (below the detection limit) to 4.0 mg/l. The next largest value was 3.0 mg/l. The values of 4 mg/l and 3 mg/l were detected to the north and northeast, respectively, of Lynden, in the Bertrand/Fishtrap/Kamm sub-basin and in areas overlying the Sumas-Blaine surficial aquifer. The land use in this area is agricultural. The rest of the concentrations were mostly between 0.1 and 0.99 mg/l. A summary of the organic nitrogen statistics by sub-basins is shown in Tables 2.15 and 2.16. These tables show that most sub-basins do not have organic nitrogen data. Only the Bertrand/Fishtrap/Kamm sub-basin had data in each of the three years. The organic nitrogen concentration in this sub-basin declined from 0.445 mg/l to 0.03 mg/l in 1998.

Plots of the areal distribution of organic nitrogen concentration in ground water over WRIA 1, for the four years of data, are shown in Appendix F. The plots indicate that in most wells in the database, the organic nitrogen concentration is greater than 0.1 mg/l over both Sumas-Blaine surficial aquifer and in areas where no surficial aquifers were identified.

Table 2.14. Summary statistics of organic nitrogen data from available data in WRIA 1.

Statistic	1990	1991	1997	1990-2000
No. data points	21	102	3	128
No. of non-zero data	21	99	3	124
% of non-zero data	100	97	100	97
99th percentile (mg/l)	0.67	2.98	0.13	2.46
95th percentile (mg/l)	0.67	0.78	0.12	0.75
75th percentile (mg/l)	0.49	0.52	0.08	0.49
Mean concentration (mg/l)	0.39	0.40	0.06	0.38
Median concentration (mg/l)	0.37	0.28	0.03	0.295
Standard deviation (mg/l)	0.17	0.50	0.06	0.46
Min. Concentration (mg/l)	0.05	BDL	0.01	BDL
Min. Non-zero conc. (mg/l)	0.05	0.03	0.01	0.01
Max. Concentration (mg/l)	0.67	4	0.13	4
Range (mg/l)	0.62	4	0.12	4
No. in range 0 - 0.2 mg/l	2	38	3	45
No. in range 0.2 - 0.6 mg/l	17	48	0	65
No. in range 0.6 - 1 mg/l	2	14	0	16
No. in range ≥0.1 mg/l	0	2	0	2
% data in range < 0.2 mg/l	9.5	37.3	100.0	35.2
% data in range 0.2 - 0.6 mg/l	81.0	47.1	0.0	50.8
% data in range 0.6 - 1.0 mg/l	9.5	13.7	0.0	12.5
% data in range ≥1.0 mg/l	0.0	2.0	0.0	1.6

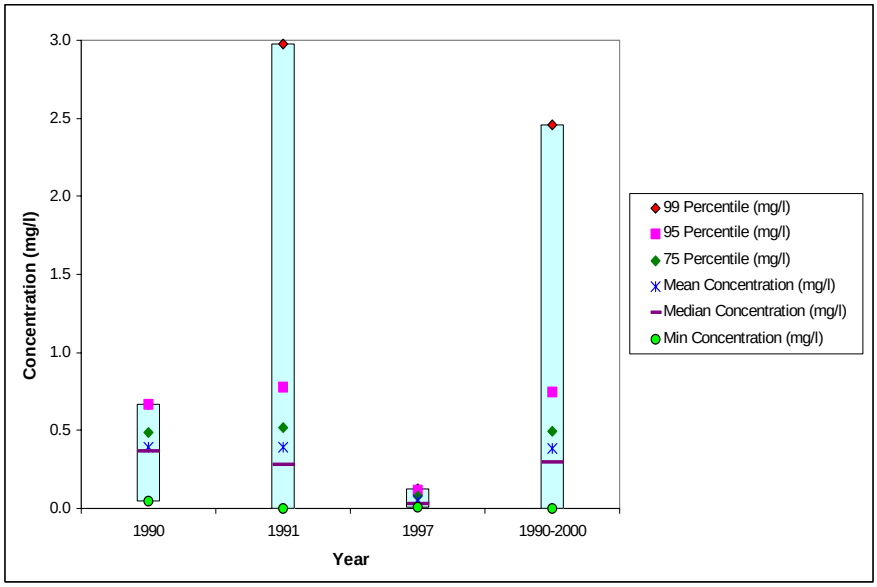


Figure 2.13. Comparison of organic nitrogen statistics for 1990, 1991, 1997, and 1990-2000.

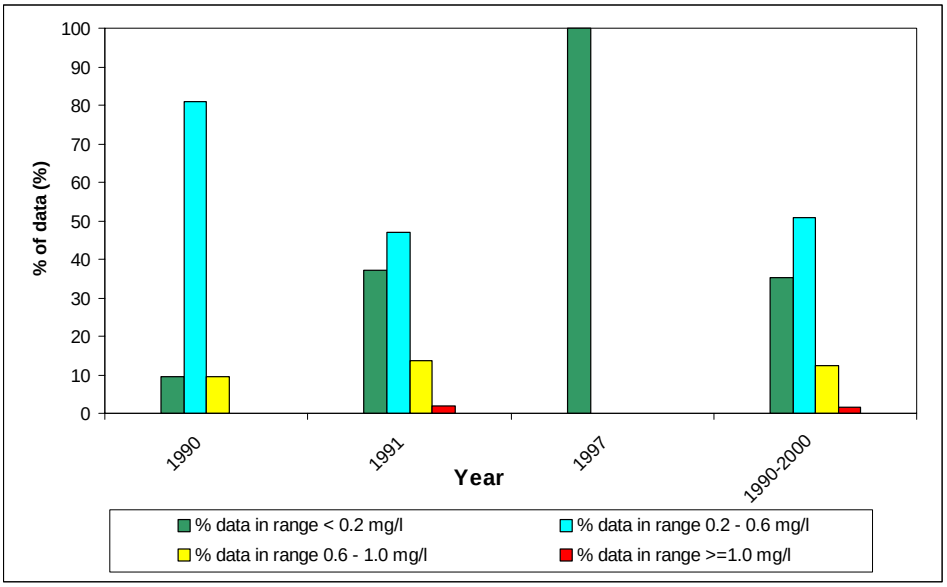


Figure 2.14. Historical trends of percentage of organic nitrogen concentration data in different concentration ranges using all available data.

Table 2.15. Annual median organic nitrogen concentration (mg/l) in different sub-basins.

Sub-Basin	1990	1991	1997	1998	1990-2000
Skagit County	-	-	-	-	-
Lummi/Eliza Island	-	-	-	-	-
Lummi Peninsula/Portage Island	-	-	-	-	-
Lummi Bay	-	-	-	-	-
South Fork Nooksack	-	-	-	-	-
Silver/Nooksack Channel & Delta	-	0.39	-	-	0.39
Fort Bellingham/Whatcom/Padden /Chuckanut/Fragrance	-	-	-	-	-
Squalicum	-	0.175	-	-	0.175
	-	0.2	-	-	0.2
Barrett Lake	0.35	0.295	-	-	0.315
Terrell/Semiahmoo/Cherry Point	-	-	-	-	-
Sumas River	0.15	0.245	-	-	0.24
Scott/Wiser/Schneider	0.56	0.49	-	-	0.525
North Fork Nooksack	-	-	-	-	-
Dakota/California/Blaine	-	-	-	-	-
Bertrand/Fishtrap/Kamm	0.455	0.275	0.03	0.14	0.275
Point Roberts		-	-	-	-
Lake Whatcom	-	-	-	-	-
Middle Fork Nooksack	-	-	-	-	-
Little Campbell	-	-	-	-	-
Fraser River	-	-	-	-	-

Table 2.16. Annual maximum organic nitrogen concentration (mg/l) in different sub-basins.

Sub-Basin	1990	1991	1997	1998	1990-2000
Skagit County	-	-	-	-	-
Lummi/Eliza Island	-	-	-	-	-
Lummi Peninsula/Portage Island	-	-	-	-	-
Lummi Bay	-	-	-	-	-
South Fork Nooksack	-	-	-	-	-
Silver/Nooksack Channel & Delta	-	0.39	-	-	0.39
Fort Bellingham/Whatcom/Padden /Chuckanut/Fragrance	-	-	-	-	-
Squalicum	-	0.175	-	-	0.175
Anderson/Smith/Deming to Everson	-	0.2	-	-	0.2
Barrett Lake	0.35	0.295	-	-	0.315
Terrell/Semiahmoo/Cherry Point	-	-	-	-	-
Sumas River	0.15	0.245	-	-	0.24
Scott/Wiser/Schneider	0.56	0.49	-	-	0.525
North Fork Nooksack	-	-	-	-	-
Dakota/California/Blaine	-	-	-	-	-
Bertrand/Fishtrap/Kamm	0.455	0.275	0.03	0.14	0.275
Point Roberts	-	-	-	-	-
Lake Whatcom	-	-	-	-	-
Middle Fork Nooksack	-	-	-	-	-
Little Campbell	-	-	-	-	-
Fraser River	-	-	-	-	-

2.3.4 Biochemical Conditions Prevailing in Ground water

The main process affecting the concentration of nitrates in ground water in WRIA 1 appears to be leaching of nitrogen from land surface sources to the surficial aquifers. Wells with historically high nitrate levels are concentrated in agricultural areas overlying soils with high permeabilities. The correlation between dissolved oxygen and nitrate in ground water was computed at various locations of WRIA 1. The 95% confidence interval Pearson correlation coefficient for [NO3+NO2] versus dissolved oxygen (refer to Figure 2.15) varied between 0.57 to 0.72. These results suggest the likelihood of aerobic conditions in surficial aquifers with minimum biological activity that can consume oxygen or nitrate as electron acceptors and the opposite is true for deeper layers (Tesoriero et al., 2000). The Pearson correlation coefficient for ammonia versus dissolved oxygen was also computed and the value was 0.25, with a 95% confidence interval of -0.38 to -0.11 (see Figure 2.16). The correlation coefficient for ammonia versus [NO3+NO2] is -0.34 with a 95% CI of -0.26 to -0.42 (see Figure 2.17). These correlations suggest that nitrates are the most important nitrogen pollutants in the aquifers of WRIA 1. The negative correlation between ammonia and [NO3+NO2] is to be expected, since under aerobic conditions, ammonia is oxidized to nitrate, while under anaerobic conditions nitrate is denitrified. It is noted from Figure 2.17 that [NO3+NO2] concentrations were generally higher than those for ammonia, an indicator of potential oxygenated conditions in the Sumas-Blaine aquifer in which most of the wells in the area are

screened. Very few ground water wells had simultaneous ammonia, nitrate, organic nitrogen, and dissolved oxygen concentration data. A few of the wells with data for all four chemicals is plotted in Appendix H. In these plots, it is noted that high concentrations of [NO3+NO2] are accompanied by low levels of ammonia, and vice versa. It is also observed that where oxygen data are available, high dissolved oxygen data correspond to high levels of [NO3+NO2] and low levels of ammonia. Figures 2.18 and 2.19 further show that the correlation between organic nitrogen and ammonia is negative, opposite to that between organic nitrogen and [NO3+NO2].

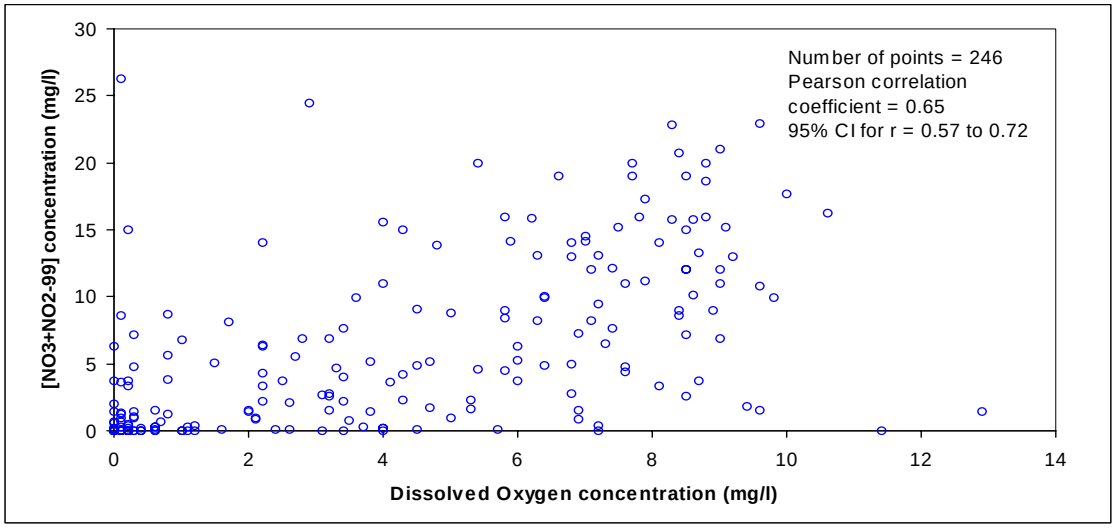


Figure 2.15. Pearson correlation test for [NO3+NO2] versus dissolved oxygen concentration data for less than the 99th percentile value.

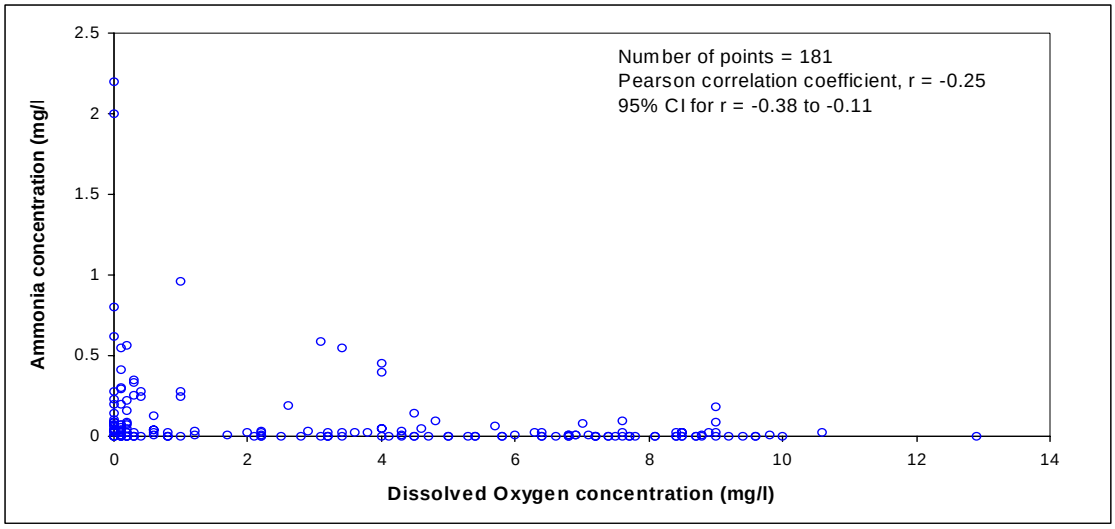


Figure 2.16. Pearson correlation test for ammonia versus dissolved oxygen.

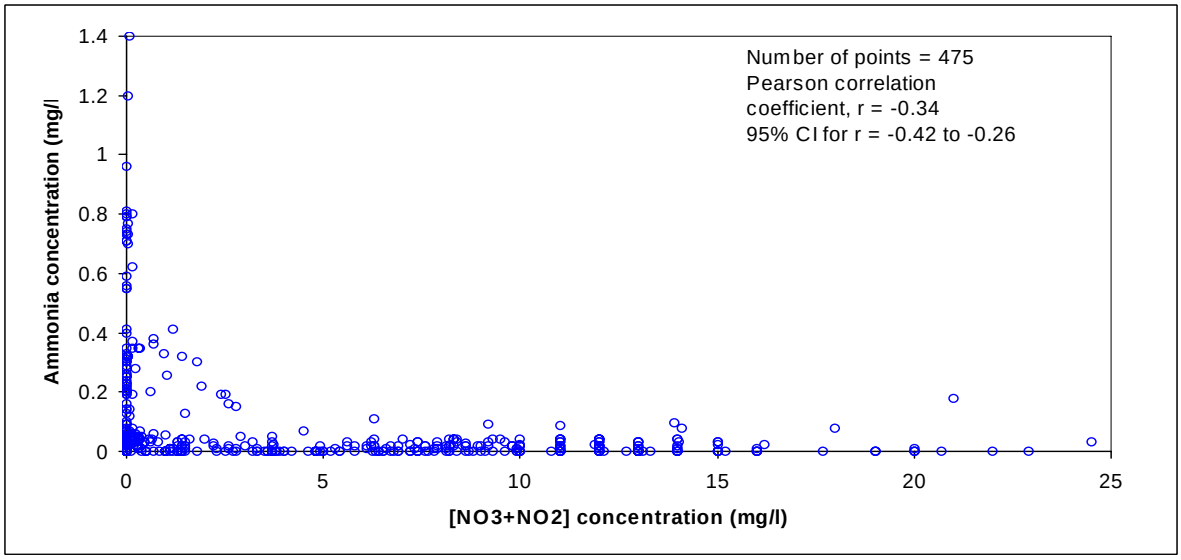


Figure 2.17. Pearson correlation test for ammonia versus [NO3+NO2] concentration data less than the 99th percentile value.

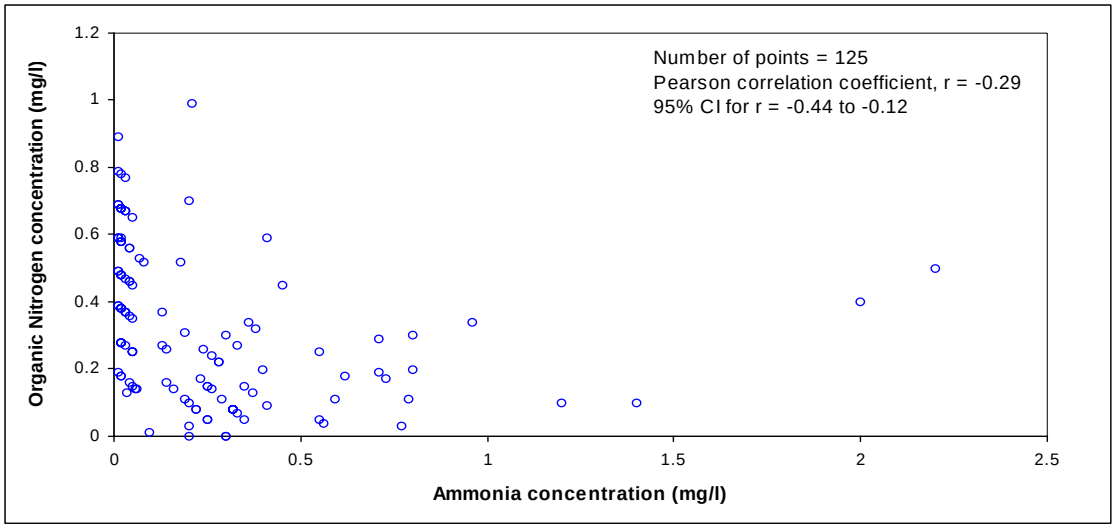


Figure 2.18. Pearson correlation test for organic nitrogen versus ammonia for data less than the 99th percentile value.

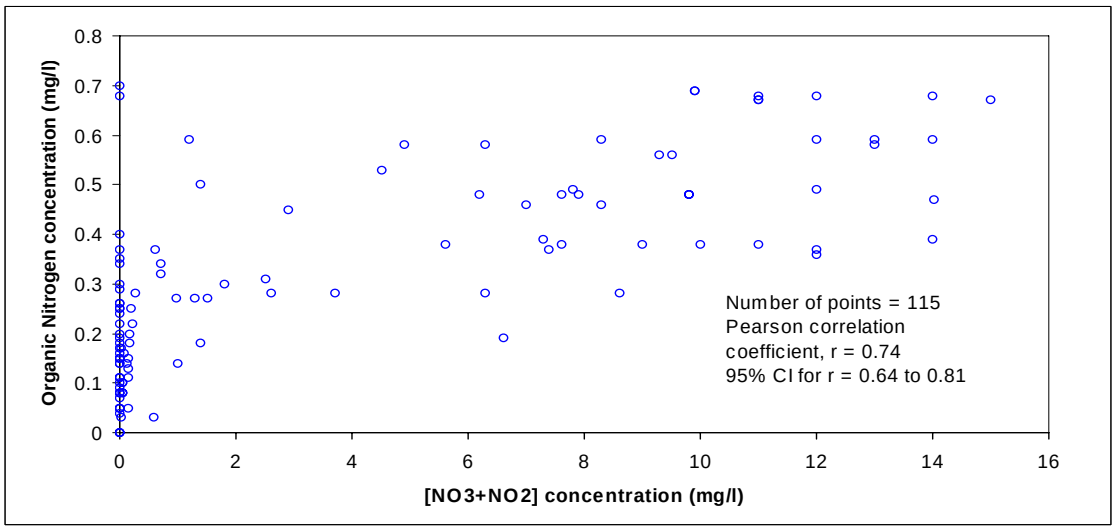


Figure 2.19. Pearson correlation test for organic nitrogen versus [NO3+NO2] for data less than the 95th percentile value.

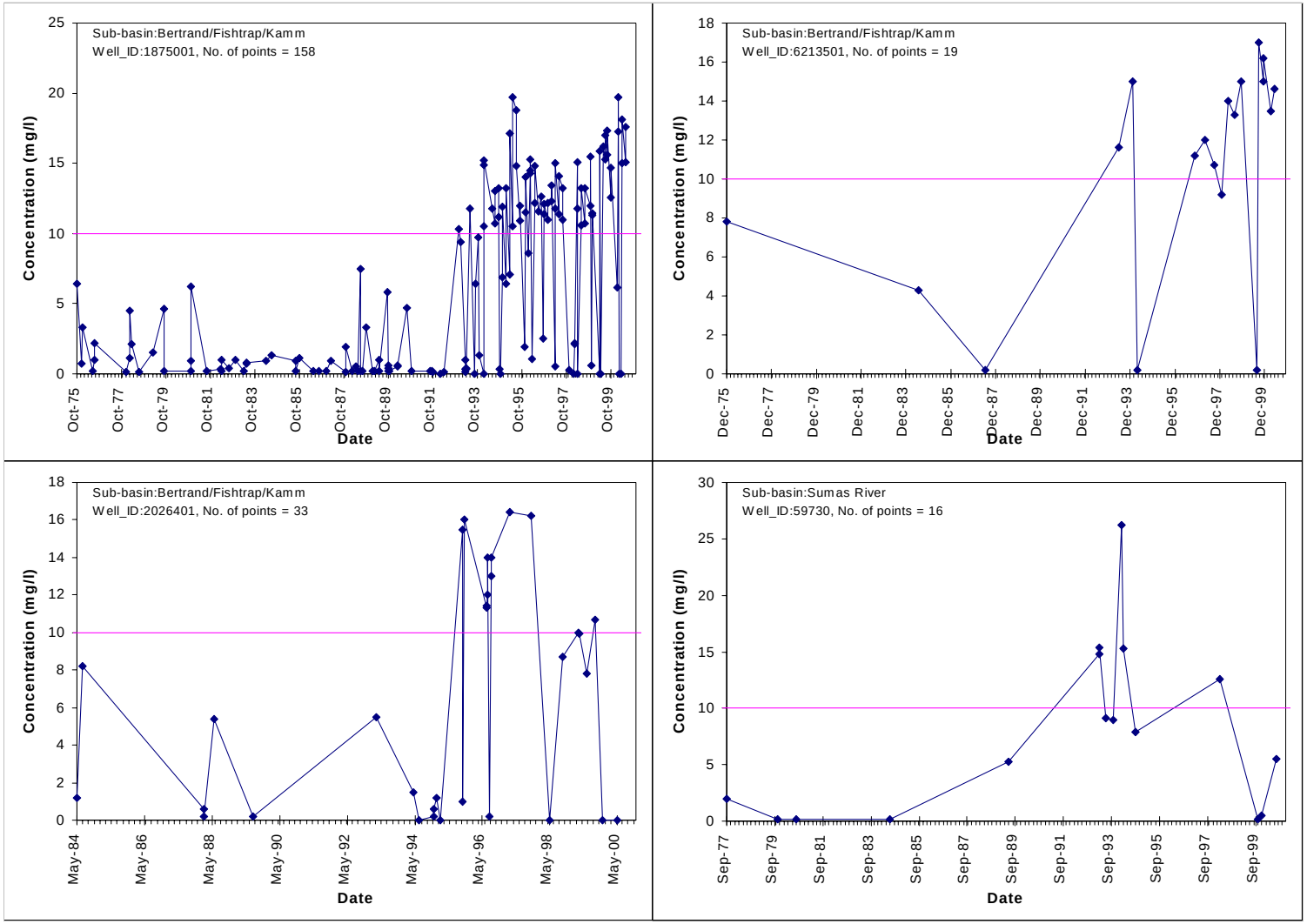


Figure 2.20. Time series plots for wells where yearly maximum concentration exceeded MCL in 3 to 9 years within the period 1990 – 2000.

2.4 Is nitrogen is chemical of concern?

From previous studies in WRIA 1, the most important nitrogen pollutant in WRIA 1 is nitrate. As discussed in the next few sections, the nitrate concentration in many wells is higher than the probable baseline concentrations in the study area. In a study of the Sumas-Blaine aquifer, Erickson (1998) found that 21% of the wells screened in the aquifer had nitrate levels above the MCL of 10 mg/l. From the data used for this report, the annual percentage of wells with maximum nitrate levels exceeding the MCL was in the range 3.8% to 13.3% from 1990 to 2000. Cox and Kahle (1999) found nitrates exceeded the MCL in 25% of the wells sampled in the Sumas aquifer. In this analysis, Bertrand Creek sub-basin had 35.8% of the wells exceeding the MCL of nitrate. The results indicate an increasing nitrate concentration in WRIA 1 except between 1991 and 1993, and, in general, there is cause for concern because the percentage of wells with elevated nitrate concentrations has been increasing.

2.4.1 Baseline Concentration of Nitrate

Baseline or background ground water concentration is the natural contaminant concentration without anthropogenic effects. Establishing the baseline concentration is essential in determining to what extent surface or land use activities have affected the ground water quality in an area. Ideally, the best way to determine the baseline concentration of a particular contaminant in an aquifer is to study pre-development data of the area. Although this type of data is not available in WRIA 1, an attempt will be made in the next sub-sections to estimate the baseline ground water concentrations for nitrates, ammonia, and organic nitrogen.

In the US, the national background concentration of nitrate in shallow ground water is estimated to be about 2.0 mg/l (as nitrogen) (USGS, 1999). In the Puget Sound region (which includes the WRIA 1 study area), ground water nitrate concentrations of greater than 3.0 mg/l are considered due to anthropogenic land use effects (Erwin and Tesoriero, 1997). Cox and Kahle (1999) after studying the nitrate concentrations in the Lynden-Everson-Nooksack-Sumas (LENS) portion of WRIA 1, concluded that 1 mg/l was the upper limit for natural nitrate levels in this area of WRIA 1.

The minimum reported non-zero [NO3+NO2] concentrations from various data sources were:

Washington State Department of Health	- 0.002 mg/l
Washington State Department of Ecology	- 0.01 mg/l
USGS	- 0.01 mg/l
Whatcom County Department of Health	- 0.04 mg/l

To avoid the limitation of lack of large undeveloped areas within the LENS study area, Cox and Kahle (1999) used data from another area with similar geologic settings and concluded that 1 mg/l was the upper limit of natural nitrate concentration for the LENS area.

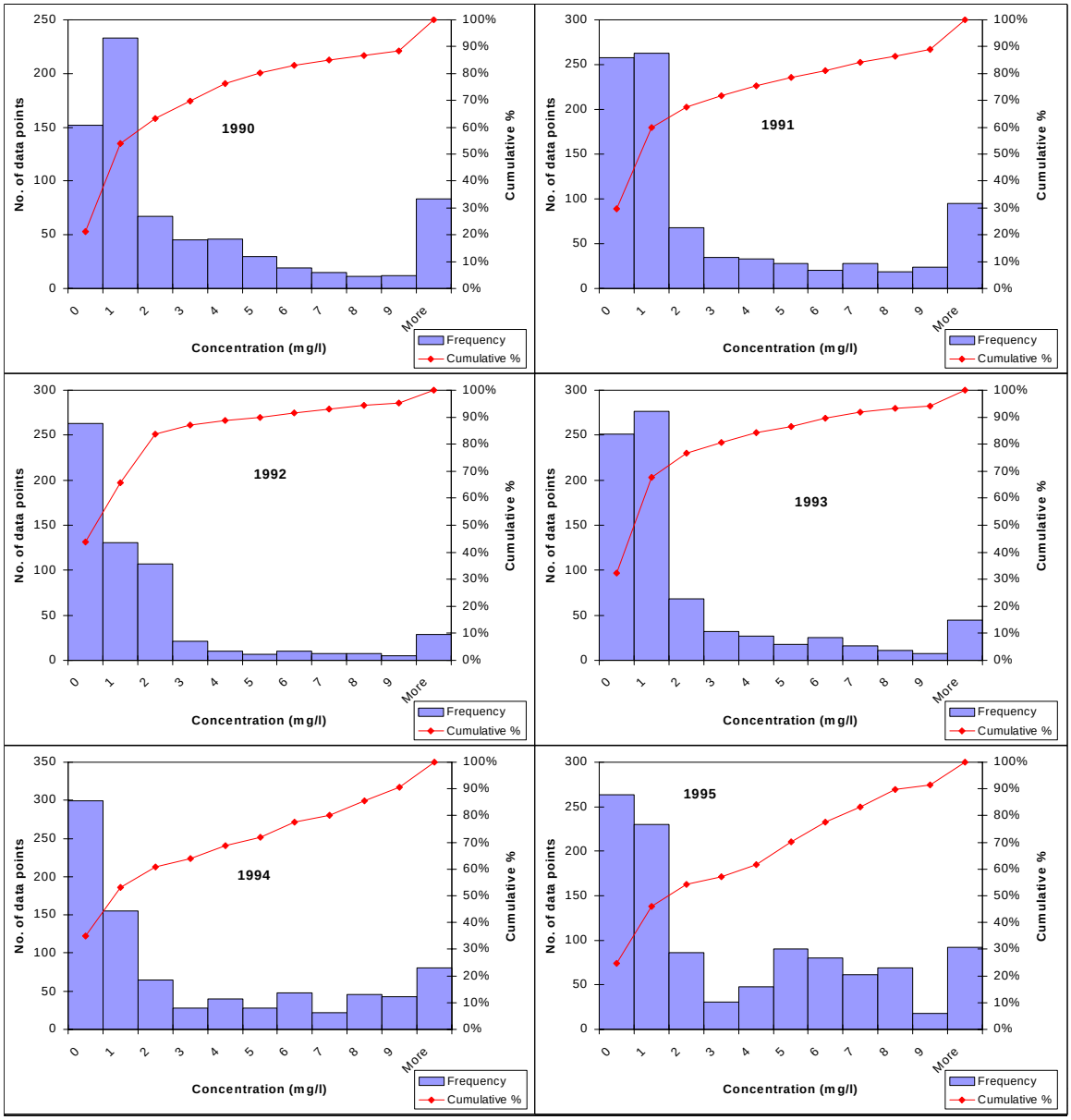


Figure 2.21. Frequency and cumulative frequency plots of [NO3+NO2] data for 1990 to 1995.

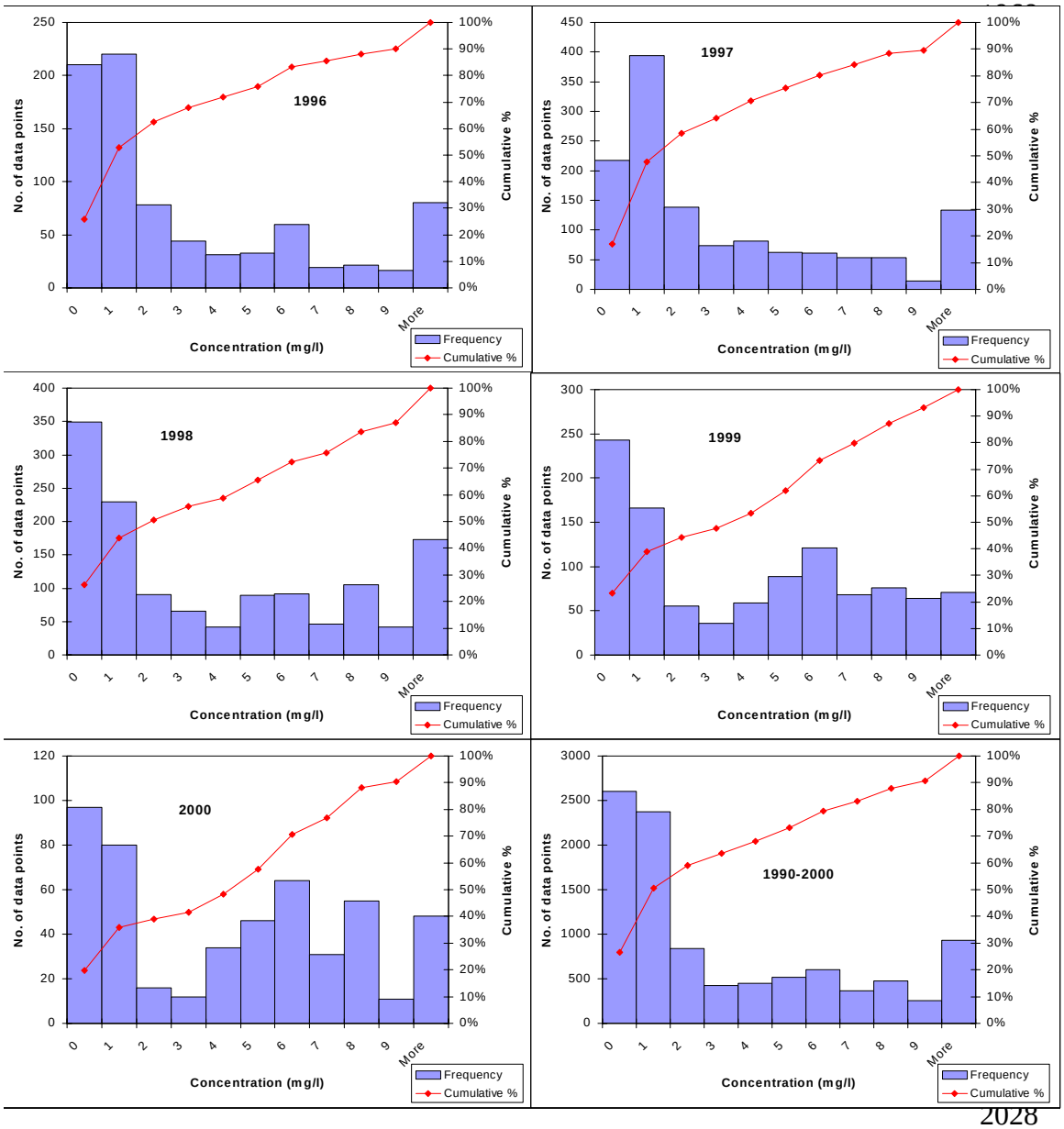


Figure 2.22. Frequency and cumulative frequency plots of [NO3+NO2] data for 1996 to 2000 and for all data for 1990-2000.

Annual statistics of [NO3+NO2] concentration data for 1990 to 2000 are shown in Table 2.1 which shows the median annual concentration of all data ranged from 0.2 to 0.8 mg/l for 1990 to 1994 and 1996, and 1.3 to 4.2 mg/l for 1995 and 1997 to 2000. It is also noted that the median concentration of all 1990 to 2000 data was 1 mg/l. The annual frequency histograms and cumulative frequency curves for all available [NO3+NO2] data are shown in Figures 2.21 and 2.22. It is seen from these charts that from 1990 to 1994 and in 1996, the median [NO3+NO2] concentration had a maximum of 1 mg/l. The median of the population of annual maximum and median [NO3+NO2] concentrations at each well varied from 0 to 1 mg/l and 0 to 1 mg/l, respectively (refer to Tables 2.2 and 2.3). Using these statistics, the upper limit of the baseline concentration of [NO3+NO2] in WRIA 1 is most probably 1 mg/l.

2.4.2 Baseline Concentration of Ammonia

There was a total of 486 total ammonia concentration data points from 251 wells from WRIA 1. The frequency distribution and relative frequency distribution plots for ammonia data for 1990, 1991, 1997, 1998, and 1999 are shown in Figure 2.23. The median concentration varied from 0.01 to 0.04 mg/l. The median of all the available data was between 0.01 to 0.02 mg/l. In 1991, the modal concentration class was 0-0.02 mg/l; in each year of available data, there were ammonia concentrations in this range. In the absence of long-term data from large undeveloped areas in WRIA 1, the analysis from this work suggests that the probable background concentration is between 0.01 to 0.02 mg/l. This value is in agreement with the work of Cox and Kahle (1999) where they found the median concentration of the LENS area to be 0.02 mg/l. Nationally, the background concentration of ammonia in streams is estimated to be 0.1 mg/l (USGS, 1999).

2.4.3 Baseline Concentration of Organic Nitrogen

There was a total of 126 unique data points from 83 wells for total organic nitrogen concentration. The frequency distributions of the data are shown in Figure 2.24. The median organic nitrogen concentration has an upper limit of 0.5 mg/l. Using these results, the baseline organic nitrogen concentration is most likely less than 0.5 mg/l.

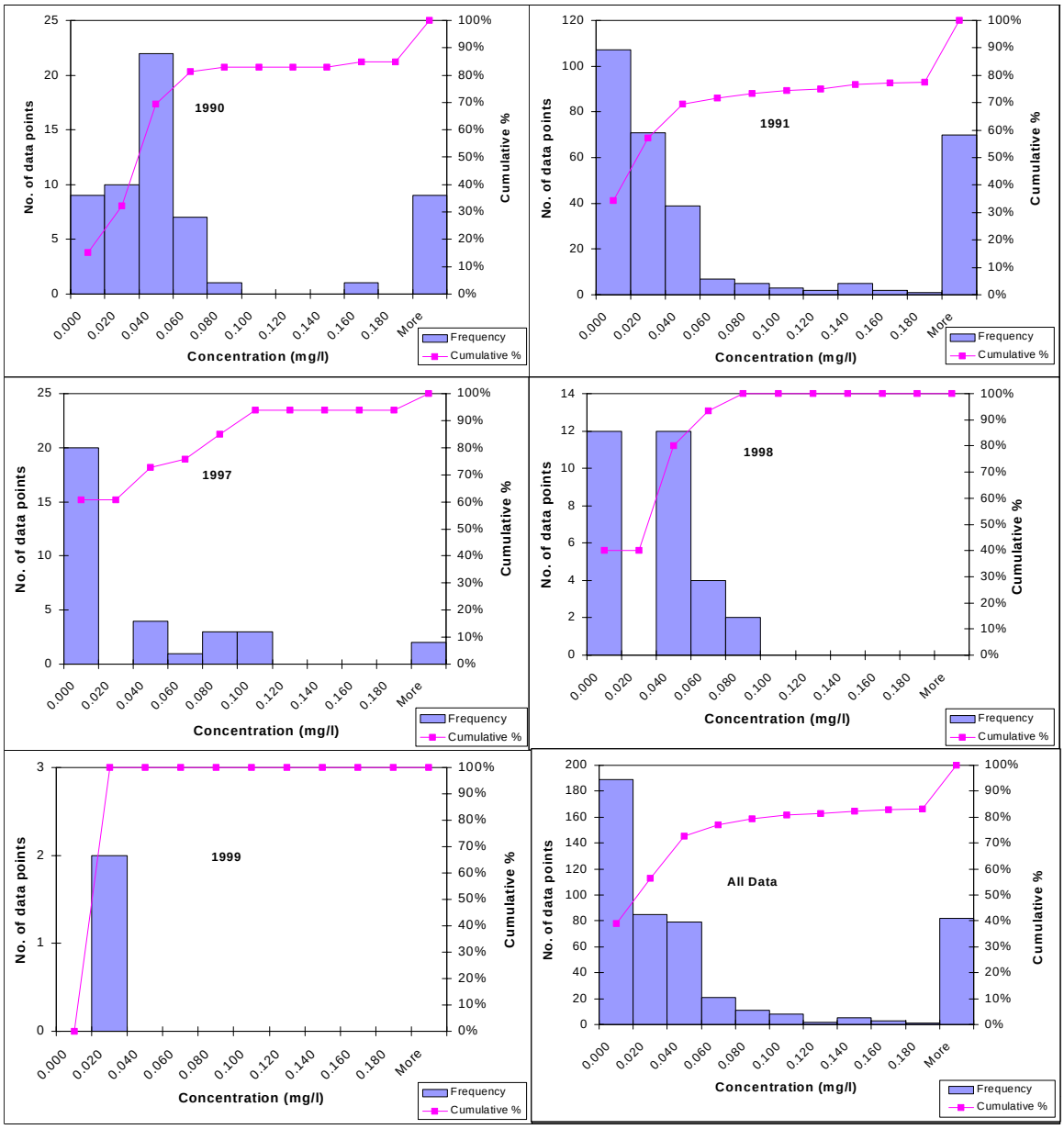


Figure 2.23. Frequency and cumulative frequency plots for ammonia data for 1990, 1991, 1997, 1998, 1999, and for all the available ammonia data.

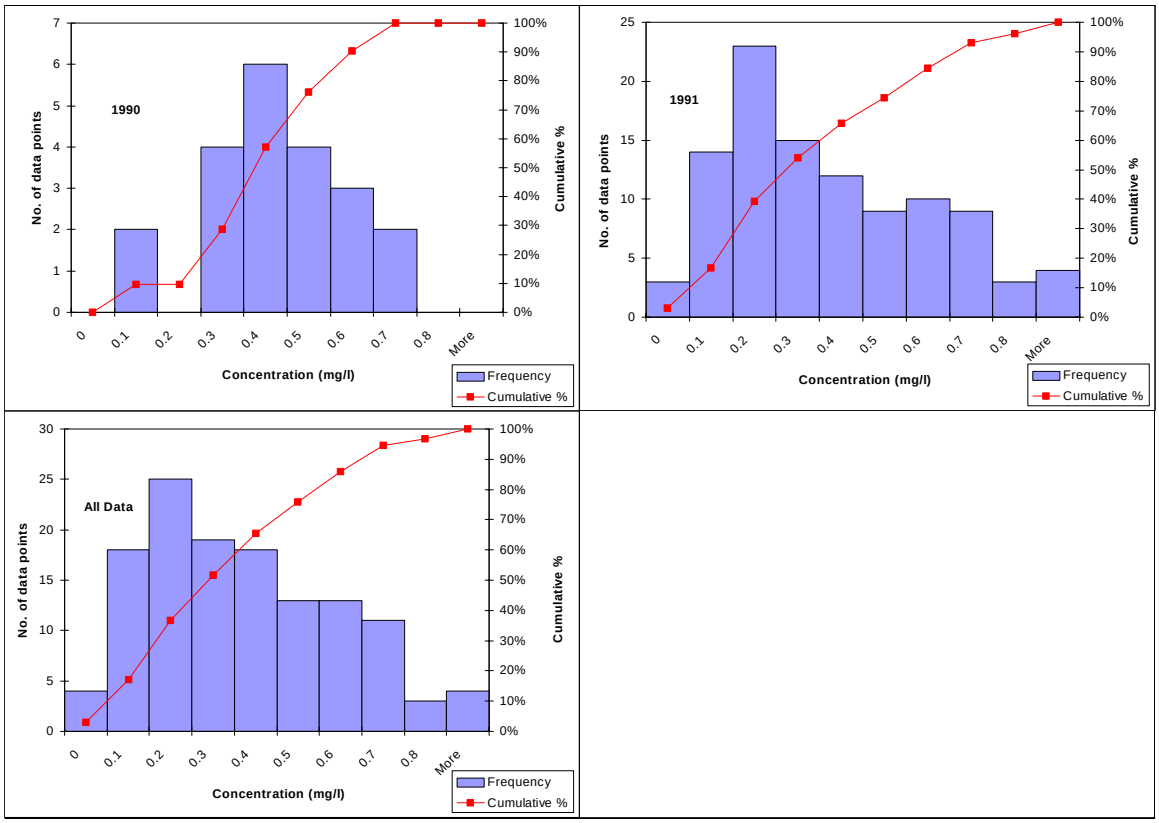


Figure 2.24. Frequency and cumulative frequency plots of organic nitrogen data for 1990, 1991, and for all available data.

2.5 Nitrogen Dynamics in WRIA 1 – Mass Balance

Nitrogen exists in many different forms in the air, water, and soil. Complex physical, chemical and biological processes are constantly transforming these various forms of nitrogen from one form to another and from one location to another in what is called the nitrogen cycle (see Figure 2.21). The nitrogen loading to ground water thus depends on the nitrogen cycle. The nitrogen cycle in any ecosystem is complex, but for the purpose of this report, the major components of the nitrogen cycle in WRIA 1 as they affect the nitrogen loading to ground water can be grouped as follows:

Nitrogen Input

1. Conversion of nitrogen gas to ammonia by nitrogen-fixing microorganisms
2. Nitrogen in applied irrigation water
3. Nitrogen from wet and dry atmospheric deposition
4. Nitrogen mineralized from soil organic matter
5. Nitrogen from fertilizer application
6. Nitrogen from septic tank systems
7. Nitrogen from dairy and poultry manure storage and application to fields
8. Nitrogen from seepage from manure storage lagoons

Nitrogen Output

1. Reduction of nitrates to nitrous oxides and nitrogen gas
2. Loss of ammonia to atmosphere
3. Loss of nitrogen in harvested and exported crop and animals
4. Loss of nitrates to ground water by leaching

2.5.1 Nitrogen Loading to Ground water

Due to the difficulty of obtaining the necessary data for a detailed nitrogen mass balance across the entire WRIA 1, one aggregate sub-basin, the Scott/Wiser/Schneider aggregate sub-basin, was selected for study. The reason for this selection is that the land use in this basin is entirely due to agricultural activities and therefore, the nitrogen loading data are readily available. This aggregate sub-basin is made up of the Scott, Wiser, and Schneider sub basins. These selected contiguous sub-basins all drain to the lower Nooksack sub-basin and have a combined area of 21,147 acres. The estimates of the input and output of nitrogen to the sub-basin were done based on the methods employed by Cox and Kahle (1999) in the LENS area. Table 2.17 shows the spreadsheet of the estimates of the various input and output values of nitrogen to the sub-basin. Table 2.18 shows the estimates of the nitrate loading to soil and ground water of the sub-basin.

The estimated nitrate loading to ground water is 563,356 lbs/year when soil organic nitrogen mineralization is not considered and 694,591 lbs/year when soil organic nitrogen mineralization is considered. The estimated ground water nitrate concentration was 3 to 3.7 mg/l, depending on whether organic matter mineralization is considered. In this wok, the assumption is made that rainfall and irrigation were the only sources of water recharge to ground water. From the available data, the median nitrate concentration for the sub-basin is 1.96 mg/l and the mean nitrate concentration is 3.64 mg/l.

Table 2.17. Estimates of annual amounts of nitrogen applied and deposited in the Scott/Wiser/Schneider aggregate sub-basin of WRIA 1.

Area							
Total study area	=	21147	acre	85578873	m^2		
Irrigated area	=	6730	acre	27235344	m^2		
Dairy farm land base area	=	11,764	acre	47607219	m^2		
Commercially farmed area	=	13,460	acre	54470687	m^2		
Non-commercially farmed area	=	3021	acre	12225553	m^2		
Residential cultivated acreage (lawns and gardens)	=	239	acre	967199	m^2		
Precipitation and applied irrigation water							
Precipitation	=	41	in	1.0414	m	89121838	m^3
Effective precipitation factor	=	0.85					
Infiltrated precipitation	=	34.85	in	0.88519	m	75753562	m^3
Irrigation	=	12	in	0.3048	m	8301333	m^3
Wet Deposition							
Average (NO3+NH3) concentration in rainwater	=	0.26	mg/L	19696	kg	43422	lb
Fraction to ground water		0.5					
Dry Deposition							
Regional deposition rate	=	1	Lbs./acre	9592	kg	21147	lb
Redeposition from dairy land base	=	15	Lbs./acre	152101	kg	335325	lb
Fraction to ground water	=	0.4					
Irrigation							
Nitrate concentration in ground water used	=	1.96	mg/L	16271	kg	27452	lb
N fraction to ground water		0.25					
Organic Matter Mineralization							
Mineralization rate in soil	=	195		3538020	kg	7800000	lb
N fraction to ground water	=	0.05		176901	kg	390000	lb

Table 2.17… /contd.

Inorganic Fertilizer Application

Crop	Acreage (acre)	% fertilized	Appl. Rate (lbs/acre)	N Applied lbs	N Applied kg
Peas	480	20	25	2400	1089
Carrots	9.6	100	100	960	435
Potatoes	192	100	70	13440	6096
Blueberries (mineral soil)	19.2	100	100	1920	871
Blueberries (peat soil)	9.6	100	50	480	218
Raspberries	480	100	80	38400	17418
Sweet corn	57.6	100	125	7200	3266
Pasture/Hay	1920	25	45	21600	9798
Pasture (dairy)	5913.6	55	60	195148	88518
Silage/feed corn (dairy)	1190.4	100	90	107136	48596
				388684	176304

Total inorganic fertilizer application to soil =				176304	kg	388685	lb
Fraction to ground water =		0.25					

Residential Area Fertilizer Application

Application rate	=	200	lb/acre	21682	kg	47800	lb
Fraction to ground water	=	0.25					

Nitrogen from Septic Systems

Population using domestic septic systems	=	3344	persons				
N loading rate to soil	=	10	lbs/person	15168	kg	33440	lb
N loading rate to ground water	=	6.8	lbs/person	10314	kg	22739	lb
Fraction to ground water	=	0.68					

Nitrogen from Dairy manure Storage, Handling and Application to Fields

Herd composition	No. of cows	Manure N produced (lbs/day)	Total N (lbs/herd/yr)
Lactating cows	150	94.5	34492.5
Dry cows	30	15.1	5511.5
Immature	20	8.7	3175.5
Total	200	118.3	43179.5

Table 2.17... /contd.

Number of dairy farm herds	=	58	farms			
Tot. N produced/farm	=	43179.5	lbs/yr/farm			
Total manure N produced	=	2504411	lbs/yr			
Fraction of farm manure lost	=	0.7				
Available nitrogen to field	=		340795	kg	751323	lb
N fraction to ground water	=	0.3				
Nitrogen from Poultry Manure Storage, Handling, and Application to Fields						
Total No. of broilers	=	171000	/year			
Lifespan of broilers	=	50	days			
Manure N production rate	=	0.001	lbs/day/broiler			
Fraction of N lost to volatilization and denitrification	=	0.5				
Available N to soil	=	1939	kg	4275	lb	
N fraction to ground water	=	0.3				
Nitrogen from Seepage of Ammonia from Dairy Manure Lagoon						
No. of dairies	=	58				
Proportion of dairies with earthen lagoons.	=	0.85				
Average surface area of lagoon	=	30000	ft2			
Seepage rate	=	0.365	m/yr			
Ammonia conc. of seepage	=	840	mg/l			
Ammonia lost to denitrification	=	0.15	fraction			
Total N in seepage to soil	=	42128	kg	92876	lb	
Total N in seepage to ground water	=	35809	kg	78945	lb	
Nitrogen from Legumes						
Acres of peas =	380	acre				
Acres of red alder =	2850	acre				
Total acreage =	3230	acre				
N to ground water =	5	lb/acre	7326	kg	16150	lb

Table 2.18. Estimates of annual nitrogen loading to soil and ground water of Scott/Wiser/Schneider aggregate sub-basin.

Natural Process or land use Land use activity	Nitrogen Applied or released to soil (lbs)	Nitrogen Entering Ground water System (fraction)	NO3-N entering ground water system (lbs)	% of total including soil organic matter mineralization	% of total excluding soil organic matter mineralization
N precipitation, wet	43,422	0.5	21,711	3.1	3.9
N precipitation, dry	21,147	0.4	8,459	1.2	1.5
Legumes			16,150	2.3	2.9
Septic tank effluent	33,440	0.68	22,739	3.3	4.0
Law/garden fertilizers	47,800	0.25	11,950	1.7	2.1
Redeposition of volatilized manure N	176,460	0.4	70,584	10.2	12.5
Nitrogen in irrigation water	35,871	0.25	8,968	1.3	1.6
Mineralization of soil organic matter	2,624,700	0.05	131,235	18.9	
Fertilizers (inorganic)	388,685	0.25	97,171	14.0	17.2
Manure storage (lagoon leakage)	92,876	0.85	78,945	11.4	14.0
Manure application	755,598	0.3	226,679	32.6	40.2
Total	4,219,999	N/A	694,591	100	100

Total nitrogen to ground water (including mineralization)	=	694,591	lb
Total nitrogen to ground water (excluding mineralization)	=	563,356	lb
Ground water recharge from rainfall	=	75,753,562	m3
Deep percolation of rainfall (fraction)	=	0.4	
Ground water recharge from irrigation	=	8,301,333	m3
Deep percolation of irrigation (fraction)	=	0.4	
Total recharge	=	84,054,895	m3
Nitrate concentration (including mineralization)	=	3.7	mg/l
Nitrate concentration (excluding mineralization)	=	3.0	mg/l

2.5.2 Best Management Practices for Minimizing Nitrogen Contamination

Inorganic fertilizers account for a relatively large proportion of the nitrogen inputs to ground water in WRIA 1. The following fertilizer management practices, adapted from Self and Waskom (1998), could reduce the quantity of nitrogen contributed by fertilizers to the ground water:

- Performing a soil nitrogen analysis to determine the nitrogen needs in farms
- Setting a realistic crop yield goal for each field
- Estimate the net nitrogen requirement of the crop by taking into account all sources of nitrogen (manures, water, organic matter, etc.) available to the crop
- Splitting fertilizer applications into as many separate applications as practicable

Other effective nitrogen management practices include:

- Discourage the use of land-based manure storage lagoons due to potential leakage of nitrate to ground water. Instead, encourage the use of lagoons lined with stable and non-porous materials.
- Although the data do not indicate large-scale nitrogen loading from septic tank systems, close monitoring of future development domestic septic tank systems in WRIA 1 and ensuring their proper operation are important.

Some these measures may already be in place due to local and state regulations and efforts. If so, regular monitoring should be performed to ensure that these measures are effective and to find alternatives and/or changes to further improve the effectiveness.

2.5.3 Field Monitoring for Nitrogen Species

Management of land use practices while maintaining good ground water quality involves making choices among alternative practices. The processes involved in the transformation of nitrogen in WRIA 1 and the corresponding effects on ground water are complex. Selection of the best management practices would require a sound understanding of the processes affecting [NO3+NO2] concentrations. Hence, a computer model of fate and transport of nitrogen should be developed to assess the effects of various management alternatives to maintain ground water quality of WRIA 1.

The currently available water quality data give a general idea of the spatial and temporal trends of the nitrate pollution problem in WRIA 1. A summary of the number of available data points and their breakdown by sub-basins is shown in Table 2.19.

The main objective of future data collection efforts should be to provide the data for detailed modeling of nitrogen dynamics across WRIA 1 and thereby developing a computer based management model to aid in decision-making. In order to assess the future data needs and develop monitoring plans, the priority areas by sub-basin that have experienced high

[NO3+NO2] stresses in excess of the MCL should be developed. For this purpose, the results of the analysis performed from this study can be readily used.

High priority areas for future data collection are identified on the basis of maximum [NO3+NO2] concentrations in a basin, the number of years the annual maximum concentration exceeded the MCL, the number of currently available wells and data, and the percentage of wells in sub-basins with concentrations exceeding MCL. Based on these conditions, the priority areas for future data collection are described in Table 2.20. It should be noted that Fraser River and Little Campbell were included as high priority sub-basins for data collection as these areas lack any form of data (see Table 2.19) to assess the true nitrogen contamination of ground water.

Table 2.19. Summary statistics of currently available nitrogen data in WRIA 1.

	[NO3+NO2]		**Ammonia**		**Organic Nitrogen**		**Dissolved Oxygen**	
Sub-Basin	**Data**	**Wells**	**Data**	**Wells**	**Data**	**Wells**	**Data**	**Wells**
Skagit County	46	3	0	0	0	0	0	0
Lummi/Eliza Island	220	84	0	0	0	0	0	0
Lummi Peninsula/Portage Island	72	31	0	0	0	0	16	16
Lummi Bay	922	183	7	5	0	0	19	18
South Fork Nooksack	64	62	0	0	0	0	0	0
Silver/Nooksack Channel & Delta	270	119	4	4	1	1	3	3
Fort Bellingham/Whatcom/Padden /Chuckanut/Fragrance	100	79	2	2	0	0	0	0
Squalicum	309	227	6	6	6	6	6	5
Anderson/Smith/Deming to Everson	562	274	6	6	3	3	6	6
Barrett Lake	1755	501	80	34	28	15	26	26
Terrell/Semiahmoo/Cherry Point	382	130	4	4	0	0	0	0
Sumas River	2253	248	54	28	21	17	21	21
Scott/Wiser/Schneider	1035	386	80	24	18	7	16	14
North Fork Nooksack	528	185	0	0	0	0	0	0
Dakota/California/Blaine	1076	555	12	10	0	0	7	5
Bertrand/Fishtrap/Kamm	1792	467	217	119	50	35	135	101
Point Roberts	288	10	9	9	0	0	0	0
Lake Whatcom	193	138	0	0	0	0	0	0
Middle Fork Nooksack	44	39	0	0	0	0	0	0
Little Campbell	1	1	0	0	0	0	0	0
Fraser River	0	0	0	0	0	0	0	0

Table 2.20. Priority distribution of sub-basins for future data collection.

High Priority	Medium Priority	Low Priority
Barrett Lake	Terrell/Semiahmoo/Cherry Point	Skagit County
Sumas River	Lummi/Eliza Island	Lummi Peninsula/Portage Island
Scott/Wiser/Schneider	Lummi Bay	Fort Bellingham/Whatcom/Padden/ Chuckanut/Fragrance
Bertrand/Fishtrap/Kamm	Silver/Nooksack Channel & Delta	Middle Fork Nooksack
Fraser River	Anderson/Smith/Deming to Everson	South Fork Nooksack
Little Campbell	Dakota/California/Blaine	Squalicum
		North Fork Nooksack
		Lake Whatcom
		Point Roberts

The most important nitrogen species to be monitored are:

- Nitrate
- Nitrite
- Organic nitrogen
- Ammonia
- Dissolved oxygen

The main modeling objectives of the future data collection program will be to assess the effects on ground water quality due to various land-uses and water management practices and thereby help to determine the best management practices to implement to minimize ground water pollution.

2.6 Modeling Approach

The objectives of detailed nitrogen modeling in WRIA 1 are to develop a mathematical model of fate and transport of nitrogen in ground water that can provide information related to nitrogen transport in ground water due to various land use and ground water management activities. Once a well-calibrated and mathematical model is developed, the model can be repeatedly used to identify the potential impact on ground water quality due to both existing land uses and proposed land uses and ground water development alternatives. Once a broad range of alternatives and the corresponding impacts on ground water quality are identified, the best alternative out of the proposed set of alternatives can be implemented through discussion between regulators and stakeholders.

Based on the physical scale of WRIA 1, a fate and transport model at very fine spatial resolution, such as grid sizes of few meters, cannot be developed and simulated due to limited data availability at fine spatial resolution and the limited computational facilities to

simulate scenarios with more than hundreds and thousands of nodes or grids. In order to obtain the necessary information from a modeling exercise while maintaining sufficient accuracy and satisfying modeling objectives, a compromise is needed.

The proposed methodology is based on two approaches, i.e., developing a lumped-parameter model to simulate the regional-scale fate and transport of nitrogen across entire WRIA 1, and a detailed vertical cross-sectional model to simulate transport at local-scale at a few selected sites.

2.6.1 Local-Scale Modeling

The objectives of local-scale modeling are to understand the dynamics of nitrogen fate and transport in selected areas of WRIA 1 such that this knowledge can be extended to regional-scale modeling. Since the local-scale modeling will be performed at finer spatial resolution, the data requirement is more intensive than in the case with a lumped parameter model. However, the advantage of this local-scale modeling approach is that more accurate information related to dynamics of nitrogen fate and transport can be obtained. For example, the modeling can provide information on nitrogen transformation potential in the unsaturated zone, mass loading of nitrogen to ground water, the net mass loading to the subsurface from an above-ground manure pile, the effect of mass loading due to recharge, effects of nitrogen kinetic parameters and soil type on nitrogen transport, etc. In addition, this modeling approach can also provide information related to relative degree of importance of various input parameters such that subsequent data gathering can be more focused and improved to eliminate redundancy.

The initial focus areas to include in this local-scale modeling will be selected from Bertrand/Fishtrap/Kamm and Scott/Wiser/Schneider sub-basins where high levels of nitrates were found in this study. The actual model areas will be selected in Phase III of this project.

The local-scale modeling will use two-dimensional vertical cross-sectional domains that include both the unsaturated zone and the aquifers. Since nitrogen dynamics in soil-ground water is controlled by a complex system of physical, chemical, and biological processes (see Figure 2.1), a few simplifications may be needed in the model to account for nonlinear mass reaction terms in the final set of mass transport equations. In addition, the actual number of species from the nitrogen cycle to be modeled will also be decided at a later stage based on the relative importance in overall decision-making.

The proposed mass transport model to be used in the analysis is HYDRUS-2D (http://talus.mines.edu/research/igwmc/) available through the International Ground water Modeling Center, Colorado School of Mines, CO. This model can simulate flow and transport of multiple species in variable unsaturated subsurface and has user-friendly interfaces for mesh generation, data input, and output processing in the windows environment. The model is capable of handling different types of chemical kinetic processes for sorption and chemical, physical, and biological decay. Model calibration will be performed based on available field data from the specific study areas. It should be noted that accuracy and applicability of the model results will be governed mostly by model calibration

and every attempt will be made to perform adequate model calibration unless data are limited. The simulation scenarios will include a series of base cases supplemented by simulations to assess parameter sensitivity. The information gathered from these simulations will be transferred to regional-scale modeling to better assess the nitrogen transport due to large-scale land use activities.

2.6.2 Regional-Scale Modeling

Analysis of spatial and temporal trends of [NO3+NO2] concentrations in WRIA1 is a difficult task due to the complexities of the processes and interactions of the nitrogen cycle and the corresponding impacts on leaching of nitrates from the unsaturated zone to ground water. The development of a conventional mathematical model would require the collection of large amounts of site-specific data over a long period of time and can be an expensive undertaking.

An appropriate alternative to a conventional model would be an areal fate and transport analysis of selected nitrogen species using a lumped-parameter model. Lumped parameter models use equivalent hydrogeologic and fate and transport parameters assumed to represent large geographic areas instead of over numerical grids or cells as in the case of detailed finite difference or finite element models. The information gathered from local-scale modeling described earlier, can now be used to develop averaging techniques to compute the lumped model parameters. In general, averaging techniques are used for estimating the lumped-parameters to account for spatial variability (Barrett and Charbeneau, 1996).

The proposed regional-scale, areal, fate and transport model using lumped-parameters is similar to the approach used by (Barrett and Charbeneau, 1996). In the proposed modeling approach for WRIA 1, the study area will be divided into a set of representative cells, around 20 cells, corresponding to different sub-basins of WRIA 1. For each cell, the key factors influencing the leaching of nitrogen to ground water would include:

- Fertilizer application rates
- Manure application rates; these rates will include estimates of manure produced in each sub-basin from dairy and poultry.
- Rainfall, irrigation, and evapotranspiration rates
- Nitrogen transformation processes such as mineralization, nitrification, denitrification, volatilization, and immobilization; in this regard, work of Cox and Kahle (1999), Tesoriero et al. (2000) and others on nitrogen dynamics in ground water of WRIA 1 can be readily used.
- Water and nitrogen flow rates across cell boundaries through stream aquifer interactions
- Crops grown in each cell and their net nitrogen consumption rates
- Soil and ground water characteristics.
- Nitrogen inputs from other sources such as domestic septic tanks

In most ground water quality modeling studies, the transformations in the nitrogen cycle are described by first-order reaction kinetics. The use of non-linear adsorption relationships is possible when the concentration of nitrate is high, but the use linear isotherm normally gives adequate practical results (Ling and El-Kadi, 1998). In this work, we propose to use first-order reaction kinetics to describe various transformation processes of the nitrogen cycle. In addition, nitrate will be the dominant species undergoing both fate and transport and other nitrogen species will be assumed to undergo reaction only. This approach is considered reasonable as nitrate has a low adsorption capacity whereas other nitrogen species become adsorbed to exchange sites more rapidly, thus reducing the mobility.

The model will be verified using other similar fate and transport models. Model calibration will be performed based on available data from WRIA 1. The output of the model will provide the nitrogen loading to ground water in a sub-basin as a result of a given set of land use and water management practices. The model will also estimate the transient concentrations of different nitrogen species in different sub-basins. The final modeling package will be dynamically linked to the ground water quality database and will be incorporated in the proposed decision-support system.

2.7 Summary and Conclusions

In recent years, there have been concerns about elevated concentrations of nitrogen in ground water of Water Resource Inventory Area-1 (WRIA 1), Washington State. Ground water is an important source of water for most inhabitants in WRIA 1. Agricultural practices involving inorganic fertilizer application, land application of manure, and the use of earthen manure storage lagoons have been identified as the main sources of nitrogen contamination of ground water in WRIA 1. The most commonly used aquifer in the area is the Sumas-Blaine surficial aquifer, which extends into Canada. The aquifer in the US portion is recharged mostly by precipitation though some recharge is from ground water flow from Canada; this recharge contains high nitrates as a result of similar agricultural practices.

Historical ground water data from four agencies, United States Geological Survey, Whatcom County Department of Health, Washington Department of Health, and the Washington Department of Ecology, were analyzed in this study. The analyses showed a general increase in ground water nitrate concentration levels in recent years. The areas with the most elevated nitrate concentrations were over the main surficial aquifer, Sumas-Blaine aquifer. The area just south of the Canadian border, near Sumas had a high proportion of wells with nitrate levels exceeding the MCL of 10 mg/l. Many other wells had concentrations in the range 1 mg/l to greater than 10 mg/l, indicating nitrate contamination from land use activities. The proportion of wells with nitrate concentrations less than probable natural levels has generally decreased in recent years.

The percentage of wells with [NO3+NO2] concentrations exceeding MCL during 1990 to 2000 varied from 0 to 35.8%, with the highest density found in the Bertrand/Fishtrap/Kamm sub-basin. [NO3+NO2] concentrations in this range are far above probable natural or baseline concentrations and are the result of anthropogenic land use activities. Only 25.1% of the wells in the Bertrand/Fishtrap/Kamm sub-basin had [NO3+NO2] levels below the

baseline [NO3+NO2] concentrations. The annual maximum nitrate concentrations in this sub-basin were always above the MCL ranging from 11.8 mg/l to 43 mg/l, with the peak occurring in 1991. The maximum nitrate concentration recorded in the early 1990s was from Barrett Lake sub-basin with a value of 260 mg/L. In the latter part of the 1990s, Bertrand Creek sub-basin recorded concentrations as high as 39 mg/L in 1999 and 20 mg/L in 2000. Sumas River sub-basin recorded nitrate concentrations as high as 26 mg/L in 1998. In general, the median and mean concentrations continued to rise. The median nitrate concentration also exceeded the MCL in three years 1995, 1996 and 1999. During the other years, the annual median concentration was close to the MCL, in the range of 3 to 10 mg/l. [NO3+NO2] concentrations in this range also are the result of anthropogenic effects.

The sub-basins with nitrate problems almost as severe as that in the Bertrand/Fishtrap/Kamm sub-basin are the two sub-basins just to the south, Scott/Wiser/Schneider and Barrett Lake sub-basins, and the Sumas River sub-basin to the east. These sub-basins had MCL exceedances in 8 to 11 years during the period of 1990 to 2000. All the sub-basins contiguous to these sub-basins have similar, though less severe nitrate contamination problems. Generally, the [NO3+NO2] concentrations in all these sub-basins have been increasing; these sub-basins have high fractions both in wells and individual observations with elevated [NO3+NO2] concentrations. It appears that the nitrate contamination problem is spreading from the Bertrand/Fishtrap/Kamm sub-basin to the rest of WRIA 1. This type of transport is expected as nitrate is less-sorbing compared to most chemicals including other nitrogen species (retardation coefficient is almost one) and therefore, highly mobile.

Considering the fact that the main source of nitrogen in ground water of WRIA 1 is agricultural practices related to manure and inorganic fertilizer applications, the following general land and water use management practices would help reduce nitrogen levels and curb the geographical spread of contamination in the area:

- Setting a realistic crop yield goal for each field.
- Analyze soils to determine available soil nitrogen for crops and applying just enough inorganic fertilizers and manure to achieve crop yield targets. Manure should also be tested to determine its nitrogen content before application.
- Estimate the net nitrogen requirement of the crop by taking into account all sources of nitrogen (manures, water, organic matter, etc.) available to the crop.
- Splitting fertilizer applications into as many separate applications as practicable, avoiding applying excess nitrogen during periods of low crop growth rate, such as early spring.
- Provide sufficient storage for manure. The use of land-based manure storage lagoons should be discouraged unless these are designed properly to prevent leakage through proper lining using stable and non-porous materials.
- Reducing the density of domestic septic tank systems and ensuring their proper operation.
- Protect streams and other water bodies by providing zones of vegetation around them and restrict livestock access to these areas.

As a management tool to predict the impacts of alternative management practices on ground water quality in the sub-basins of WRIA 1, necessary modeling tools need to be developed to

assist in management decision-making. A lumped parameter model is the most appropriate for WRIA 1, considering the currently available data, the size of the area, and the complexity of the task. Limited local-scale modeling needs to be performed at selected sub-basins to obtain a good understanding of nitrogen dynamics pertaining to different soil and chemical conditions.

In gathering data for future modeling,

- highest priority should be given to the Bertrand/Fishtrap/Kamm and surrounding sub-basins.
- The important species to be monitored are nitrate, nitrite, ammonia, and organic nitrogen.
- Ground water quality data should be collected at least twice a year.
- More sub-basin specific data on land use should be gathered.

3. PESTICIDES IN GROUND WATER

In the past, more than 12 pesticides have been detected in ground water in WRIA 1, primarily in the Sumas-Blaine aquifer. These chemicals include pesticides such as ethylene dibromide (EDB), 1,2-dichloropropane, and 1,2-dibromo-3-chloropropane (DBCP) which have been banned in USA. The presence of pesticides in ground water of WRIA 1 is not surprising as the Sumas-Blaine aquifer is used heavily for commercial agriculture. Also, the direction of ground water flow across most of the international boundary is from north to south, which may influence the transport of pesticides used in Canada to the US part of the Sumas-Blaine aquifer. Typically, pesticide contamination, if any, is of concern in the region occupying Sumas-Blaine aquifer. However, the threat of pesticide contamination of ground water exists in other areas of WRIA 1 as well due to shallow ground water depths and the potential mobility of pesticides in ground water.

Generally, soil fumigants have been a major concern in WRIA 1 due to their use on raspberries, seed potatoes, and berries, prior to planting. Among the soil fumigants that have been detected in WRIA 1 are EDB, 1,2-dichloropropane, and DBCP.

3.1 Ethylene Dibromide

Ethylene dibromide (EDB) is a soil fumigant that was used as a nematocide to kill rootworms in strawberry, raspberry, seed potato, and other crops over several decades. It was also used in grain storage and as a component of leaded gasoline. EDB was banned from agricultural use by the US EPA in 1983 (http://www.epa.gov/oppfead1/international/piclist.htm) due to potential carcinogenic effects. The chemical name of EDB is 1,2-dibromoethane, and synonyms include DBE; alpha, beta-dibromoethane; dibromoethane; ethylene bromide; glycol bromide; glycol dibromide; and sym-dibromoethane. Product names include bromofume, celmide, dibrome, dowfume, EDB-85, fumo-gas, kopfume, nephis, and soilfume.

Since raspberries, berries, and seed potatoes are the main crops in the region, ethylene dibromide, was used on these crops until 1983/1984.

3.1.1 Health Effects

EDB is highly toxic to humans via ingestion producing severe skin and eye irritation, and can cause skin blistering with contact (EXTOXNET database- (http://ace.ace.orst.edu/info/extoxnet/pips/edb.htm). The available laboratory data from exposure studies on animals indicate that reproductive effects may be possible in humans, but that there may be a threshold level below which effects are unlikely. The available data suggest that EDB does not cause tetrogenic effects. It has not been shown to be mutagenic in humans, either in cell cultures or in the whole organism. The available evidence indicates that EDB may be carcinogenic. Human contact with EDB may damage the lungs, skin, and

eyes. Acute and chronic systemic effects may be seen in the liver, kidneys, and heart, and other internal organs and systems. Lung injuries can also lead to secondary effects such as pneumonia and respiratory tract infections. EDB is readily and rapidly absorbed across the lungs, skin, and gastrointestinal tract in liquid and vapor forms. Once absorbed, it is rapidly broken down, and metabolites may be found in the urine, kidneys, liver, adrenal glands, pancreas, and spleen of EDB-exposed animals shortly after application. About two-thirds of the applied dose are excreted through urine or expired air.

The drinking water standard or the MCL for EDB is 0.05 µg/L or parts per billion (ppb).

3.1.2 Chemical Fate in Soil and Ground water

The fastest degradation of EDB occurs at or near the soil surface. It is moderately persistent in the soil environment; a representative field half-life was estimated to be about 100 days. In one study, after two months, almost all of EDB was dissipated. Sunlight readily degrades EDB where near the soil surface, EDB is converted to ethylene and bromide ions. The small percentage of the EDB which remains is unchanged, possibly adsorbed to soil organic matter or clay particles or entrapped in soil micro pores. EDB thus entrapped may be inaccessible to microbial degradation and may slowly leach to ground water over long periods. Such leaching is slow at normal temperatures, but increases with higher temperatures. Ground water contamination by EDB has been confirmed by various studies conducted in WRIA 1 (Cox and Kahle, 1999). Although EDB was banned in 1983/1984, it was detected in ground water in WRIA 1 as recent as 2000.

Although EDB has a density higher than water, commonly called as a dense-nonaqueous phase liquid (DNAPL), the application rates of pesticides are much smaller than typical rates associated with long-term leakages and spills of industrial solvents; hence, the actual transport of EDB in ground water as a soluble plume may be small compared to most DNAPLs.

3.1.3 Use of Ethylene Dibromide

Historically, WRIA 1 has been dominated by agricultural related land uses. Agricultural practices have been concentrated in the LENS area. In 1984, Whatcom County was described as having more than 550 acres of berry farms that yielded more strawberries than any other county in the state (O' Herron, 1999). Since berry plantations used significant amounts of EDB, this information describes the importance of EDB usage and the potential for ground water contamination. The following Bellingham Herald newspaper report on February 15, 1984 by Tappersohl and Fernando describes the use of EDB in the area;

> "Soil on more than 416 acres of Whatcom County farmland was fumigated with the chemical EDB in the last three years…(according to the State Dept. of Agriculture)… More than 69 acres in Whatcom County were treated with EDB in 1981, at least 32 acres of strawberry and 37 acres of potato fields…In 1982, 124 acres were treated…That rose again last year to 223 acres."

The use of EDB in the Canadian portion of the area is not known clearly. EDB was banned from use by Canadian authorities in 1984. However, the use of EDB prior to1984 is possible as berries were the major crop in the Canadian portion of WRIA 1.

3.1.4 Analysis of EDB Contamination

The available water quality data suggest that EDB has been detected in two areas of WRIA 1 and these are the Meadowdale and Bertrand Creek areas. Both these sites are near the city of Lynden. The Meadowdale area is northeast of Lynden and the Bertrand Creek area is to the west of the city (see Figure 3.1). It is interesting to note that both areas are located in the Sumas-Blaine aquifer.

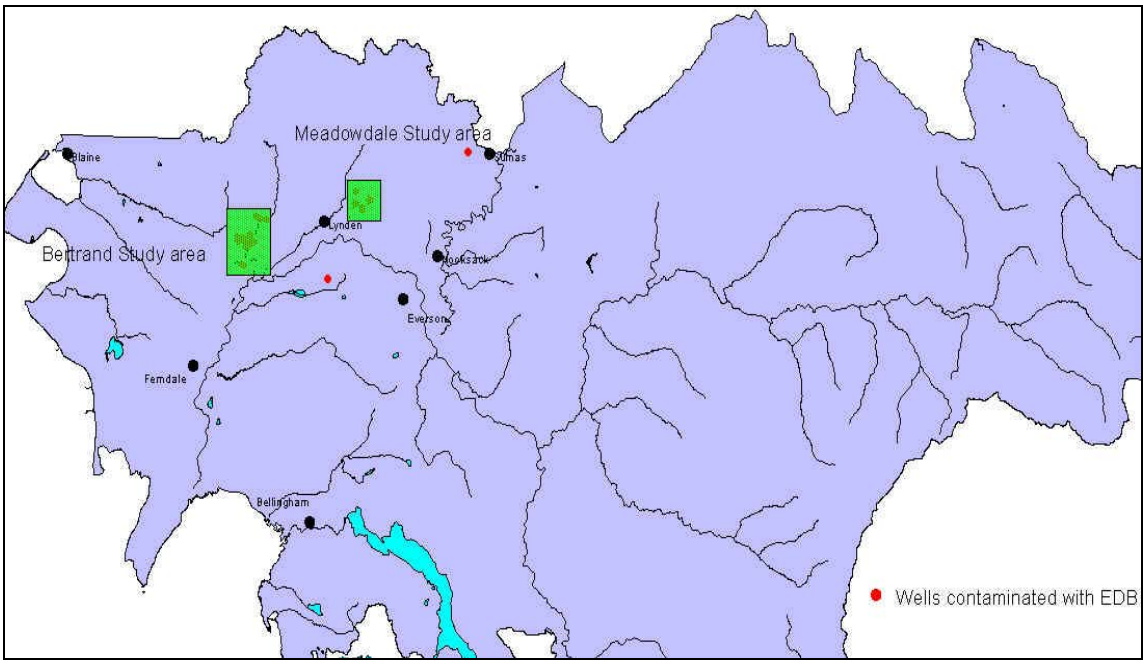


Figure 3.1. Map showing Meadowdale and Bertrand Creek areas (in green) that are contaminated with EDB.

While the MCL for EDB is 0.05 ppb, concentrations as high as 6.1 ppb have been detected in WRIA 1. The ground water wells exceeding the MCL for each year are described in Appendix I. Figure 3.2 shows the variation of maximum concentration of EDB detected in WRIA 1 with time. A decreasing trend with time is observed which indicates a steady degradation of EDB in ground water of WRIA 1.

Although the concentration of EDB has been decreasing through the past decade due to natural attenuation, the threat of EDB in ground water cannot be ruled out, especially in the

southern part of the Bertrand Creek where concentrations as high as 0.186 ppb have been observed as recent as 1999.

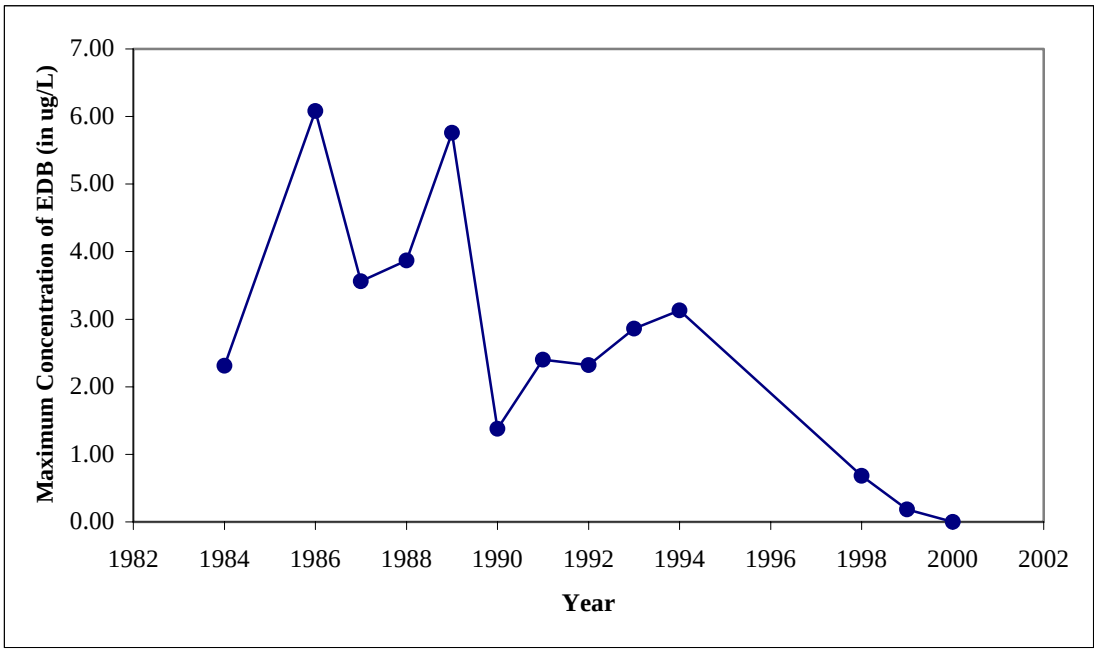


Figure 3.2. Time series of maximum concentration of EDB detected in WRIA 1.

Impacts on Sub-Watersheds

Based on the ground water quality data collected in WRIA 1, the maximum concentration of EDB at each sub-basin was compared with the MCL to identify which sub-basins have been impacted. Based on this approach, the following EDB affected sub-basins have been identified (see Table 3.1):

1. Bertrand Creek
2. Fishtrap Creek
3. Schneider
4. Kamm
5. Scott
6. Johnson

The affected sub-watersheds and the wells showing undesirable levels of EDB during any given year between 1984 and 1999 are shown in Figure 3.3. It should be noted that concentrations exceeding MCL were not detected in the above-mentioned sub-watersheds in each year; instead all sub-watersheds where EDB was detected in excess of MCL at least once between 1984 and 1999 were listed. Table 3.1 provides information related to sub-watersheds with EDB concentrations exceeding the MCL for each year.

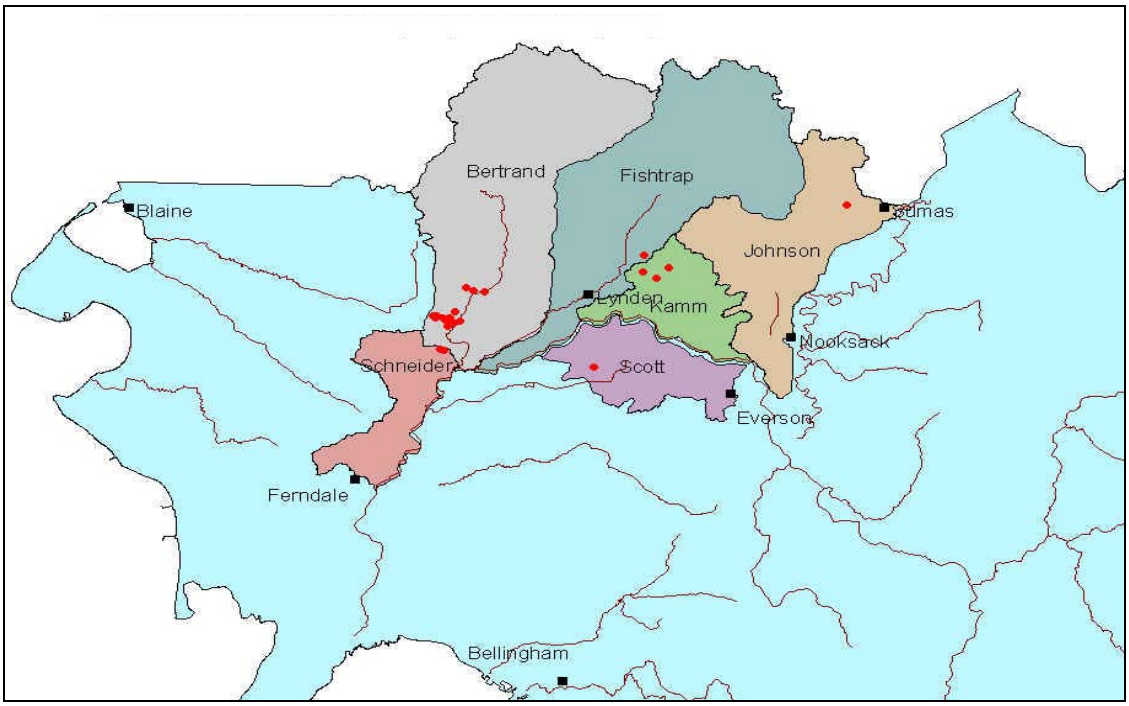


Figure 3.3. Map showing the sub-watersheds affected by EDB contamination between 1984 and 1999.

Table 3.1. Information of sub-watersheds that exceeded MCL of EDB.

Year	Sub-Watershed	Maximum EDB Concentration (ppb)
1984	Bertrand Creek	2.31
	Fishtrap Creek	1.25
	Scott	0.72
1986	Bertrand Creek	6.10
1987	Bertrand Creek	3.56
	Kamm	0.09
1988	Bertrand Creek	3.87
	Kamm	0.35
	Johnson	0.54
1989	Bertrand Creek	5.76
1990	Bertrand Creek	1.38
	Kamm	0.17
1991	Bertrand Creek	2.40
1992	Bertrand Creek	2.32
1993	Bertrand Creek	2.86
	Kamm	0.28
1994	Bertrand Creek	2.39
	Kamm	0.06
	Johnson	0.50
1998	Bertrand Creek	0.68
	Schneider	0.07
1999	Schneider	0.19

Bertrand Creek Area

Bertrand Creek area comprises the southern part of Bertrand Creek and the northern edge of Schneider sub-watershed. Figure 3.4 shows the Bertrand Creek area with wells that exceeded MCL. This area is the focus of study of EDB contamination in WRIA 1 because a large number of wells have been detected with EDB exceeding the MCL with values as high as 6.1 ppb. The extent of EDB contamination in this area is shown in Appendix I. As recent as 1999, an EDB concentration of 0.186 ppb was observed in a well located near the Willeys Lake Road in the northern edge of Schneider sub-watershed (ATSDR, 2000, Weston, 1999).

A large number of these ground water wells shown in Figure 3.4 are located along the Birch Bay-Lynden Road. Some of these wells are located near the Loom-Trail Road-Weidkamp Road area and along the Willeys Lake Road near the Osoyoos Lake (located in the Schneider sub-watershed).

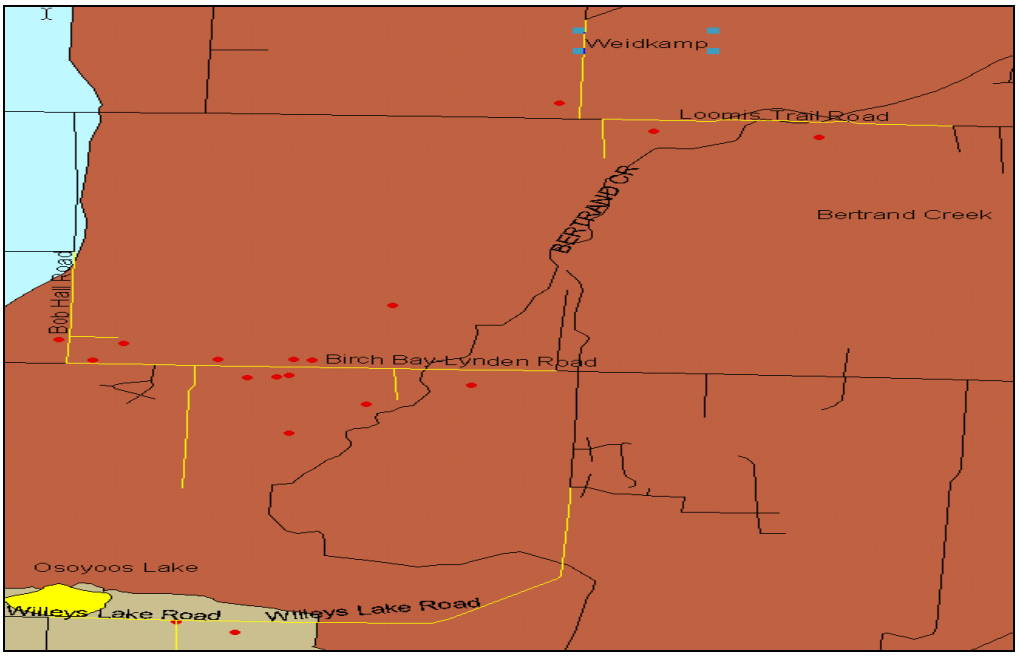


Figure 3.4. Map of Bertrand Creek area with ground water wells in red that exceeded the MCL between 1984 and 1999.

Bertrand Creek area is a part of the larger, berry-growing zone in the northern county (O'Herron, 1999). Strawberry cultivation was common in the area prior to 1983 and used EDB as a pesticide. This historical use of EDB in the area has caused the contamination of the surficial unconfined aquifer of the area. The EDB concentrations present in ground water at the present time are the residual mass from historical use that is still undergoing natural attenuation and some transport.

Meadowdale Area

The existing data show that Meadowdale area has also been affected by EDB contamination. As many as four wells that have been used to supply water (one of which supplied water to the Meadowdale Water Association) have been detected in the past with undesirable EDB levels. Three of these wells are located in the block formed by Haveman Road, Kamm Road, Line Road, and Northwood Road. The other well is located to the east of the Northwood Road. Figure 3.5 shows the Meadowdale area along with the wells that exceeded the MCL.

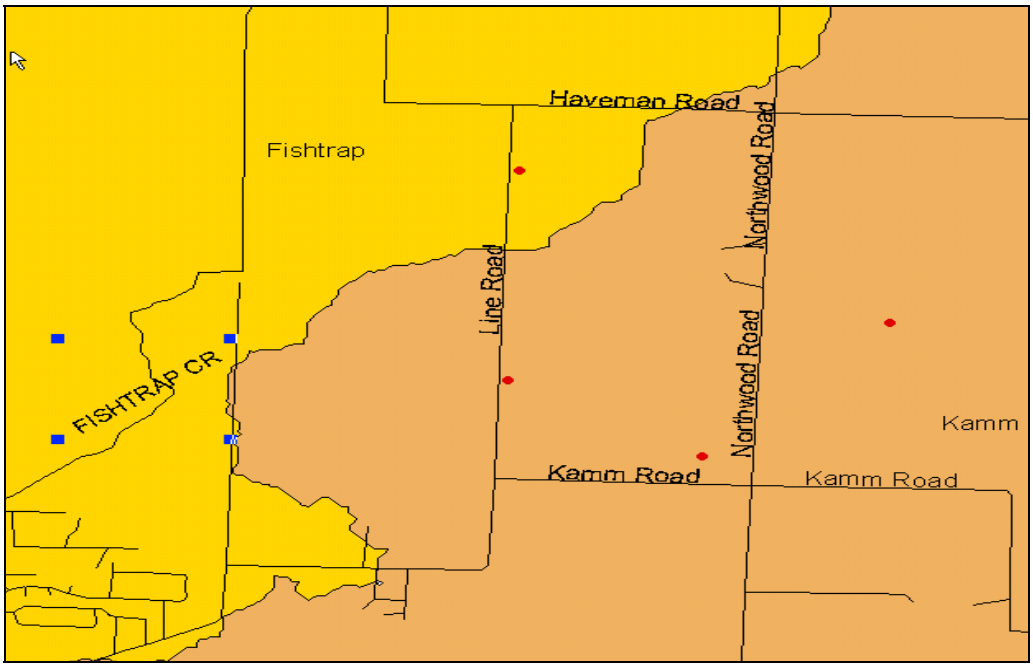


Figure 3.5. Map indicating the Meadowdale area and the ground water wells that exceeded the MCL of EDB after 1983.

The area has low to moderate runoff and moderate to high infiltration (refer to Figure 1.13). The maximum hydraulic conductivity is between 6 to 20 inches per hour (refer to Figure 1.14). Due to these hydrologic conditions, vertical migration of EDB through the unsaturated zone in to shallow ground water is relatively fast. EDB has been detected in this area between 1984 and 1994 only and the observed maximum concentration was 1.25 ppb in 1984. Unpublished results of sampling performed by Washington Department of Ecology in 2000 showed EDB still present in several drinking water wells in the Meadowdale area. Figures 3.6 to 3.16 show the EDB concentrations in the area from 1984 to 1998. In addition to the Bertrand Creek and Meadowdale areas, one well to the south of Van Dyk Road (located in the Scott sub-watershed) and one well to the north of State Highway-9 near Sumas (located in Johnson sub-watershed) were detected with excessive levels of EDB.

The major crop in this area is currently raspberries, but strawberries and other field crops such as blueberries are also grown (O'Herron, 1999). The historical use of EDB in these crops may have been the reason for serious EDB contamination of ground water in this area.

Regional-Scale Analysis

EDB was detected in the Bertrand Creek and Meadowdale areas during the period of 1985 to 1994 (see Figures 3.6-3.16). However, serious EDB contamination has been detected mainly in Bertrand Creek area after 1994. In the recent past, EDB has also been detected close to the Willeys Lake Road near the Osoyoos Lake. The high levels of EDB in Bertrand Creek as recent as 1998 and 1999 can be attributed to slow natural attenuation in soil and ground water.

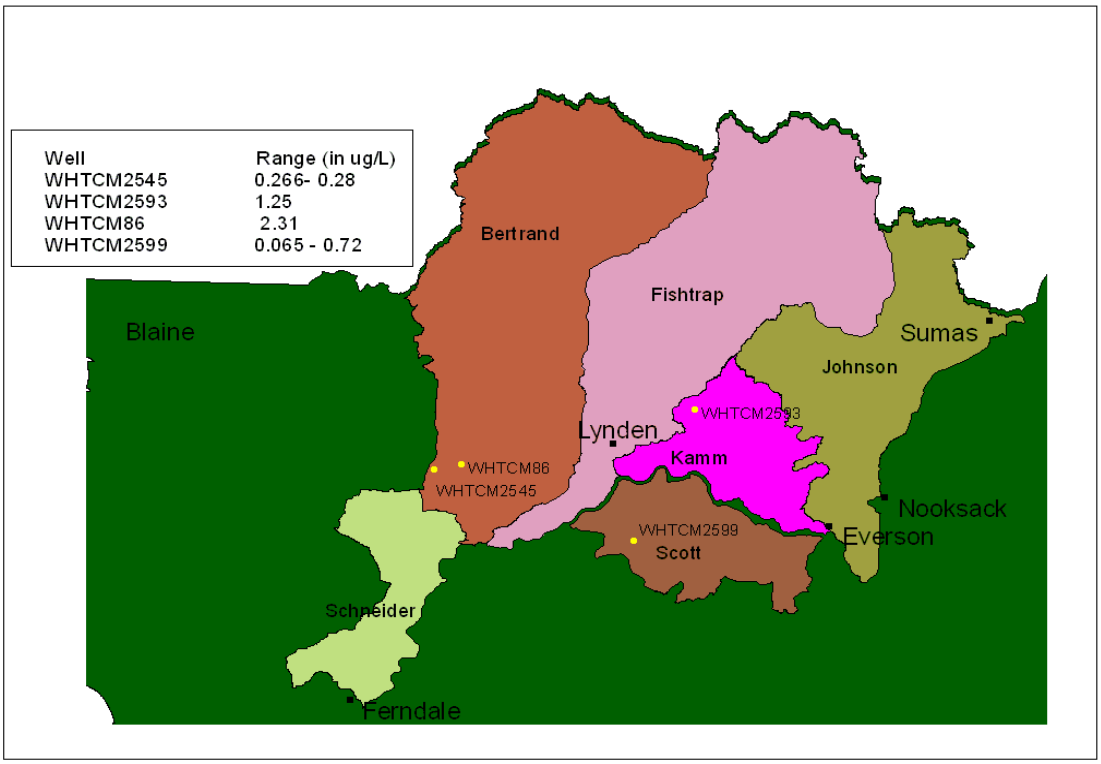


Figure 3.6. Map showing the ground water wells that exceeded MCL in 1984.

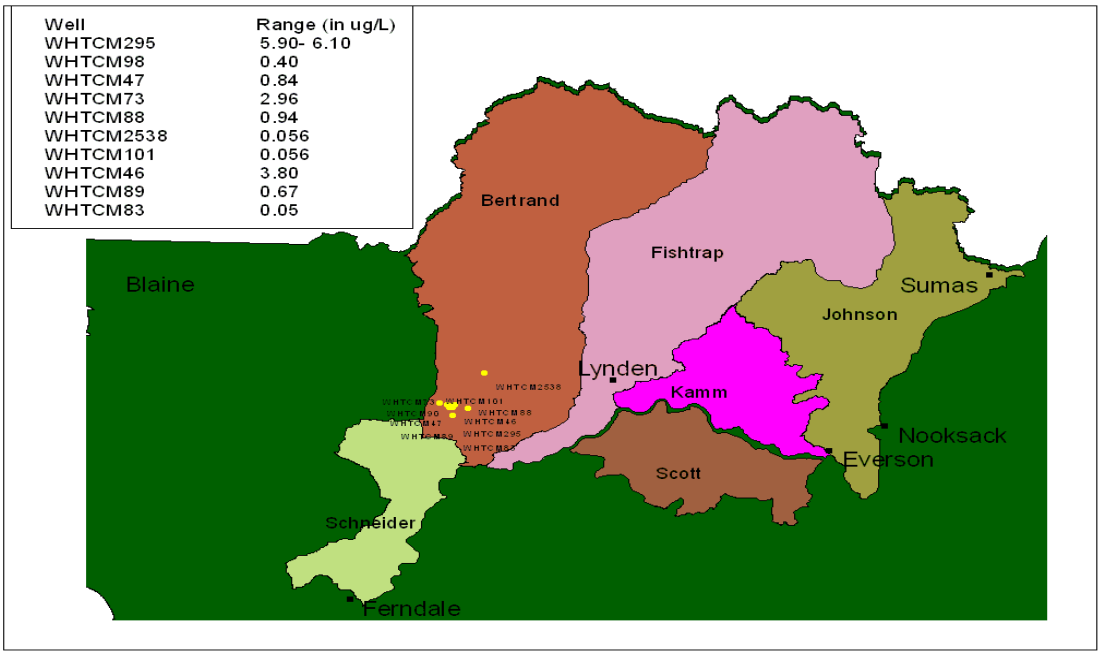


Figure 3.7. Map showing the ground water wells that exceeded MCL in 1986. Note the cluster of wells at the southern part of Bertrand Creek.

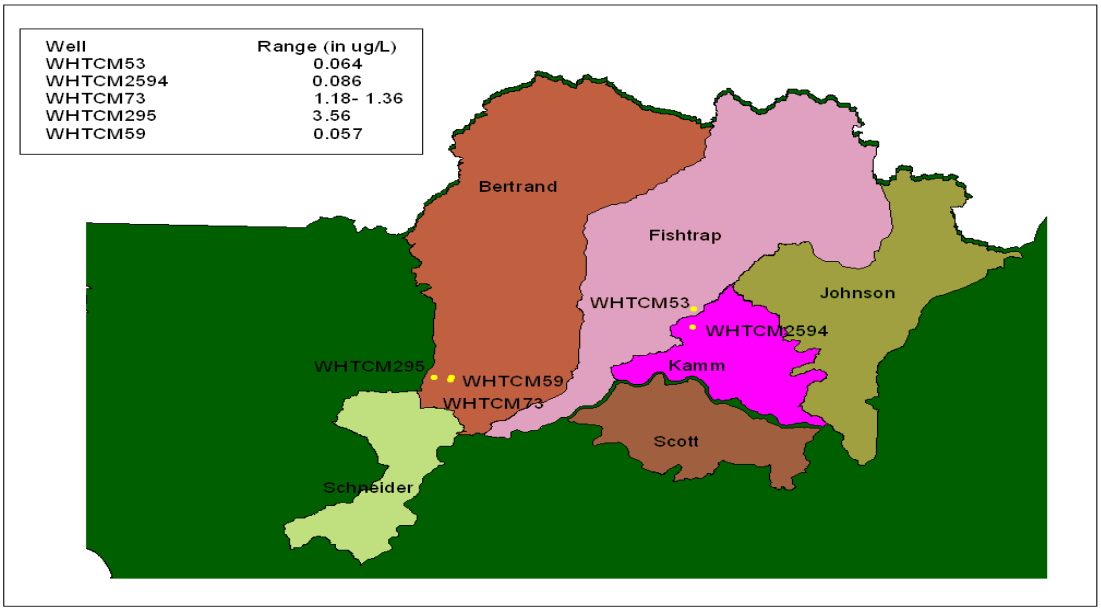


Figure 3.8. Map showing the ground water wells that exceeded MCL in 1987.

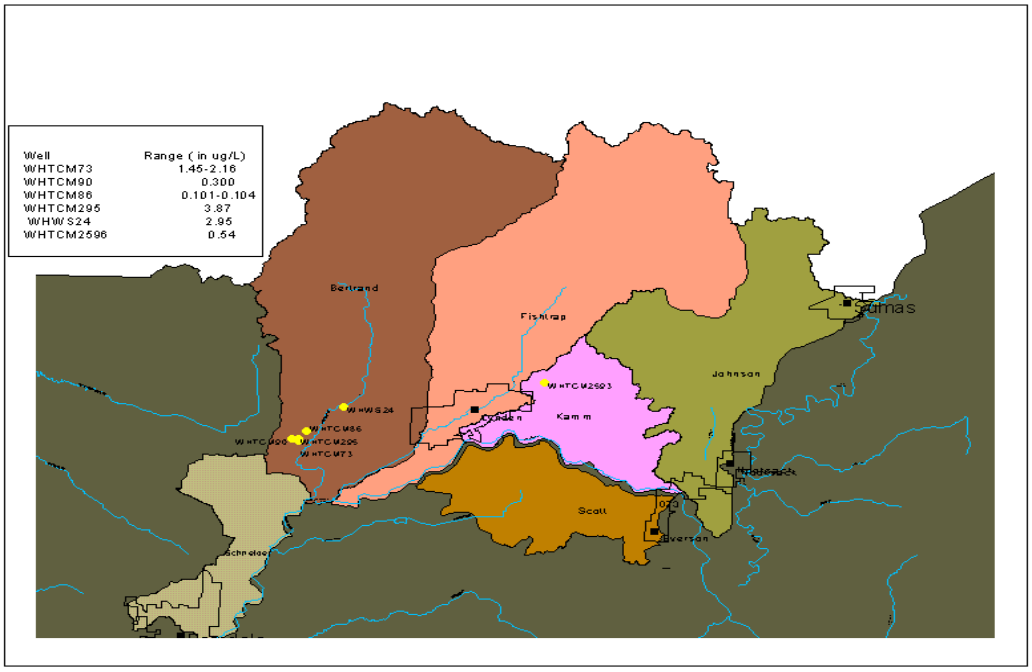


Figure 3.9. Map showing the ground water wells that exceeded MCL in 1988. Note the cluster of wells in the southern part of Bertrand Creek.

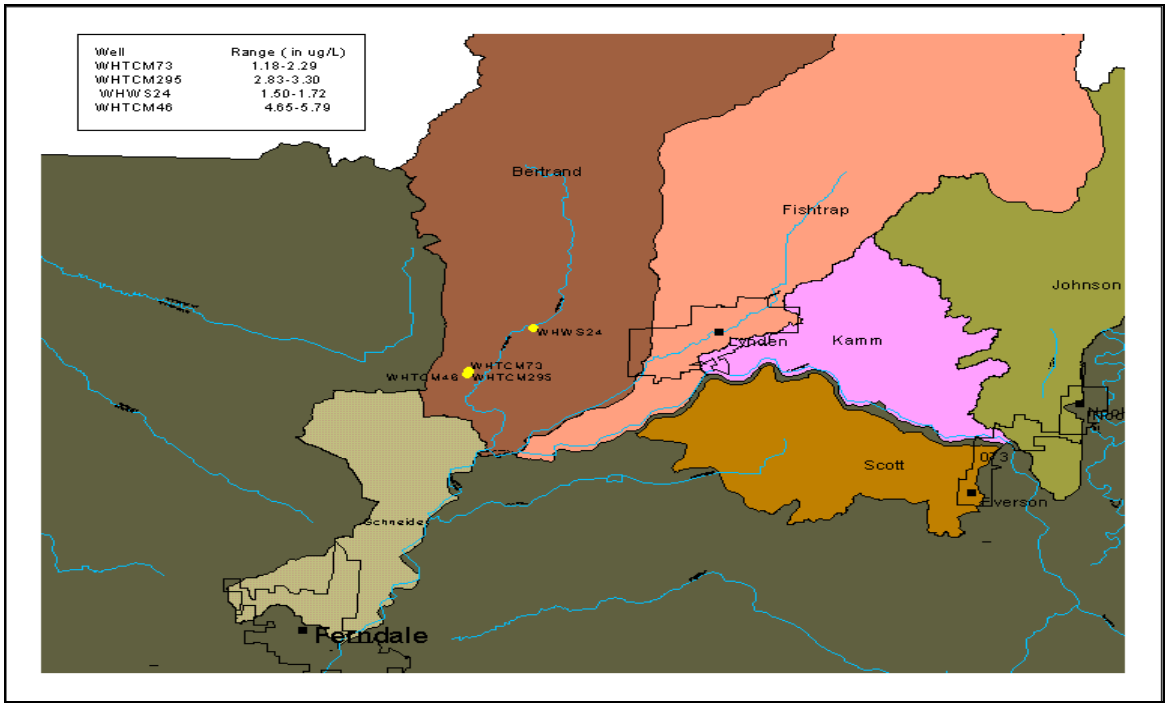


Figure 3.10. Map showing the ground water wells that exceeded MCL in 1989.

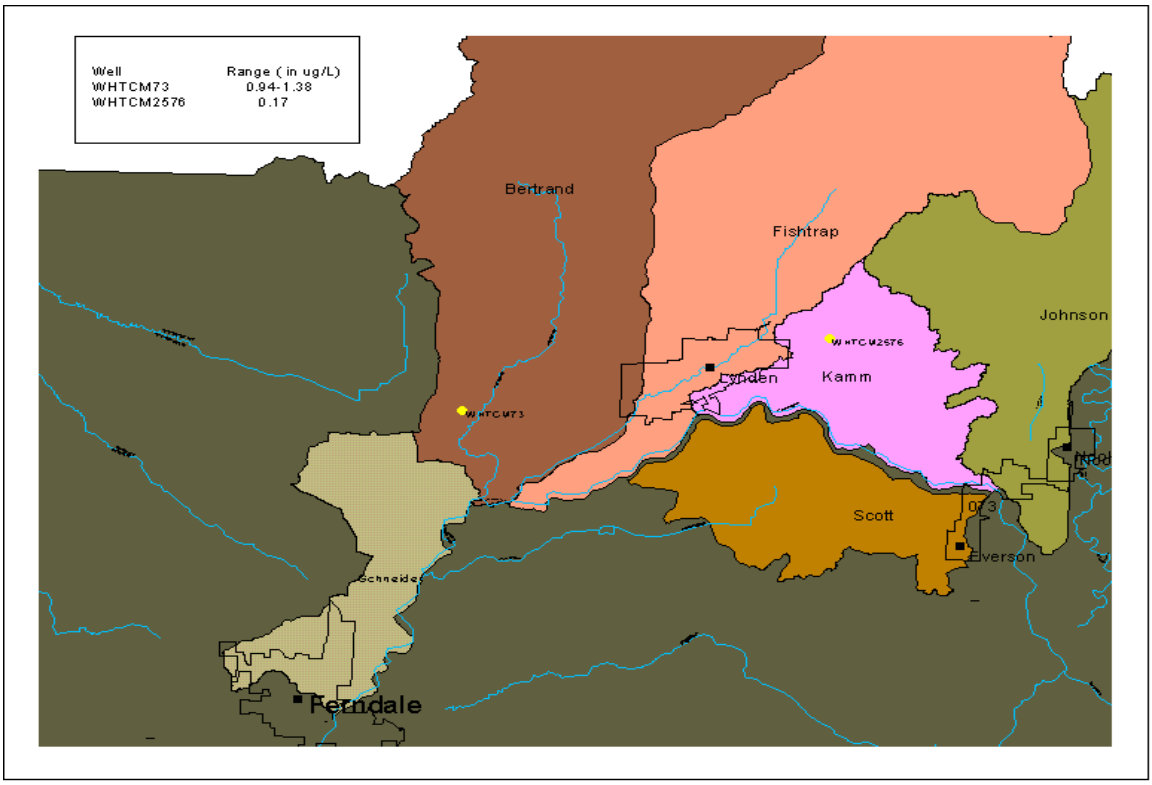


Figure 3.11. Map showing the ground water wells that exceeded MCL in 1990.

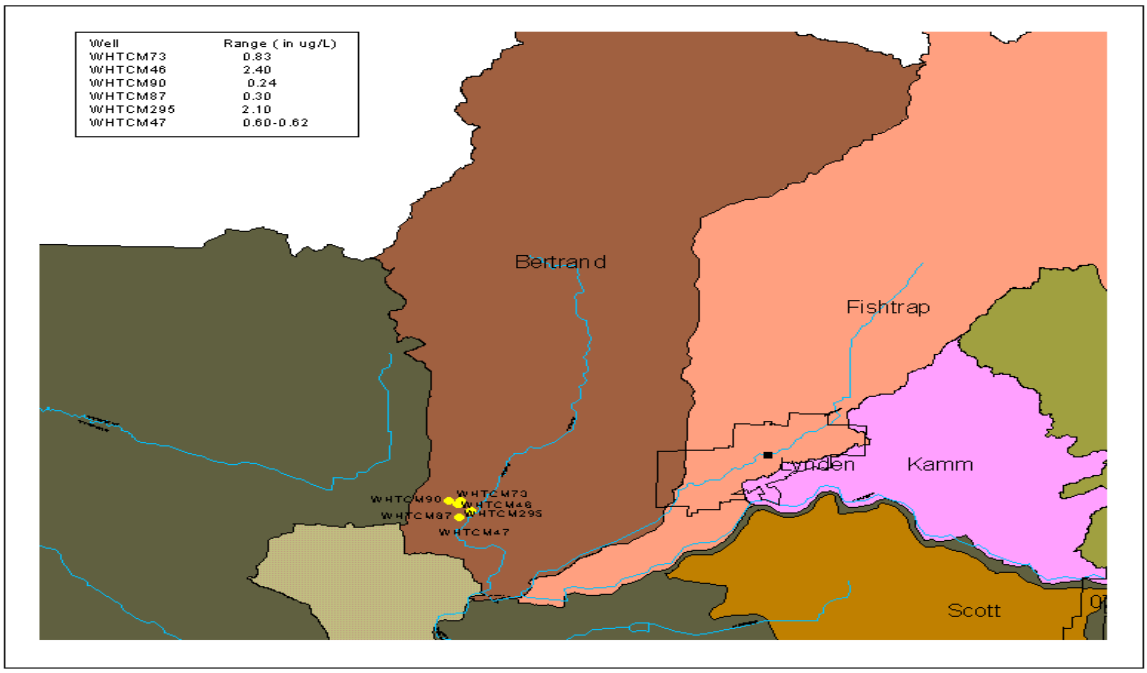


Figure 3.12. Map showing the ground water wells that exceeded MCL in 1991.

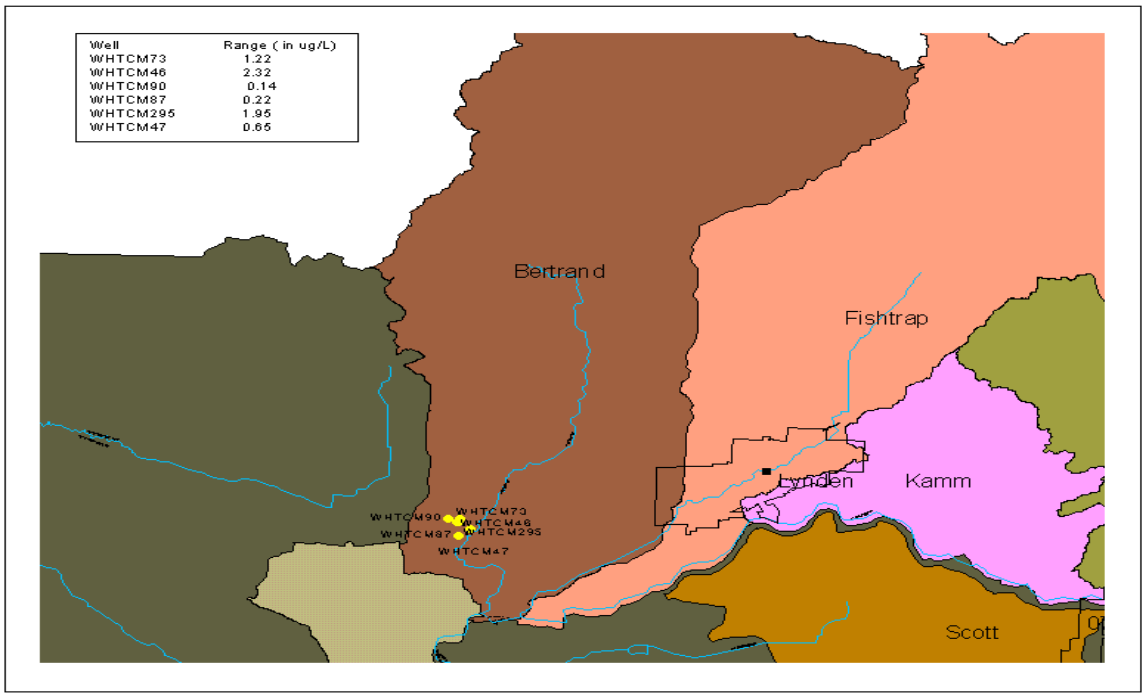


Figure 3.13. Map showing the ground water wells that exceeded MCL in 1992.

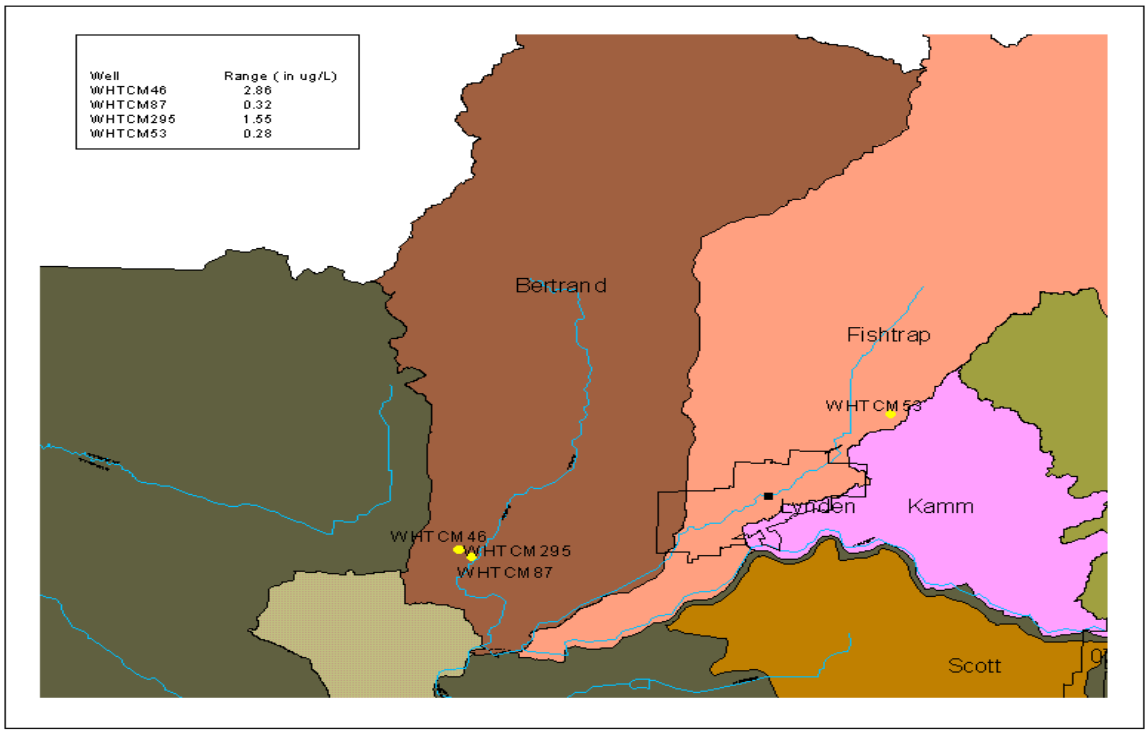


Figure 3.14. Map showing the ground water wells that exceeded MCL in 1993.

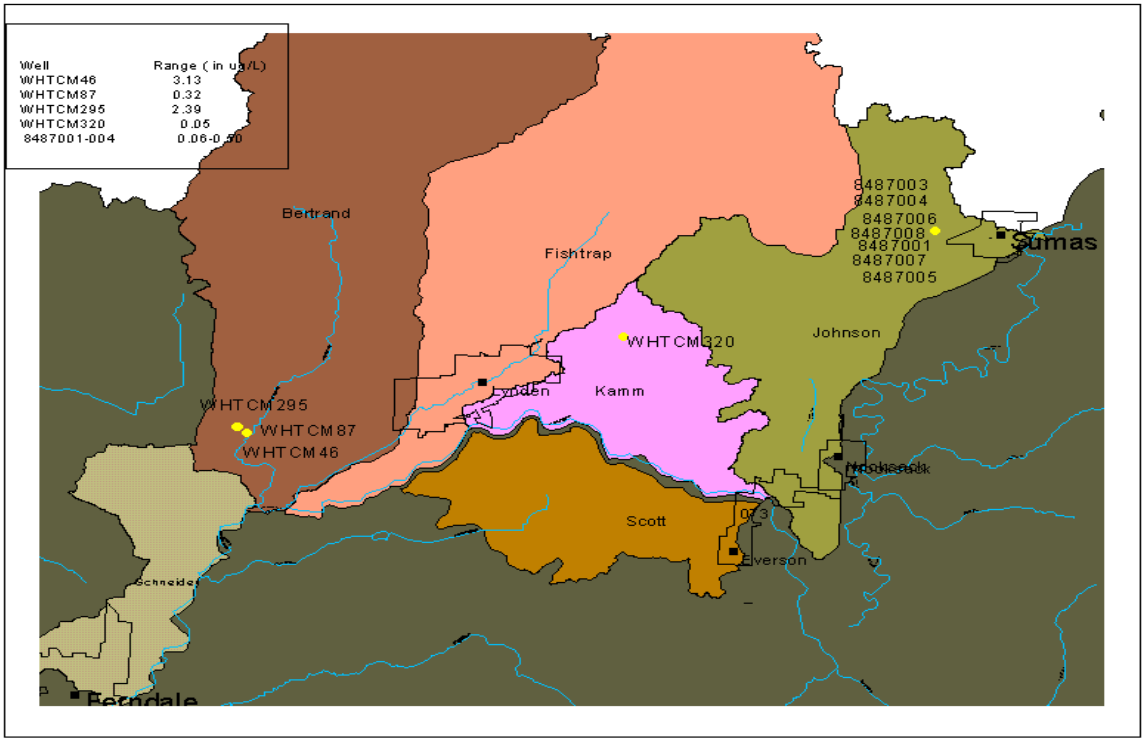


Figure 3.15. Map showing the ground water wells that exceeded MCL in 1994.

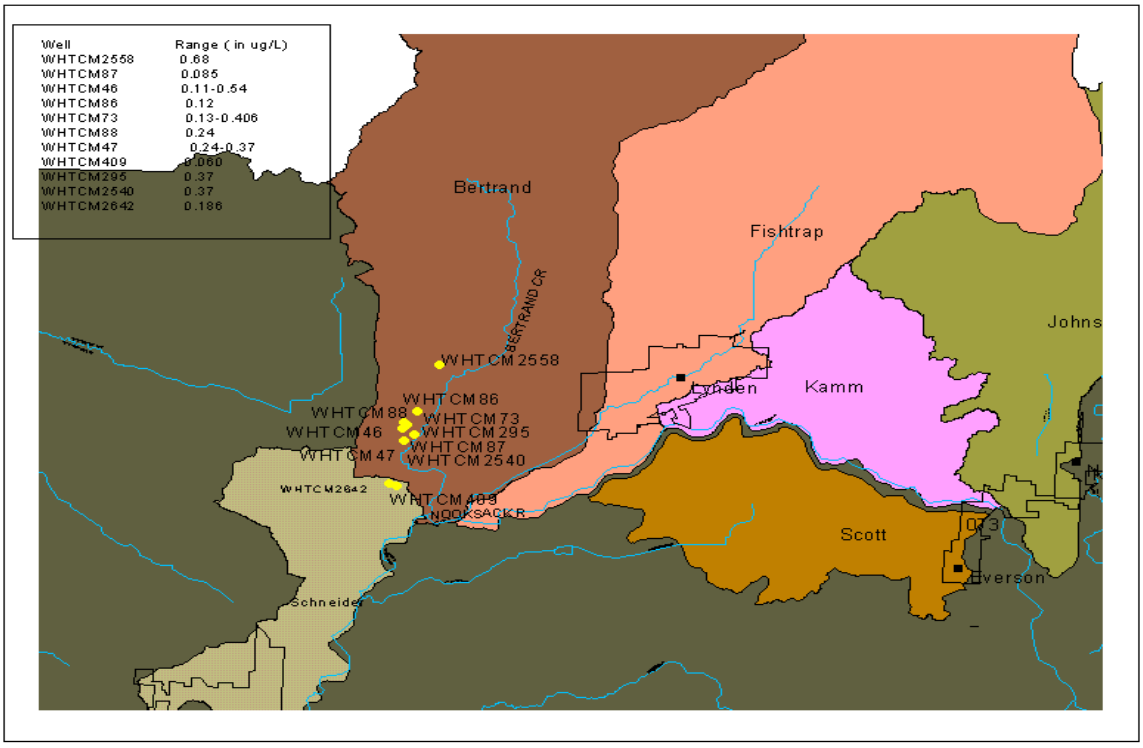


Figure 3.16. Map showing the ground water wells that exceeded MCL in 1998.

Table 3.2. Summary of EDB contamination data in WRIA 1.

Year	Number of Observations	Number of Non-Zero Observations	Number of Observations >MCL	Range (ppb)
1984	8	8	7	0.01-2.31
1986	40	13	12	0-6.1
1987	34	8	6	0.00-3.56
1989	39	23	20	0-5.79
1990	12	6	6	0-1.38
1991	43	8	7	0-2.4
1992	62	14	6	0-2.32
1993	155	7	4	0-2.86
1994	97	26	21	0-3.13
1995	60	8	0	0-0.025
1996	54	0	0	0
1997	22	0	0	0
1998	340	26	15	0-0.68
1999	139	1	1	0-0.186

Table 3.2 provides the summary of the EDB concentrations recorded in the area. The results show that there has been a steady degradation of EDB in ground water ever since it was banned in 1983. Figures 3.6 to 3.16 show a decreasing trend with time in the maximum concentration of EDB in sites that exceeded the MCL.

Potential Fate and Transport Mechanisms

It should be noted that there are two potential mechanisms responsible here for chemical transport, i.e., natural attenuation and advective-dispersive mass transport. Natural attenuation of EDB is possible at local application sites with an average half-life of 100 days. However, if the residual EDB is trapped in low permeable zones that cannot be readily accessed by microbes, then degradation may be small. However, the natural attenuation possibility cannot be ruled out at these sites unless pilot studies are conducted using soil and ground water samples.

The next possibility is advective-dispersive mass transport together with some natural attenuation. The area surrounding WRIA 1 has low to moderate runoff and moderate to high infiltration (refer to Figure 1.13). The maximum hydraulic conductivity is between 6 and 20 inches per hour (refer to Figure 1.14). Ground water flow in the Fishtrap Creek, Bertrand Creek, and Kamm Creek areas is in the south-southwestern direction. The average organic-carbon partitioning coefficient of EDB varies between 14 L/kg to 160 L/kg (http://pested.unl.edu/; http://wizard.arsusda.gov/acsl/ppdb3.html). Using these values, together with an organic-carbon fraction of 0.1% and porosity of 0.25, the retardation coefficient varies between 1.1 and 2.3. Even though the hydraulic conductivity is relatively high, the average retarded pore water velocity is only about 6.5 feet/year with a hydraulic conductivity of 20 inches/hour, hydraulic gradient of 0.001, and a retardation coefficient of 2.

On the other hand, if the average hydraulic gradient is 0.01 which is relatively high, the pore water velocity is about 292 feet/year. If the travel time is 15 years since the ban of EDB, then the travel distance can vary between 435 feet to 4350 feet depending on the hydraulic gradient. Once dispersion is added, the travel distance can easily increase by 20% to 40%. If the latter case with a hydraulic gradient of 0.01 is true, then the possibility of advective-dispersive mass transport with some natural attenuation is valid in this area.

The work of Mitchell et al. (2000) found that ground water flow within the Sumas aquifer is fast due to the high average hydraulic conductivity of 930 feet/day or 465 inches/hour. Using this hydraulic conductivity and an estimated hydraulic gradient of 0.0072, the unretarded pore water velocity was estimated to be 20 feet/day. These recent field observations further support the possibility of advective-dispersive mass transport with some natural attenuation. One contradiction to this possibility is that there was no evidence to support the presence of a soluble plume in ground water. Since the available data are scattered spatially and not available at consistent time intervals, the presence of a soluble plume cannot be verified from existing data.

Based on the previous discussion, EDB may have been transported from the local application sites to the south over the past two decades. On the other hand, if the hydraulic gradient is small and sorption is high, then EDB transport may be insignificant from the application sites. Even in the case of strong advective-dispersive mass transport, EDB may not move further south beyond the Bertrand Creek area because of the presence of low permeable silty clay deposits along the southern boundary of the Sumas-Blaine aquifer. The presence of the silty clay deposits can be observed from the low-permeable conditions present along the southern part of Bertrand Creek (see the soil permeability map in Figure 3.17). Due to these hydrogeologic conditions, EDB may accumulate near the Bertrand Creek area (near Osoyoos Lake). This may be one of the reasons why EDB has been persistently observed as recently as 1999 near Osoyoos Lake (along the Willeys Lake Road) while, at other locations, EDB concentrations are decreasing.

It should be noted that the above discussion refers to possibilities only and the validity of these possibilities needs to be verified through more careful monitoring of the areas for at least a few years. A good approach will be to perform a pilot study of ground water quality of the area using ground water sampling, soil analysis, and fate and transport modeling.

The time series plots of selected wells (see Figure 3.18) are shown in Figure 3.19. The plots show that the northern wells have decreasing trends while the southern wells of Bertrand Creek area have relatively less steep decreasing trends.

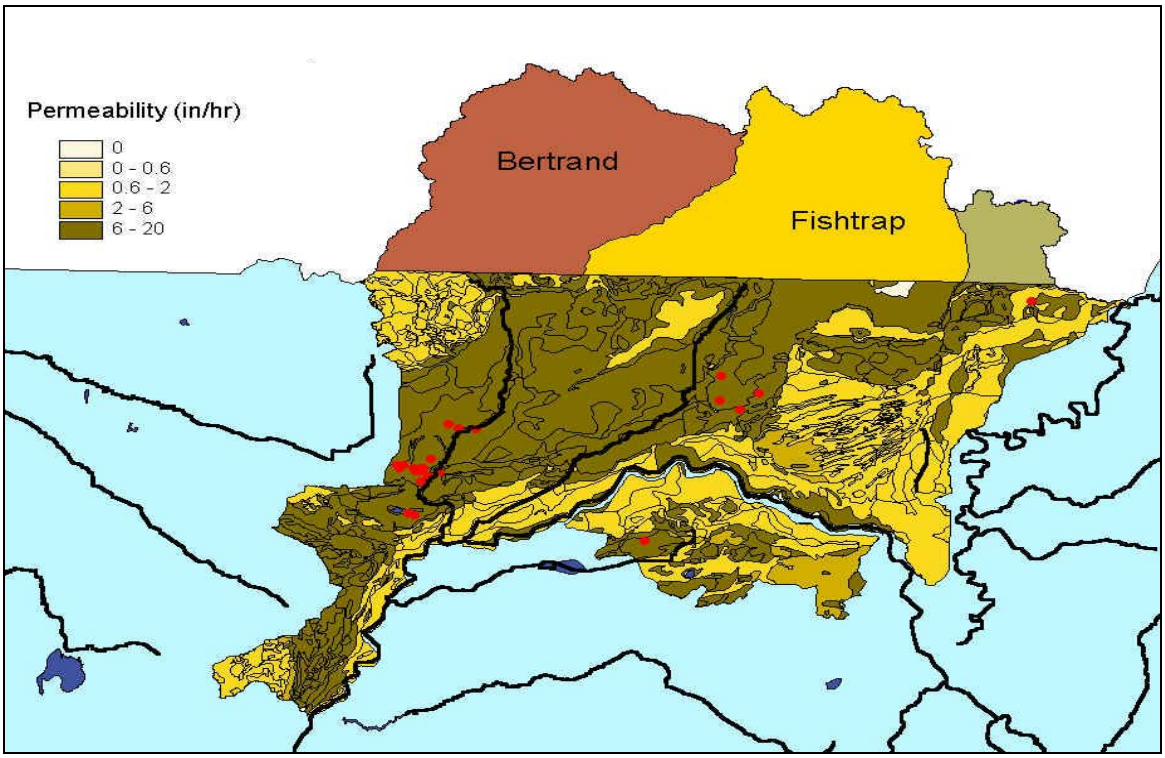


Figure 3.17. Permeability in the area of interest with respect to EDB. Note the low permeable conditions that exist in the area surrounding the ground water wells that exceeded the MCL for EDB.

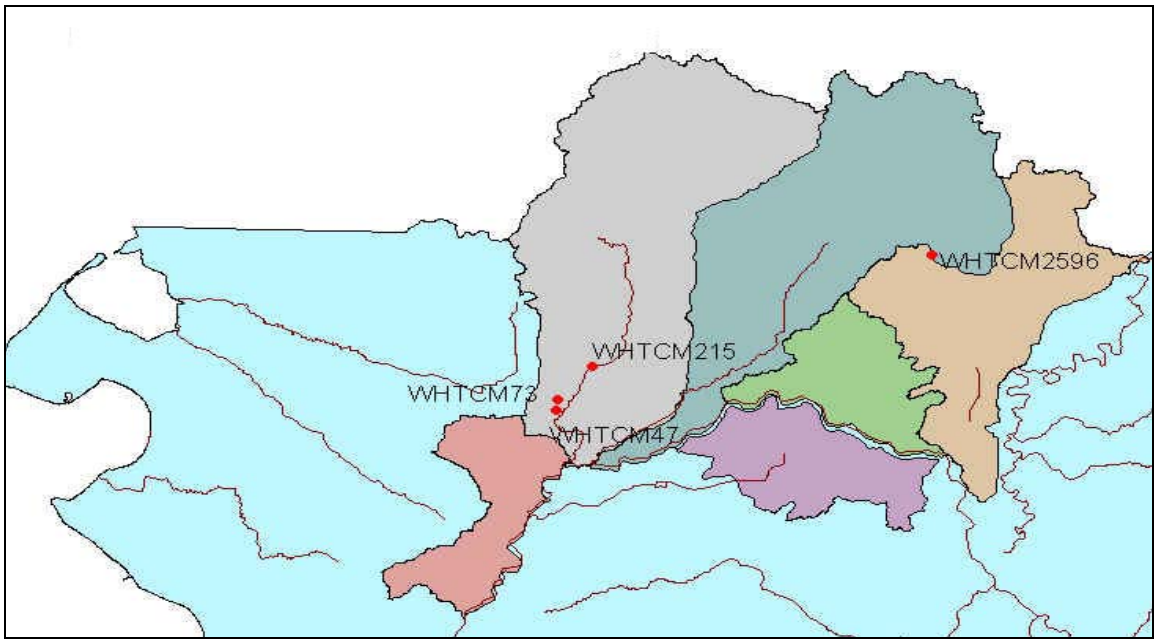


Figure 3.18. Map showing the ground water wells selected for time series analysis.

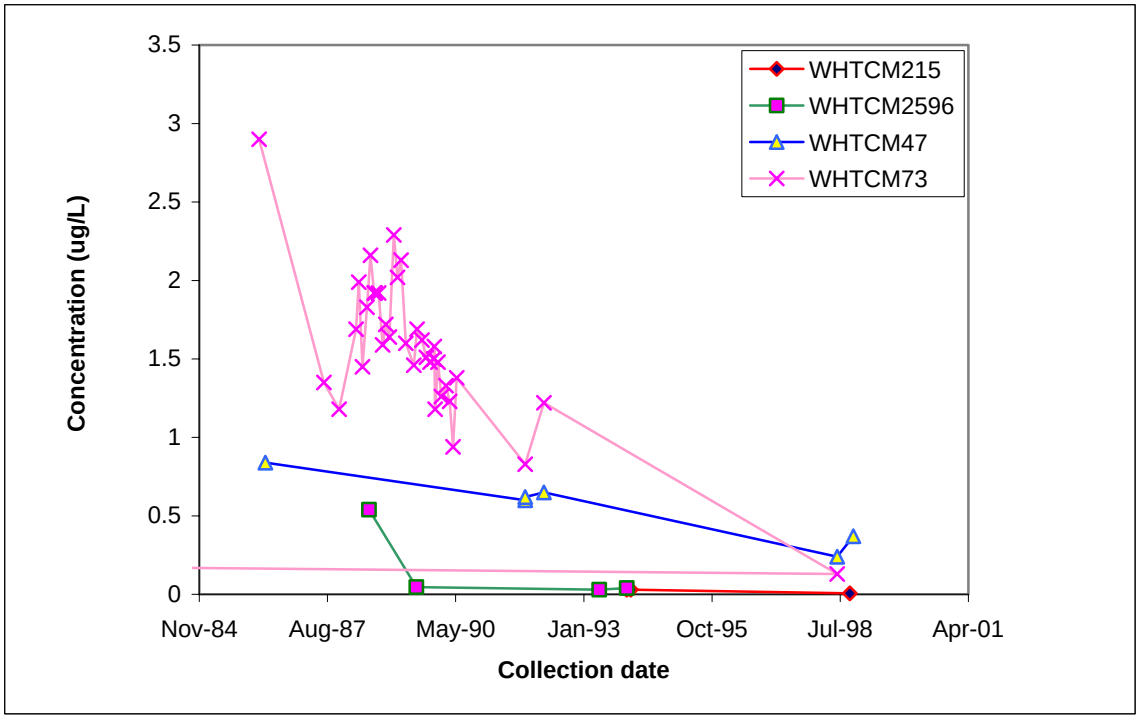


Figure 3.19. Time series of EDB for the selected ground water wells from Figure 3.18.

It is not certain whether any trans-border ground water flow from Canada has contributed to EDB contamination. Whatever EDB that may have contaminated ground water due to trans-border flow from Canada should have transported through the aquifer and not through runoff into the south-flowing streams.

As far as EDB contamination of ground water in WRIA 1 is concerned, Bertrand Creek and Meadowdale areas (shown in Figure 3.1) are of serious concern. A previous study by US EPA in the same area also concluded that pesticides are present above the drinking water standards (ATSDR, 2000). EDB contamination has been observed in Bertrand Creek area since 1984. Considering the stability of EDB in ground water and the fact that a concentration of 0.186 ppb was observed in the Bertrand Creek area as recently as 1999, EDB still poses a substantial concern to the residents of Bertrand Creek area.

3.2 <u>1,2-Dichloropropane</u>

1,2-dichloropropane (1,2-DCP) is now used in the United States only in research and industry. Before the early 1980s, 1,2-DCP was used in farming as a soil fumigant and was found in some paint strippers, varnishes, and furniture finish removers. 1,2-DCP was used extensively in WRIA 1 as a soil fumigant for nematodes on berries and other crops. It was banned in 1988 by US EPA and in 1985 by Canadian environmental authorities for possible carcinogenic effects. Since 1983, 1,3-dichloropropane (1,3-DCP) has replaced EDB use. 1,2-

dichloropropane has been associated in the past with 1,3-DCP use (Erickson, 1998). 1,2-dichloropropane was present in 1,3-DCP products sold by Dow Chemicals before 1962 and by Shell after 1986. The current level of 1,2-DCP in dichloropropane formulations is 0.05% or less (Erickson, 1998). 1,3-DCP is still being used under the restricted use (RU) class.

The trade names and synonyms of 1,2-DCP are propylene dichloride, nematox, vidden D, and dowfume EB-5.

Most of the 1,2-DCP released into the environment through various applications finally enters the air or ground water. When applied to soil in one experiment, all but 1% dispersed in 10 days. Breakdown in both the air and ground water is slow. The half-life of 1,2-DCP in air is not known exactly, but it is longer than 23 days, which means that 1,2-DCP can spread to areas far from where it is released. In ground water, the half-life of 1,2-DCP is estimated to be between 6 months and 2 years with an average of about 700 days.

1,2-DCP was extensively used as a soil fumigant in WRIA 1 before it was banned from agricultural use in 1988 by the US EPA and in 1985 by the Canadian Government. 1,2-DCP is a minor component of 1,3-DCP (about 0.05%), which is still being used under restricted use class. 1,2-DCP was more extensively used in the Canadian portion of WRIA 1. Approximately 10,000 Kg of 1,2-DCP was sold in the Abbotsford area in 1984 (Liebscher and others, 1992).

3.2.1 Health Effects

Drinking of 1,2-DCP contaminated water by humans has produced poisoning at high levels of exposure. Health effects include dizziness, headache, nausea, injury to the liver and kidneys, anemia, coma, and ultimately death (http://www.eco-usa.net/toxics/12dcp.html). Breathing high levels of 1,2-DCP by humans, for example, vapor from cleaning solutions, produces similar effects. No reports have been made of any health effects in humans following low-level exposure to1,2-DCP over short or long periods.

In animal experiments, low amounts of 1,2-DCP inhaled over short and long periods result in damage to the liver, kidney, and respiratory systems, while high amounts resulted in death. Short-term exposure to high levels of vapors also causes irritation to eyes and throat. When 1,2-DCP is given by mouth to animals over short or long periods, damage to the liver and kidneys is seen at low doses, and death occurs at high doses.

1,2-DCP inhaled or eaten over a short period has not been reported to produce cancer in humans, but long-term exposure by mouth in animals has produced evidence of liver cancer in mice and breast cancer in female rats. The significance of the animal cancer studies to humans is not well understood. Irritation of the skin after contact with 1,2-DCP has been seen in both humans and rabbits. 1,2-DCP has not been shown to cause birth defects in humans or animals, but a delay in the growth of bones has been seen in fetal rats following exposure of the mother rats.

The drinking water standard or the MCL of 1,2-DCP is 5 ppb.

3.2.2 Analysis of 1,2-DCP Contamination

1,2-DCP was detected throughout the LENS area, but ground water wells with concentrations exceeding the MCL were found in the southern part of Bertrand Creek, to the northeast of Lynden, and to the east of Sumas.

Figure 3.20 shows the wells and areas where the MCL was exceeded in any year over the past decade.

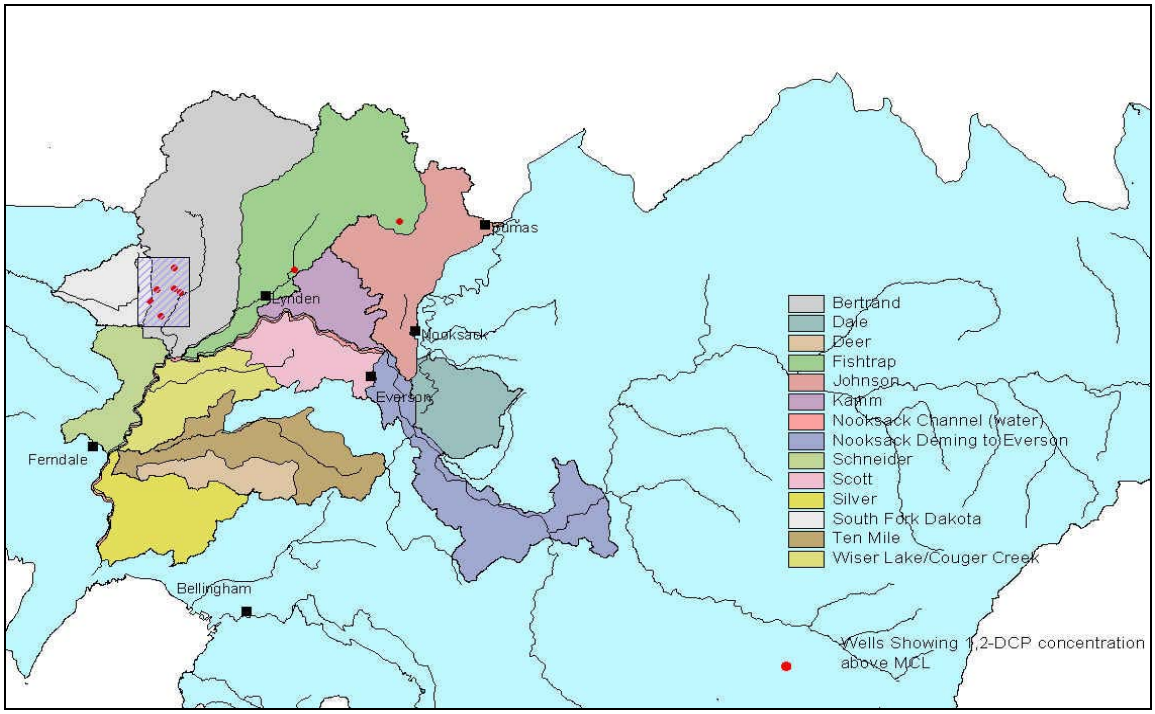


Figure 3.20. Map showing the areas of concern and wells due to 1,2-DCP contamination.

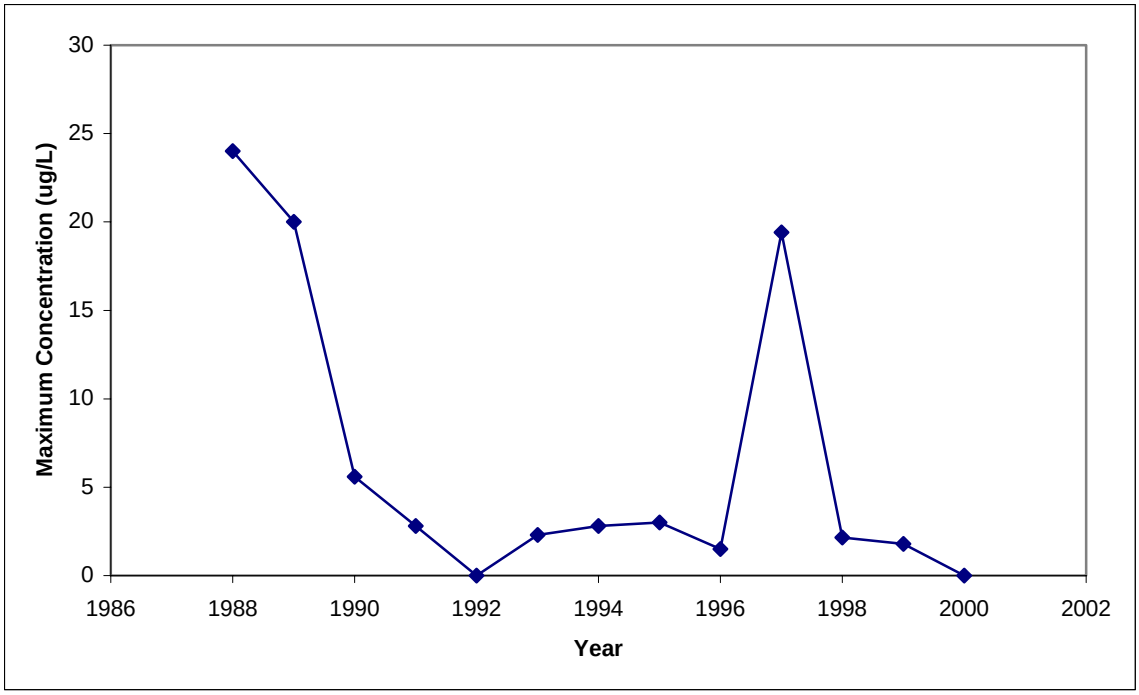


Figure 3.21. Time series of maximum concentration of 1,2-DCP in WRIA 1.

While the MCL of 1,2-DCP is 5 ppb, concentrations as high as 24 ppb have been detected in the area. Figure 3.21 shows the time series of maximum concentration observed over the years in WRIA 1.

Impacted Sub-Watersheds

Based on the ground water quality data available from WRIA 1, the observed maximum ground water concentration of 1,2-DCP was compared against the MCL. The results of this comparison was used to identify the sub-watersheds affected by 1,2-DCP, and these are as follows:

- Bertrand Creek
- Fishtrap Creek
- Kamm Creek
- Johnson Creek
- Deer
- Dale
- Scott
- Silver
- South Dakota Creek
- Schneider
- Ten Mile
- Wiser Lake
- Nooksack Deming to Everson
- Nooksack Channel

Table 3.3 provides additional information related to the affected sub-watersheds for years that data are available. Figure 3.22 shows the location of sub-watersheds affected by 1,2-DCP contamination.

Table 3.3. Details of sub-watersheds that exceeded MCL for 1,2-DCP.

Year	Sub-Watershed	Maximum Concentration (ppb)	Source
1988	Bertrand Creek	24	Washington Department of Ecology
	South Dakota Creek close to the Bertrand Creek sub-watershed	14	Washington Department of Ecology
1989	Bertrand Creek	20	Washington Department of Ecology
	South Dakota Creek close to the Bertrand Creek sub-watershed	8.8	Washington Department of Ecology
1990	Bertrand Creek	5.6	USGS
1997	Bertrand Creek	19.4	USGS
	Fishtrap Creek	11.4	USGS

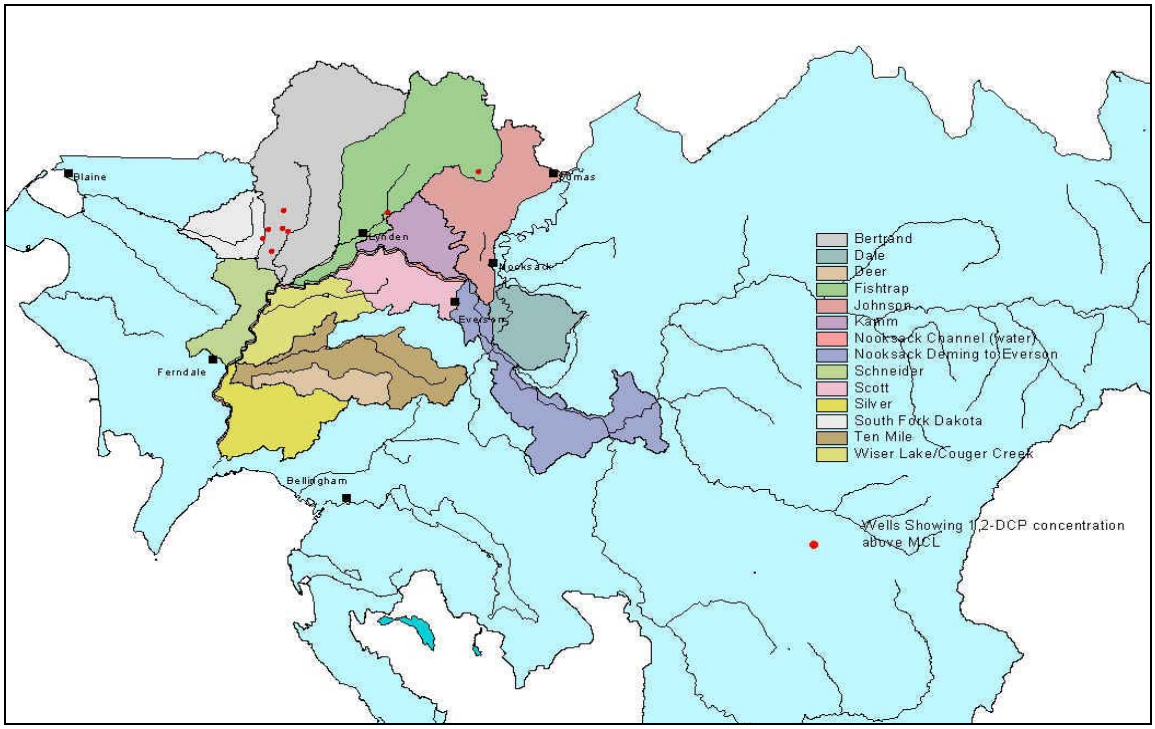


Figure 3.22. Map showing sub-watersheds affected by contamination of 1,2-DCP.

Although as many as 14 sub-watersheds in WRIA 1 may have been affected by 1,2-DCP at least one year, the wells that exceeded MCL tend to cluster near the southern part of Bertrand Creek, to the northeast of Lynden, and to the east of Sumas. Therefore, just as in EDB, the southern portion of Bertrand Creek holds significant importance in the overall analysis and needs to be studied at a more local-scale.

Bertrand Creek Area

A number of wells exceeding MCL for 1,2-DCP have been observed in the southern portion of Bertrand Creek. Figure 3.23 shows the area with the locations of ground water wells that exceeded MCL. These wells are located near the Loomis Trail Road, Birch Bay-Lynden Road, Bob Hall Road, Weidkamp Road, or Glendale Road. It may be recollected that it was the same area that evoked interest with EDB as well (refer to Figure 3.4). The trend of degradation of 1,2-DCP in general is similar to that of EDB.

A well located to the east of the Bertrand Creek area and near the Loomis Trail Road (shown in Figure 3.23) showed 1,2-DCP at 11.4 ppb in 1997. Based on the available information from the area, the increase of 1,2-DCP is most probably due to locally accumulated and distributed mass rather than any regional-scale transport.

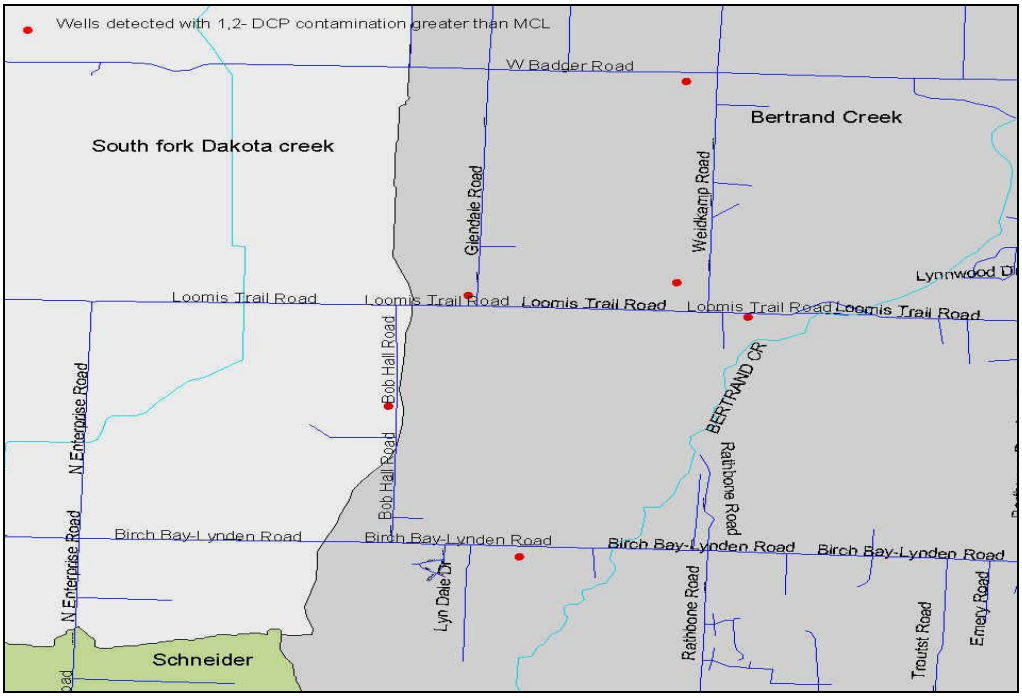


Figure 3.23. Map showing the Bertrand Creek area and the wells exceeding the MCL of 1,2-DCP between 1988 and 1997.

Other than the Bertrand Creek area, concentrations greater than MCL were observed to the north of City of Lynden and to the east of City of Sumas. In 1997, a high concentration of 1,2-DCP was observed in one of the wells in the area. The well was located along State highway 546 just to the east of the intersection of Fishtrap River with the Highway (see Figure 3.24). Again, a possible reason for this sudden increase may be due to local accumulation of 1,2-DCP rather than regional-scale transport.

Regional-Scale Analysis

Concentration of 1,2-DCP in WRIA 1 has generally decreased except in 1997 at two wells. This observation is based on the results shown in Table 3.4 and Figure 3.21. A strong reason for this decreasing trend may be the degradation of 1,2-DCP in ground water. While the fate and transport of 1,2-DCP are similar to that of EDB in Bertrand and Fishtrap Creeks, 1,2-DCP contamination in other sub-watersheds has also decreased with time. It should be noted that the average biological half-life of 1,2-DCP is about 700 days which is almost seven fold higher than EDB (http://pested.unl.edu/; http://wizard.arsusda.gov/acsl/ppdb3.html). 1,2-DCP exhibits higher resistance to degradation than EDB and therefore, the potential for advective-dispersive mass transport is high.

Concentrations of 1,2-DCP in Whatcom County may have been influenced by some trans-border ground water flow that occurs from north to south except in the Johnson sub-watershed. In the Johnson sub-watershed, 1,2-DCP may tend to move towards the Fraser River located in Canada because the local hydraulic gradient is from south to north.

Figure 3.24. Map showing the well recording abrupt increase of 1,2-DCP concentration in 1997.

Table 3.4. Summary data of 1,2-DCP concentrations in WRIA 1.

Year	Number of Observations	Number of Non-Zero Observations	Number of Observations >MCL	Range (ppb)
1988	45	28	5	0-24
1989	49	12	6	0-20
1990	72	25	2	0-5.6
1991	237	20	0	0-2.8
1992	139	0	0	0
1993	196	8	0	0-1.3
1994	106	2	0	0-3.5
1995	100	15	0	0-3.9
1996	92	22	0	0-1.5
1997	167	51	2	0-19.4
1998	117	31	0	0-2.15
1999	161	0	0	0

Figure 3.26 shows the time-series plots of four wells shown in Figure 3.25. Wells 6835006 and 6200006 are the wells that have shown the high 1,2-DCP contamination in 1997. The results show that the wells in the north part of the area have lower concentrations while the wells in the south have higher concentrations especially between 1997 and 1998. Again, the monitoring is not consistent and therefore, no strong conclusion can be drawn from these data. It may be possible that transport of 1,2-DCP occurred along the ground water gradient together with some natural attenuation in a manner similar to EDB, or the results may simply be showing local-effects at different application sites with some natural attenuation over the years.

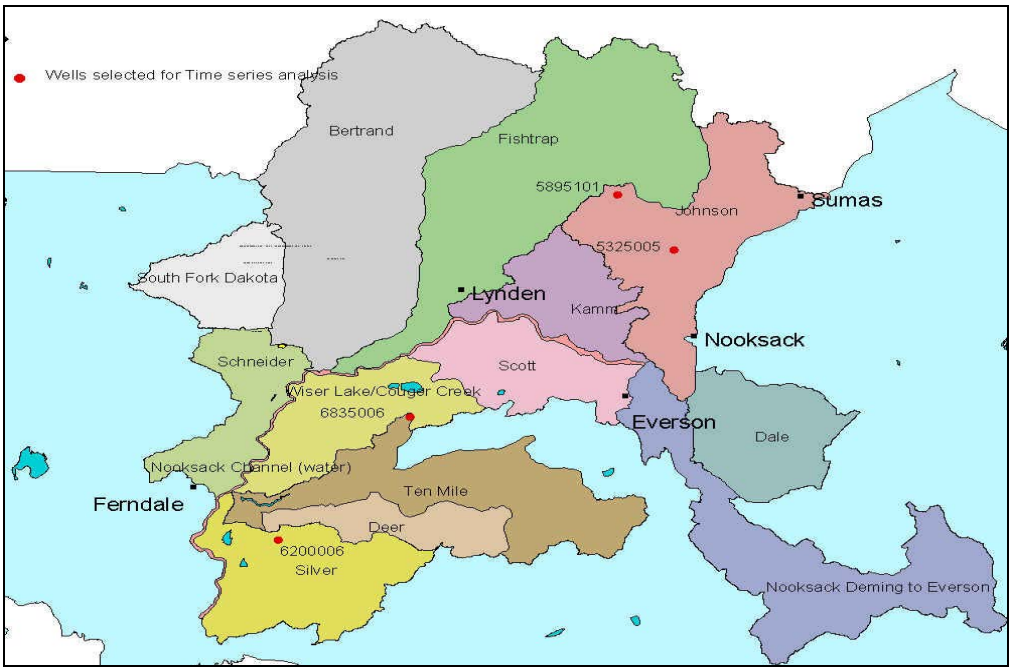


Figure 3.25. Map showing ground water wells selected for time series analysis.

In general, 1,2-DCP may not be a concern for most parts of WRIA 1. However, the threat of 1,2-DCP as a pollutant in ground water in Bertrand Creek and Fishtrap Creek sub-basins still exists. This is evident from the data where the concentrations in ground water were as high as 19.4 ppb in Bertrand Creek and 11.4 ppb in Fishtrap Creek sub-basins as early as 1997.

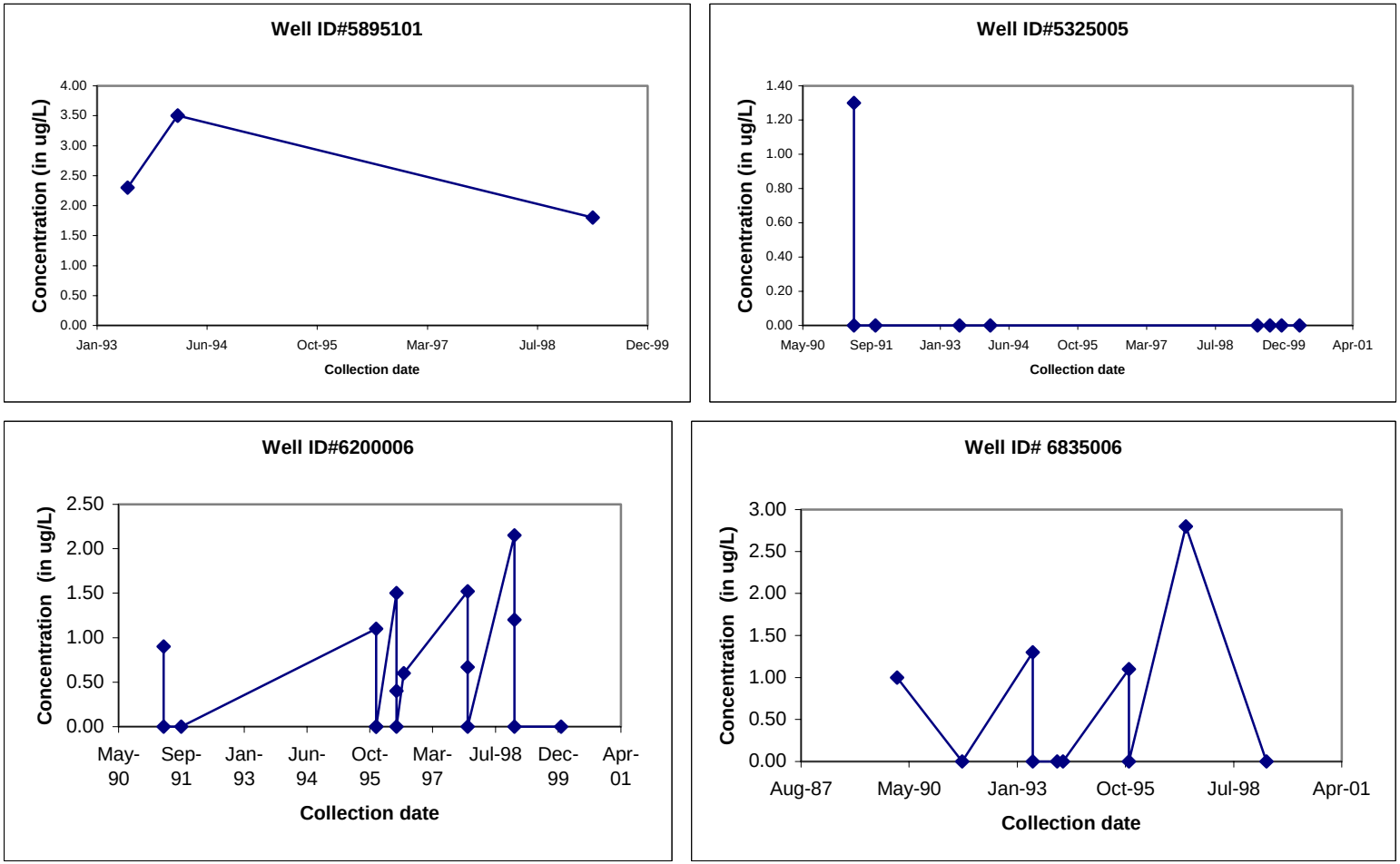


Figure 3.26. Time series of 1,2-DCP at selected wells shown in Figure 3.25.

3.3 1,2-Dibromo-3-chloropropane

Dibromochloropropane or 1,2-dibromo-3-chloropropane (DBCP) is primarily a soil fumigant, which was used in crops such as berries, pineapples, etc. The chemical name of DBCP is 1,2-dibromo-3-chloropropane. Prior to its ban from agricultural use except for pineapples by US EPA in 1979 (http://www.epa.gov/oppfead1/international/piclist.htm), it was used in conjunction with other soil fumigants such as EDB, 1,2-DCP, etc. The trade names and synonyms of DBCP are BBC 12, fumagon, fumazone, nemabrom, nemafum, nemagon, nemanax, nemapaz, nemaset, nemazon, gro-tone nematode, and durham nematocide.

US EPA has found DBCP to cause the following health effects when exposed to levels above the MCL for relatively short periods of time (http://www.epa.gov/ttn/uatw/hlthef/dibromo-.html): kidney and liver damage, and atrophy of the testes. DBCP has the potential to cause kidney damage and anti-fertility, and cancer from lifetime exposure to levels above the MCL of 0.2 ppb. DBCP released to soil will most likely volatilize or leach to ground water. Biodegradation of DBCP is slow by comparison. Once in the atmosphere, DBCP is expected to breakdown fairly quick by sunlight. DBCP is not likely to accumulate in aquatic phase.

3.3.1 Analysis of DBCP Contamination

The use of DBCP in WRIA 1 is not well-known. However, heavy usage of DBCP prior to 1979 in the area may be possible as DBCP was commonly used on berries and seed potatoes. DBCP contamination in WRIA 1 is present in the southern part of Bertrand Creek. Ground water wells exceeding the MCL were located in this area as in the case with EDB contamination and Figure 3.27 shows this area of interest. Figure 3.28 shows the time series plots of maximum concentration of DBCP at selected wells from this area. The results show a decreasing trend of DBCP with time indicating degradation of DBCP in ground water.

Impacted Sub-Watersheds

Based on the available ground water quality data on DBCP and in comparison to the MCL, it was found that Bertrand Creek and South Fork Dakota Creek sub-watersheds have been affected by DBCP. Although only two sub-watersheds are affected, it was found that all heavily DBCP contaminated wells are clustered in an area located on the boundary of the two watersheds near the southern part of Bertrand Creek. Figure 3.29 shows the two sub-watersheds affected by DBCP contamination.

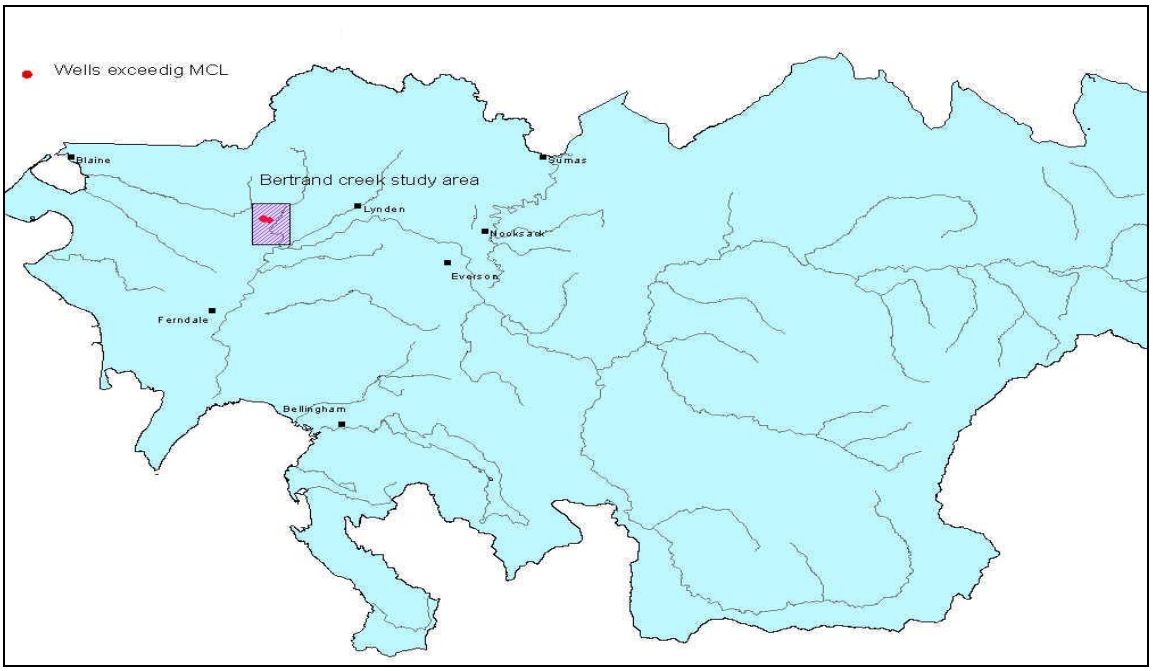


Figure 3.27. Map showing the area of concern for DBCP contamination.

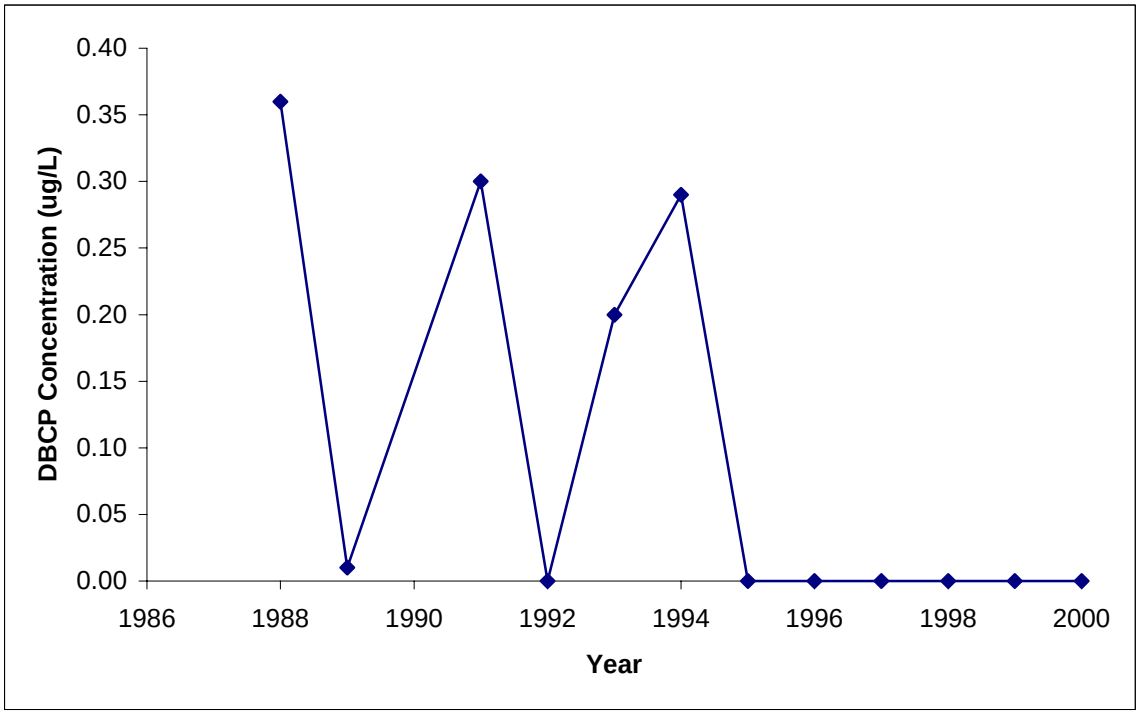


Figure 3.28. Time series of maximum concentration of DBCP in the area of interest shown in Figure 3.27.

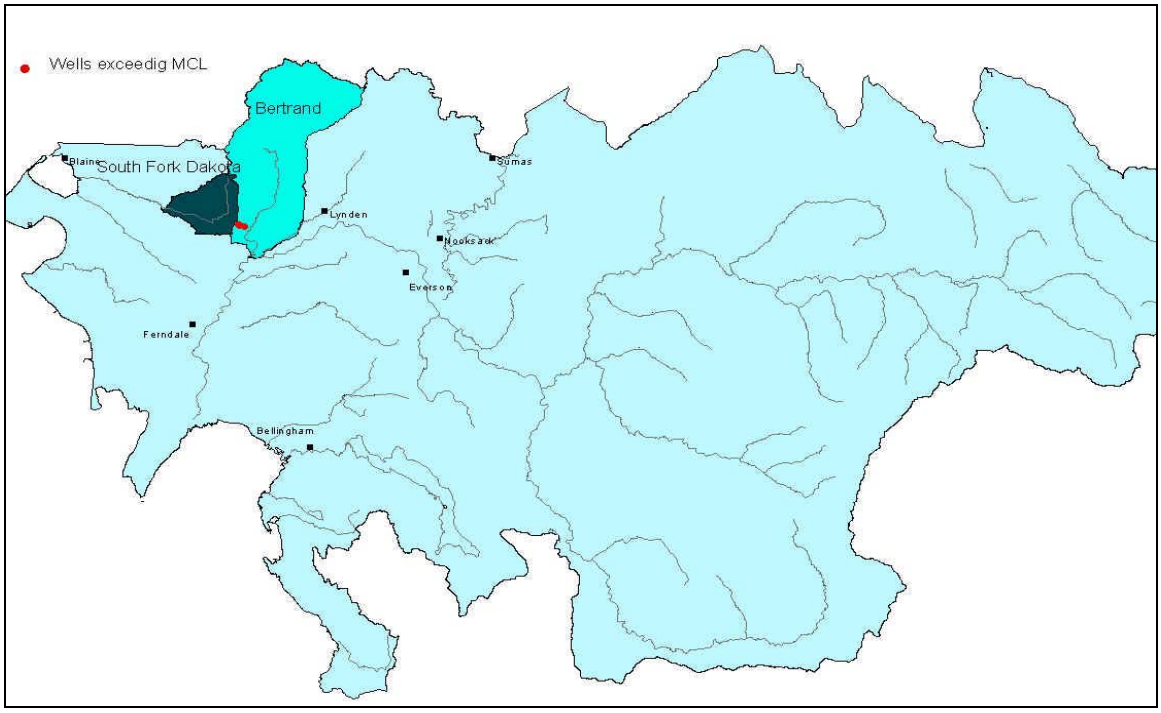


Figure 3.29. Two sub-watersheds affected by DBCP contamination.

Table 3.5 shows the summary data of DBCP contamination for each year and the respective maximum concentration observed. Figure 3.30 shows the area with the locations of ground water wells that exceeded MCL. These wells are located along the Birch Bay-Lynden Road to the east of Bob Hall intersection. It is interesting to note that at least one of the areas of interest with EDB, 1,2-DCP and DBCP is located in the southern part of Bertrand Creek (refer to Figures 3.4, 3.23, and 3.30). This observation may be due to the unique hydrogeologic features that may have caused the pesticides applied in the Sumas-Blaine aquifer to accumulate in the southern part of Bertrand Creek. It should also be noted that DBCP has not been detected after 1994.

In summary, high concentrations of DBCP above 0.2 ppb have been observed up to early 1990's. However, the available ground water quality data suggest that DBCP has decreased from ground water since that time.

Table 3.5. Summary DBCP contamination data of WRIA 1.

Year	Number of Observations	Number of Non-Zero Observations	Number of Observations >MCL	Range (ppb)
1988	31	29	1	0-0.36
1989	9	2	0	0-0.01
1990	-	-	-	-
1991	40	4	2	0-0.3
1992	39	0	0	0
1993	153	1	1	0-0.2
1994	80	3	1	0-0.29
1995	42	0	0	0
1996	55	0	0	0
1997	21	0	0	0
1998	217	2	0	0
1999	142	0	0	0
2000	42	0	0	0

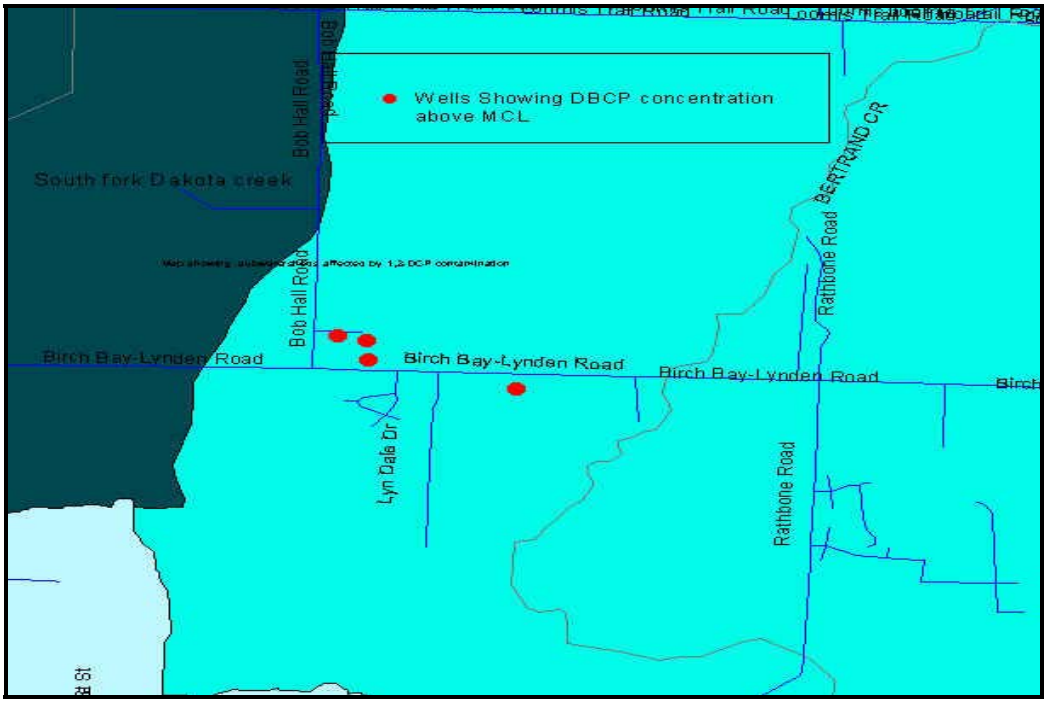


Figure 3.30. Map showing the Bertrand Creek area affected by DBCP contamination.

3.4 Other Pesticides

More than 12 pesticides have been detected in the past in WRIA 1, primarily in the Sumas-Blaine aquifer. These pesticides include banned pesticides such as EDB, 1,2-DCP, DBCP, etc. The presence of these pesticides is not surprising as the Sumas-Blaine aquifer is used heavily for commercial agriculture. Also, the direction of ground water flow across the international boundary is from north to south, which may cause some transport of pesticides used in Canada to the US part of the Sumas-Blaine aquifer. The pesticide contamination is of concern in the Sumas-Blaine aquifer only and not in other parts of WRIA 1. Tables 3.6 through 3.14 summarize the occurrences of other pesticides in WRIA 1. Some pesticides were detected only in 1988 and Table 3.6 provides the details of pesticides that were detected in 1988.

It should be noted that the available ground water quality data from WRIA 1 have not been collected through a well-focused spatially distributed monitoring scheme; rather the data collected are typically biased towards areas with heavy agricultural practices and known previous contamination. With these limitations in mind, the existing data shown in Tables 3.6 through 3.14 indicate that pesticides other than EDB, 1,2-DCP, and DBCP have not exceeded the respective MCLs. However, this observation does not support the notion that ground water in WRIA 1 is free of serious concerns with pesticides other than EDB, 1,2-DCP, and DBCP. Instead, the data suggest that more consistent monitoring of entire WRIA 1 is required to understand the real threat to ground water from all pesticides and develop management measures to reduce the levels of contamination.

Table 3.6. List of pesticides detected in WRIA 1 during 1988.

Pesticide	Number of Observations	MCL (ppb)	Number of Non-Zero Detections	Range (ppb)
Propazine	31	-	31	0.2
Picloram	28	-	28	1
Metribuzin	31	-	31	0.4
Metolachlor	31	-	31	1.5
Fenamiphos	33	-	33	0.3
Hexazinone	33	-	33	0.3
Ametryn	31	-	31	0.3
Dalapon	28	200	28	5
Cyanazine	28	-	28	0.80
Chloramben	28	-	28	0.50
Carboxin	28	-	28	1.0
Bentazon	28	-	28	0.50
Diuron	31	-	31	0.50
Diphenamid	31	-	31	0.40
Dicamba	28	-	28	0.20
Baygon (Propoxur)	31	-	31	1.10
Aldicarb Sulfoxide	28	-	28	1
Aldicarb Sulfone	28	-	28	1
Aldicarb	28	3	28	1.5

Table 3.7. Summary data of 2,4-D contamination in WRIA 1; MCL=70 ppb).

Year	Number of Observations	Number of Non-Zero Values	Number of Observations >MCL	Range (ppb)
1988	28	28	0	0.50
1992	1	0	0	0
1993	7	0	0	0
1994	56	0	0	0
1995	30	0	0	0
1996	9	0	0	0
1997	28	0	0	0
1998	22	0	0	0
1999	49	0	0	0
2000	19	0	0	0

Table 3.8. Summary data of 2,4-DB contamination in WRIA 1.

Year	Number of Observations	Number of Non-Zero Values	Number of Observations >MCL	Range (ppb)
1988	28	28	-	2
1992	1	0	-	0
1993	7	0	-	0
1994	60	1	-	<0.24
1995	20	0	-	0
1996	12	3	-	<0.24
1997	46	18	-	<0.24
1998	24	2	-	<0.24
1999	49	0	-	0
2000	19	0	-	0

Table 3.9. Summary data of alachlor contamination in WRIA 1; MCL=2 ppb.

Year	Number of Observations	Number of Non-Zero Values	Number of Observations >MCL	Range (ppb)
1988	28	28	0	1
1989	2	2	0	1
1990	-	-	-	-
1991	-	-	-	-
1992	6	0	0	0
1993	48	0	0	0
1994	54	0	0	0
1995	39	0	0	0
1996	17	0	0	0
1997	38	0	0	0
1998	26	0	0	0
1999	38	0	0	0
2000	41	0	0	0

Table 3.10. Summary data of carbofuran contamination in WRIA 1; MCL=40 ppb.

Year	Number of Observations	Number of Non-Zero Values	Number of Observations >MCL	Range (ppb)
1988	28	28	0	0.5-2.4
1989	1	1	0	2
1990	-	-	-	-
1991	-	-	-	-
1992	1	0	0	0
1993	46	3	0	0-0.03
1994	57	5	0	0-0.05
1995	24	0	0	0
1996	7	0	0	0
1997	30	0	0	0
1998	24	0	0	0
1999	38	0	0	0
2000	19	0	0	0

Table 3.11. Summary data of chlordane contamination in WRIA 1; MCL=2 ppb.

Year	Number of Observations	Number of Non-Zero Values	Number of Observations >MCL	Range (ppb)
1988			0	0.5-2.4
1989		1	0	2
1990		-	-	-
1991	-	-	-	-
1992	1	0	0	0
1993	46	3	0	0-0.03
1994	57	5	0	0-0.05
1995	24	0	0	0
1996	7	0	0	0
1997	30	0	0	0
1998	24	0	0	0
1999	38	0	0	0
2000	19	0	0	0

Table 3.12. Summary data of terbacil contamination in WRIA 1; Health Advisory Level=90 ppb.

Year	Number of Observations	Number of Non-Zero Values	Number of Observations >MCL	Range (ppb)
1988	28	28	-	3.5
1989	3	3		3.5
1990		-	-	-
1991	-	-	-	-
1992	6	0	-	0
1993	41	0	-	0
1994	-	-	-	-
1995	33	0	-	0
1996	18	0	-	0
1997	38	0	-	0
1998	26	0	-	0
1999	37	0	-	0
2000	41	0	-	0

Table 3.13. Summary data of oxamyl contamination in WRIA 1; Health Advisory Level=0.2 ppm, MCL=200 ppb.

Year	Number of Observations	Number of Non-Zero Values	Number of Observations >MCL	Range (ppb)
1988	28	28	0	0.6
1989				
1990		-	-	-
1991	-	-	-	-
1992	1	0	0	0
1993	7	0	0	0
1994	13	0	0	0
1995	24	0	0	0
1996	7	0	0	0
1997	28	0	0	0
1998	24	0	0	0
1999	37	0	0	0
2000	19	0	0	0

Table 3.14. Summary data of bromacil contamination in WRIA 1; Health Advisory Level=90 ppb.

Year	Number of Observations	Number of Non-zero Values	Number of Observations >MCL	Range (ppb)
1988	29	29	-	2.2
1989	3	3		2.2
1990		-	-	-
1991	-	-	-	-
1992	6	0	-	0
1993	48	0	-	0
1994	46	2	-	0-0.21
1995	33	0	-	0
1996	18	0	-	0
1997	38	0	-	0
1998	26	0	-	0
1999	37	0	-	0
2000	41	0	-	0

3.5 Pesticide-Related Management Issues

3.5.1 Current Status and Historical Trends of Pesticides in Ground water

Presently, pesticides appear not to be a concern in most parts of WRIA 1 except Bertrand Creek and Fishtrap Creek sub-basins based on available ground water quality data. However, as discussed earlier, the available data do not represent the true ground water quality across the entire WRIA 1 due to inconsistent monitoring periods and limited spatial uniformity of monitoring locations. Much of the data have been collected from areas with heavy agricultural practices and known prior contamination. Due to this reason, true ground water quality across entire WRIA 1 cannot be accurately evaluated. The available ground water quality data definitely indicate serious ground water quality impacts in Bertrand and Fishtrap Creek sub-basins especially from EDB and 1,2-DCP.

Regional ground water flow is toward the Nooksack River to the south, but, since the aquifer is shallow and unconfined, ground water flow pattern is locally affected by surface water and seasonal variations due to pumping and irrigation practices (Erickson et al., 1990). Bertrand Creek is an ungauged stream that discharges to the Nooksack River which is the primary discharge in the area. The main drainages and ground water are hydraulically interconnected (Erickson et al., 1990). Pesticide contamination has been identified in Canadian ground water that is hydraulically connected to the Sumas Aquifer in the Nooksack Basin (Erickson et al. 1995). Also, the mobility of 1,2-DCP is considerable (refer to Section 3.2). Under these conditions, the potential for some movement of 1,2-DCP and other pesticides from surface water to ground water or vice-versa exists. However, quantification of this chemical transport is not possible due to lack of data. Therefore, land use activities in Canada may have some influence on ground water quality in WRIA 1. More evidence can be gathered if controlled local-scale field monitoring can be performed in the future.

At minimum, further field investigations for currently used pesticides, EDB and 1,2-DCP are needed in the Bertrand Creek/Fishtrap Creek area. Although natural attenuation, reduced application, and regulatory measures such as banning of pesticides may have contributed to the decrease of pesticides in ground water, this possibility cannot be confirmed across entire WRIA 1 unless a consistent, spatially distributed monitoring scheme is implemented for the next few years.

Agriculture is the major reason for pesticide contamination. Since the Sumas-Blaine aquifer, which supports most of the agriculture, is a highly vulnerable aquifer, any pesticide that is applied to crops has the potential to move into shallow ground water.

As discussed earlier, pesticides remain a concern in certain parts of WRIA 1 and in other areas, the limited information and existing ground water quality data are inadequate to rule out pesticide concerns completely at the present time. The areas of serious known concerns are (a) *Bertrand Creek area* - This area is located in the southern part of Bertrand Creek and the area is prone to high pesticide contamination. Pesticides applied elsewhere in the area

may be accumulating in this area due to the existing local ground water flow pattern and hydrogeologic conditions and (b) *Fishtrap-Johnson Creek area* - This area is located to the northeast of Lynden and to the west of Sumas.

3.5.2 Data Availability and Subsequent Modeling

Currently available data are not sufficient for detailed modeling of pesticide fate and transport in WRIA 1. This lack of data is especially visible in Bertrand Creek and Fishtrap Creek sub-watersheds. Surface water data of pesticides, which is essential to analyze the surface water-ground water interactions, are not available. Also, sufficient ground water quality data are not available to understand the trans-border flow from Canada to US.

The recommendation from this study is to expand the existing field monitoring to include pesticide measurements in ground water across entire WRIA 1 with special emphasis on areas with heavy agricultural practices and known contamination. The monitoring scheme should include consistent data collection at regular intervals and the monitoring locations should be spatially distributed such that a good understanding of overall fate and transport of pesticides across WRIA 1 can be developed. In addition to ground water samples, soil samples should be collected to determine the sorption and natural attenuation characteristics under controlled laboratory conditions. A collection of data from both field samples and laboratory experiments can provide a good scientific estimation of natural attenuation and transport potential of different pesticides such that the information can be used in management decision-making relevant to corrective actions.

Once sufficient data are available, then a re-assessment of pesticide contamination in WRIA 1 should be conducted to evaluate if pesticides are long-term concerns, and if so, the level of future monitoring and modeling needed. Modeling will also provide a better understanding of pesticide fate and transport in WRIA 1 and available options for preservation of ground water quality. The improved understanding from these actions can help decision-making related to various water management options to ensure ground water quality is preserved due to existing land use activities and other water development actions. Such decision-making will help to determine best management options related to ground water withdrawal affecting pesticide transport, application rate and frequency of pesticides, appropriate selection of environmentally-safe pesticides, and public education.

3.5.3 Future Data Monitoring Requirements

The important requirements for future monitoring are as follows:

- A comprehensive long-term field monitoring plan should be developed for both currently used pesticides and banned pesticides of concern.
- The monitoring plan should include two geographic areas:
    - Entire WRIA 1 to establish baseline concentrations
    - More focused sampling at selected areas based on agricultural practices and known prior contamination

- The plan should have consistent regular monitoring at spatially representative and uniform locations
- The monitoring should include soil sampling and subsequent laboratory experiments to assess natural attenuation potential and sorption characteristics
- A limited set of surface water quality sampling locations should be included in the plan
- Once data from 2-3 years are available, a re-assessment of ground water quality needs to be performed to determine the existing conditions and management interventions needed.

The areas for detailed data collection are Lynden-Everson-Nooksack-Sumas area that includes Bertrand Creek, Fishtrap Creek, and Johnson Creek sub-watersheds. Surface water needs to be monitored in Bertrand Creek, Fishtrap Creek, and other smaller tributaries that are located near the US-Canada border. Future modeling work should concentrate more on the southern part of Bertrand Creek, which seems to be a nodal point for all pesticides that have been applied in the area to accumulate.

3.5.4 Baseline Concentrations

The analysis performed from this study indicated degradation of pesticides in ground water to the extent of non-detectable levels in ground water. In addition, pesticides in ground water are due to man-made causes such as various land uses and not naturally occurring chemicals. Also, most pesticides that have shown serious contamination in the early part of 1990's have been banned from further use. Due to the combination of these reasons, no baseline concentration can be defined for pesticides that were analyzed in this report. Even for pesticides that are not banned and currently used in WRIA 1, there appears no substantial presence in ground water to describe baseline concentrations.

3.5.5 Management Options to Minimize Potential Pesticide Contamination

Pesticides are a concern presently and probably in the future in the southern part of Bertrand Creek, areas west of Sumas, and north of Lynden. Hence, proper land and water management decisions need to be enforced to minimize future pesticide concerns and avoid enhancing the current concerns. Management decisions should focus on proper pesticide and crop selection, application rate, and frequency of application. Careful attention should be focused on ground water extraction from these areas and areas that are hydraulically connected to the contaminated areas.

Natural degradation seems to be the best alternative to decrease the pesticide contamination in WRIA 1. At the same time, care has to be taken, through sound management decisions, to see that pesticides are not applied in excess in the LENS area, as the aquifer is highly vulnerable to contamination and to ensure the residents have access to good quality drinking water. Pesticides that are used on the major crops in WRIA 1 such as berries and seed potatoes deserve careful attention in field monitoring to ensure that ground water quality is preserved.

3.6 Summary and Conclusions

There were serious pesticide contamination issues present in the past in WRIA 1, especially with EDB, 1,2-DCP, and DBCP. Although some of these concerns appear to be decreasing in some parts of WRIA 1, still pesticides continue to be a serious concern in Bertrand and Fishtrap Creek sub-basins. Even in other parts of WRIA 1, the threat of pesticides cannot be completely ruled out based on existing ground water quality data alone. The reason for this conclusion is that ground water quality data are not collected through a consistent, spatially distributed monitoring network; instead, the data are more biased towards areas with heavy agricultural activities with prior known contamination of ground water. However, there is some degradation of pesticides that has produced decreasing concentrations in ground water. Other reasons for decreasing concentrations may be regulatory actions such as banning of pesticides, reduced demand from crops, and reduced applications of currently used pesticides.

The affected area in Bertrand Creek sub-basin is located in the southern part of the Bertrand Creek sub-basin. The next affected area is the Fishtrap-Johnson Creek area located to the northeast of Lynden and to the west of Sumas.

- Agricultural related land use activities are the major reason for pesticide contamination in WRIA 1. Since the Sumas-Blaine aquifer, which supports most of the agriculture, is a highly vulnerable aquifer, whatever pesticide is applied to crops can easily leach into ground water.

- Detailed modeling of pesticide transport in WRIA 1 is difficult due to the lack of sufficient surface water and ground water quality data. However, a lumped-parameter modeling approach is possible to evaluate the transient mass balance in selected areas. Such a modeling exercise will provide the natural attenuation potential of soil and ground water to reduce the pesticide concentrations and the time span for degradation. Surface water quality data of pesticides, which are essential to analyze the surface water-ground water interactions, are not adequate.

- The Bertrand Creek area shows a potential threat of pesticide contamination in the future. Pesticides are a concern at the present time in the southern part of Bertrand Creek sub-watershed, and some areas to the west of Sumas and to the north of Lynden. Improved management decisions related to pesticide application need to be implemented across WRIA 1, especially in areas with heavy agricultural activities. Future ground water development should avoid these vulnerable areas due to potential pesticide contamination of ground water.

- Pesticides will always remain to be a concern in WRIA 1, in general, due to heavy agricultural activities. Therefore, regular monitoring of ground water quality across WRIA 1 should be carried out on a regular basis. The monitoring network should be carefully designed to capture key land-use activities contributing to pesticide

contamination and special hydrologic features in selected areas that may contribute to fate and transport of pesticides.

- There seems to be some pesticide movement between the surface water and ground water, especially in tributaries within the LENS area. However, quantification of this interaction is not possible due to lack of data.

- Land-use activities in Canada can influence the ground water quality in WRIA 1 as the aquifers in Canada and in Whatcom County are hydraulically connected.

Natural degradation is one of the most cost-effective alternatives to reduce the existing pesticide contamination in WRIA 1, together with other management interventions. Despite the potential of natural attenuation, sound management decisions should be taken to ensure that excess pesticides are not applied in the LENS area due to high vulnerability of the aquifer and to ensure that the residents have access to good quality drinking water.

3.7 Limitations of the Study

As with all studies dealing with real-life data and goals, this study also has some limitations. Therefore, it is the desire of the USU team to provide a good insight to the limitations of this study before the results and conclusions are extended to additional work. The limitations are described as follows:

- *Lack of consistent monitoring plan* - The data obtained from various agencies have not been collected following a consistent, regular monitoring plan using a spatially distributed monitoring network. Instead, the monitoring locations and the frequency of monitoring are biased to areas with heavy agricultural activities and prior known ground water contamination.

- *Quality control of data* - The data obtained from various agencies have been circulated through various local databases before available to the USU team. The information submitted with the data did not provide any indication related to quality control and quality assurance. Although the USU team made every attempt to identify possible data with concerns, the validity and accuracy of the data cannot be guaranteed by the USU team at this stage of database preparation.

- *Biased data* - Due to biased sampling from previous years, the existing ground water quality data may not necessarily represent the true ground water across the entire WRIA 1, but a detailed view of ground water quality of selected areas. There may be other areas of WRIA 1, such as the southern and western parts of WRIA 1, where much less monitoring has been performed, but may have more serious ground water quality concerns than expected. Unless a sound long-term monitoring plan is developed and implemented to obtain a set of reliable and well-represented data, the true ground water across the entire WRIA 1 cannot be assessed at the present time.

- *Repeated sampling* - The available ground water quality data have been collected from different types of well, such as regular monitoring wells, municipal wells, private wells, etc. In the analysis, we have assumed that each data point represents the ground water quality at that location at the time of measurement. However, this assumption may not necessarily hold at each location due to the method of sample collection, analysis, and other quality control protocols. In addition, some wells have been sampled more frequently and repeatedly by different organizations. When such sampling bias is found, appropriate measures were taken in the analysis to avoid the bias in the results.

- *Level of contamination* - Due to limited availability of spatially distributed data at consistent time intervals, the level of contamination defined by different sub-basins may be too conservative. For example, the basis to define a given sub-basin as contaminated during a given year is the presence of at least one data point exceeding the respective MCL. In reality, this worst case scenario may represent a small area with residual contamination that may not necessarily have any advective-dispersive mass transport, but natural attenuation only.

- *Interpretation and conclusions* – This work was based on existing data, and no new data were collected to the satisfaction of the USU team. Therefore, the discussion and conclusions drawn from this study are based on existing data only and are somewhat qualitative due to lack of detailed data to quantify the interpretations. As discussed in the previous sections, more focused pilot studies, at least in the Bertrand Creek and Fishtrap Creek areas, need to be conducted to validate some of the findings from this study.

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APPENDIX A - ANNUAL MAXIMUM [NO3+NO2] CONCENTRATIONS IN DIFFERENT SUB-BASINS

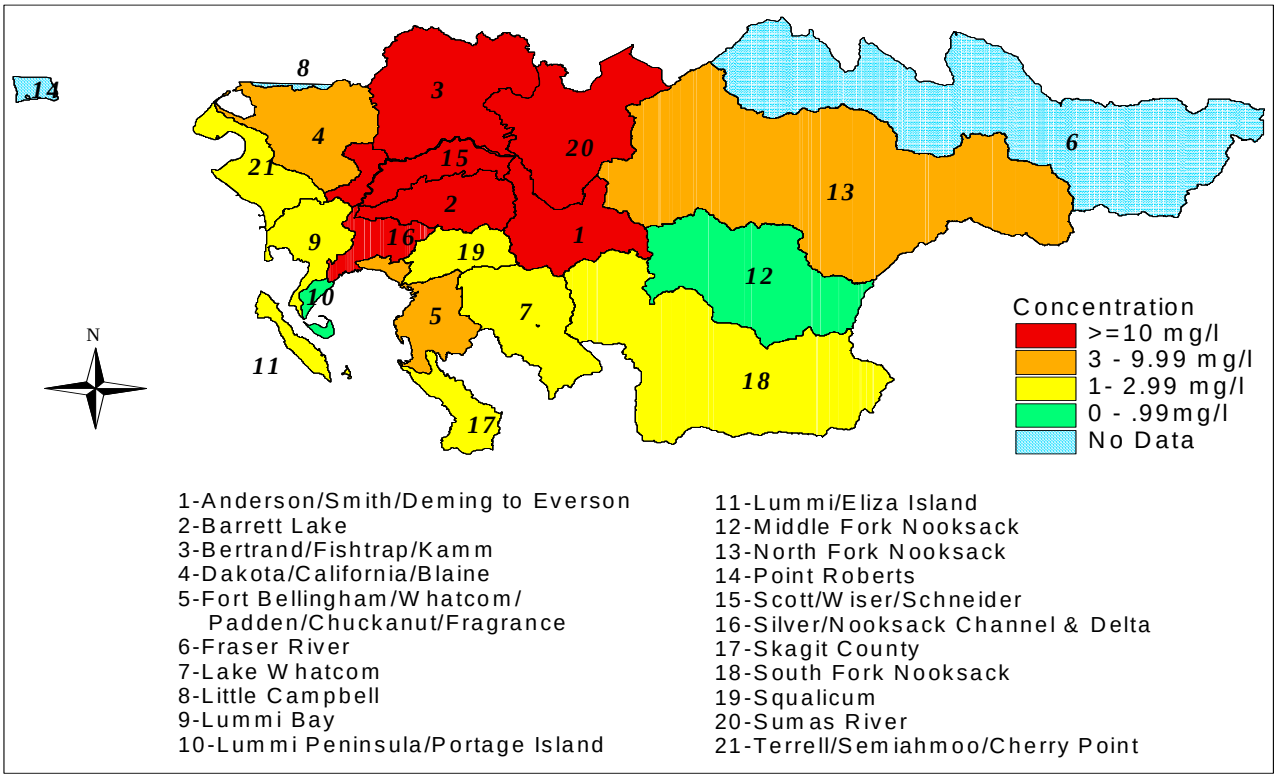

Figure A-1. Maximum [NO3+NO2] concentration in different sub-basins in 1990.

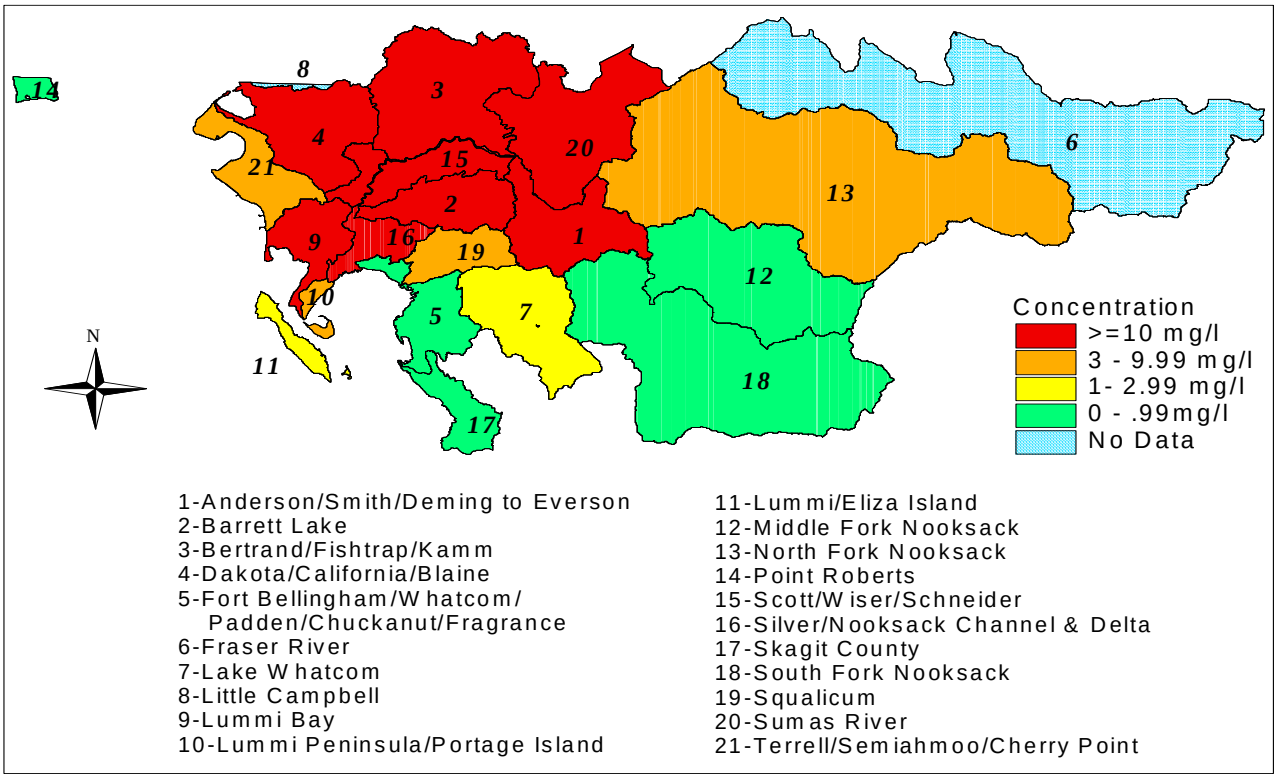

Figure A-2. Maximum [NO3+NO2] concentration in different sub-basins in 1991.

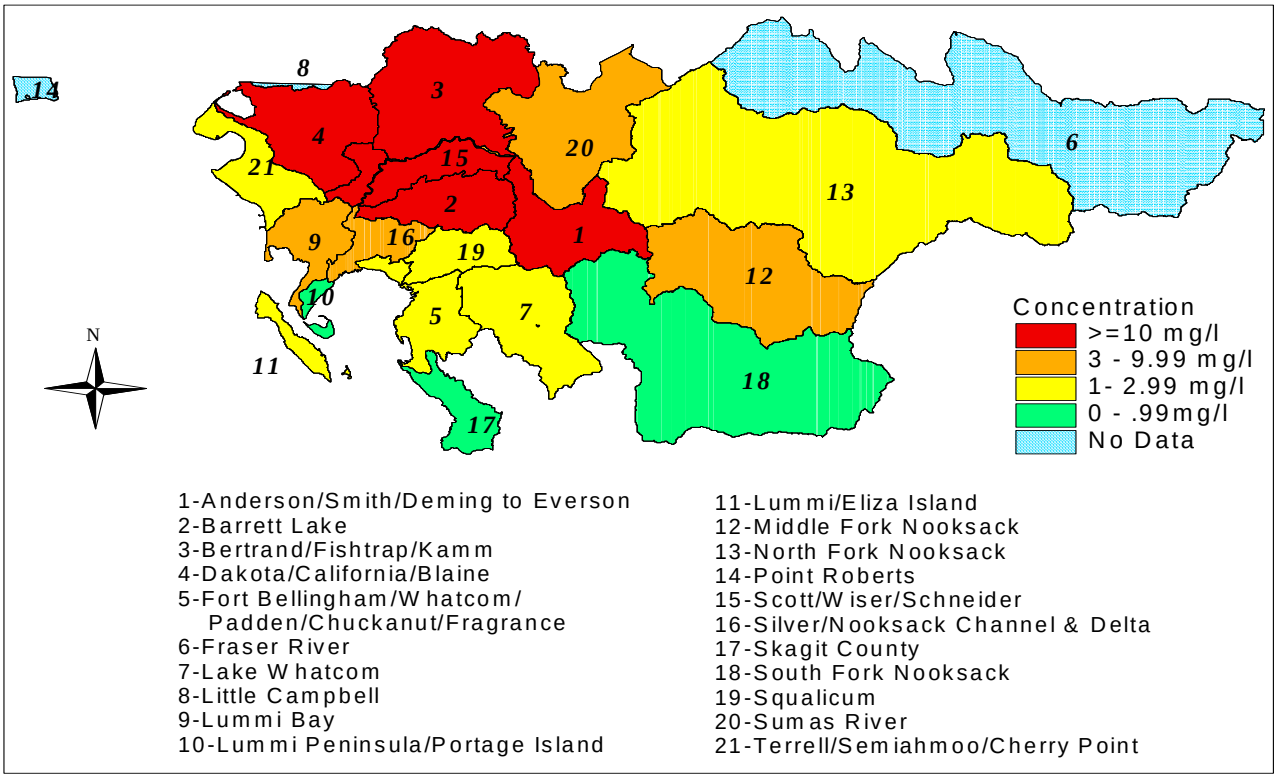

Figure A-3. Maximum [NO3+NO2] concentration in different sub-basins in 1992.

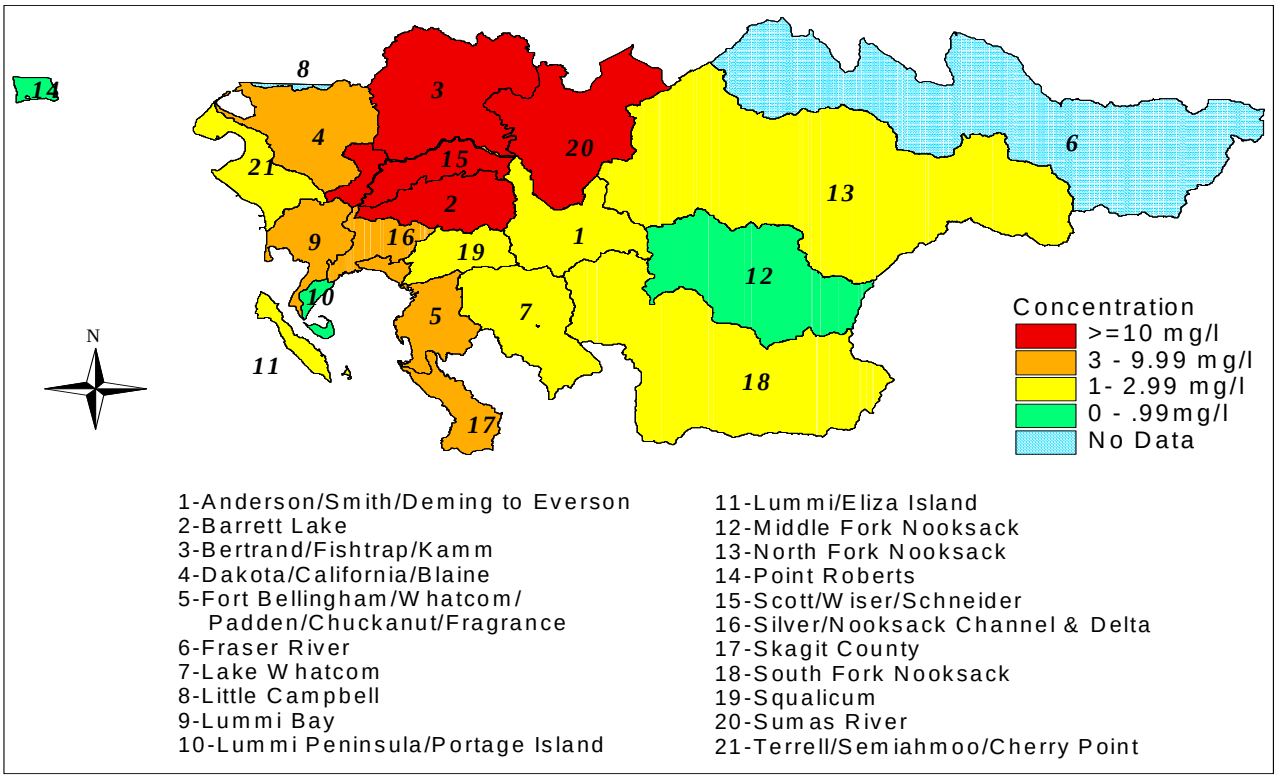

Figure A-4. Maximum [NO3+NO2] concentration in different sub-basins in 1993.

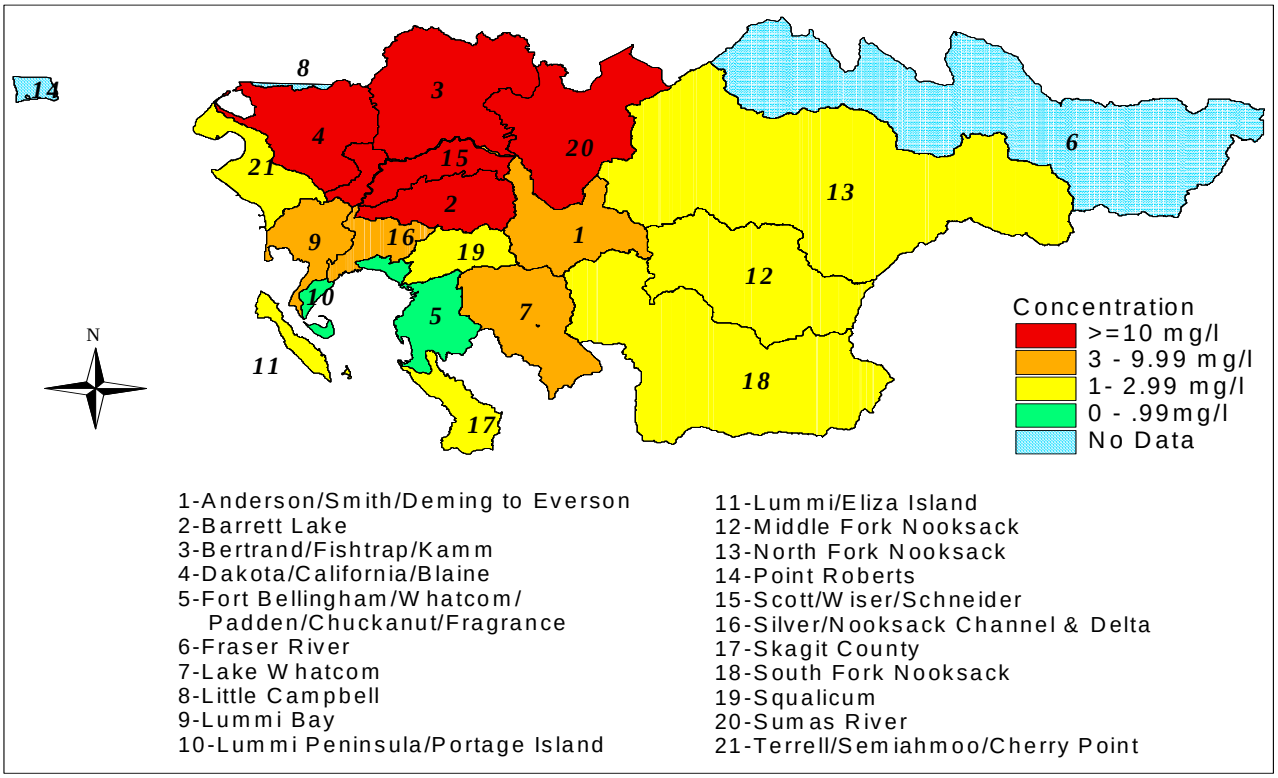

Figure A-5. Maximum [NO3+NO2] concentration in different sub-basins in 1994.

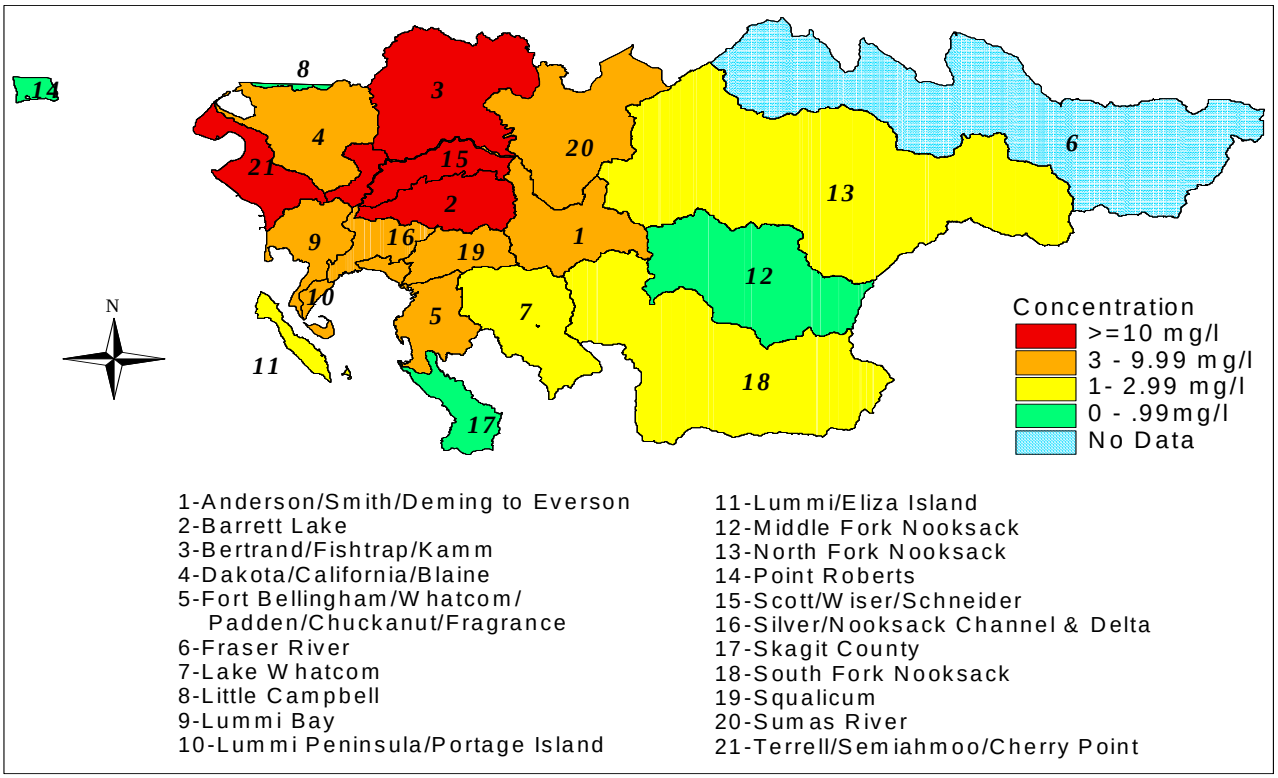

Figure A-6. Maximum [NO3+NO2] concentration in different sub-basins in 1995.

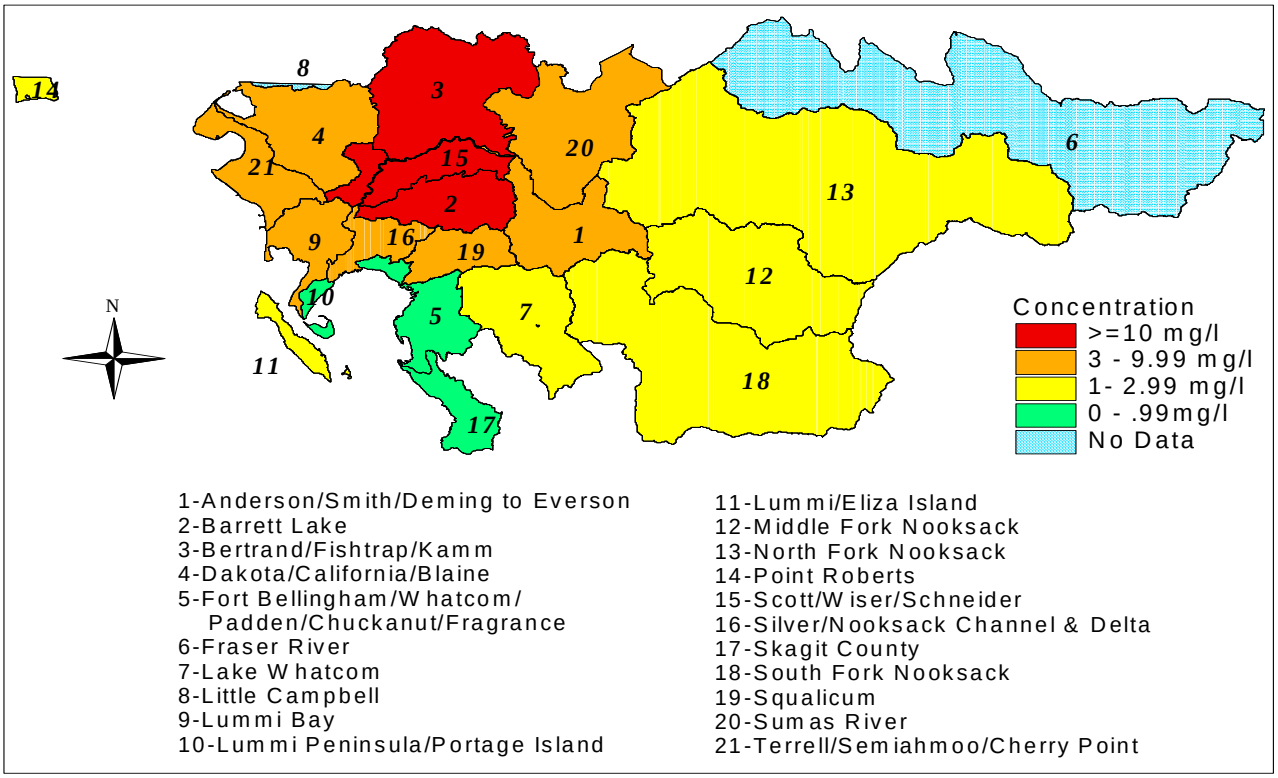

Figure A-7. Maximum [NO3+NO2] concentration in different sub-basins in 1996.

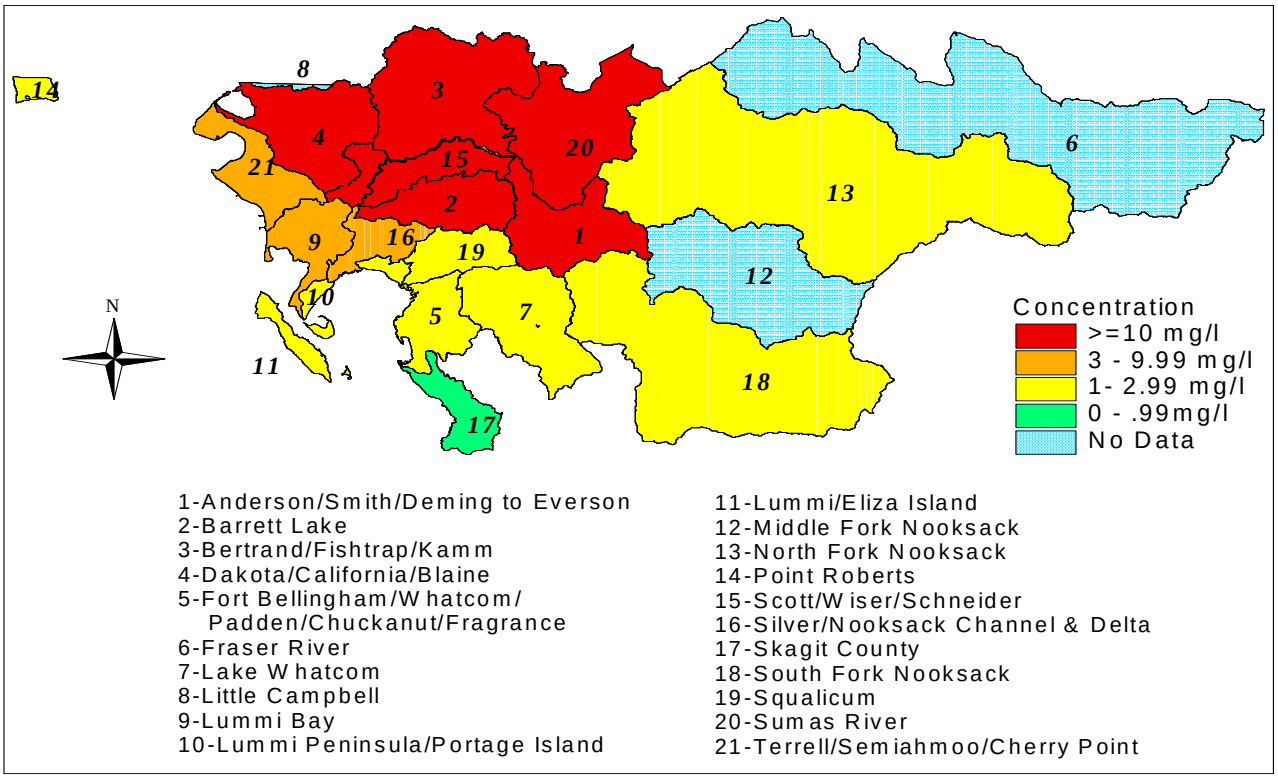

Figure A-8. Maximum [NO3+NO2] concentration in different sub-basins in 1997.

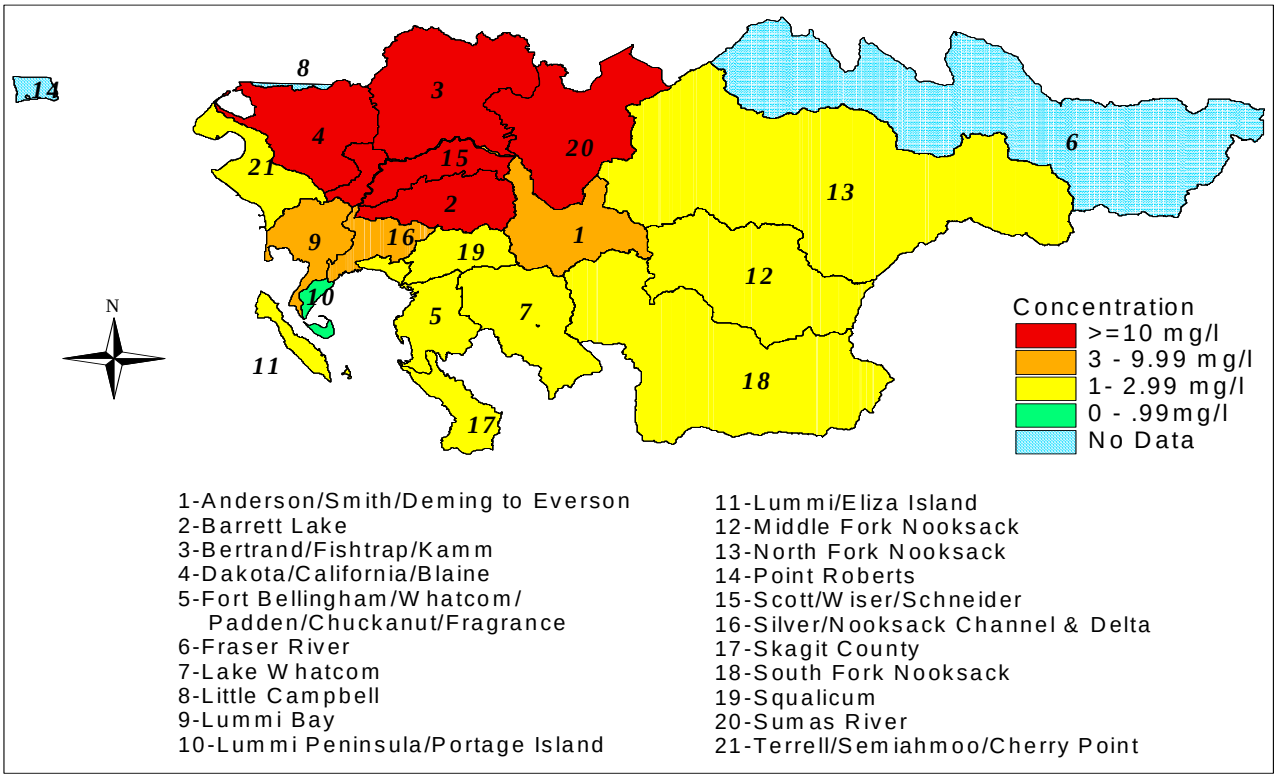

Figure A-9. Maximum [NO3+NO2] concentration in different sub-basins in 1998.

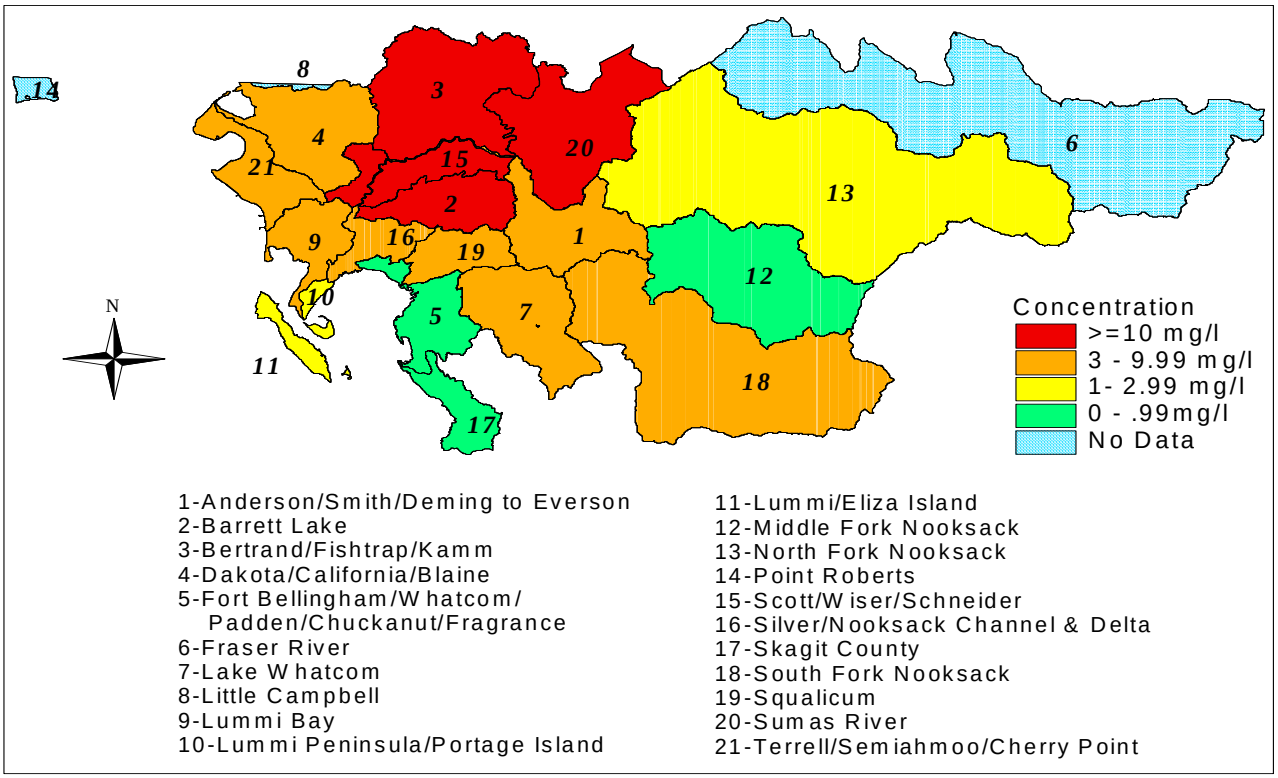

Figure A-10. Maximum [NO3+NO2] concentration in different sub-basins in 1999.

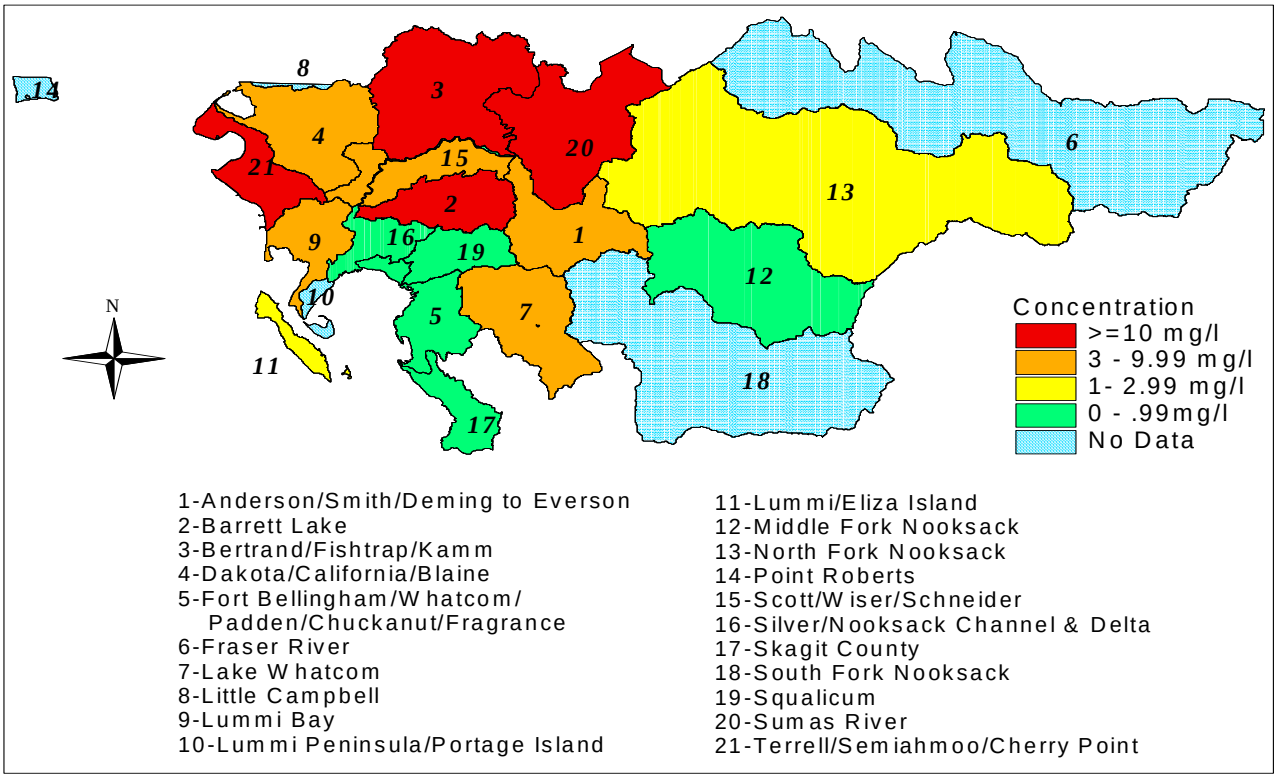

Figure A-11. Maximum [NO3+NO2] concentration in different sub-basins in 2000.

APPENDIX B - ANNUAL MEDIAN [NO3+NO2] CONCENTRATIONS IN DIFFERENT SUB-BASINS

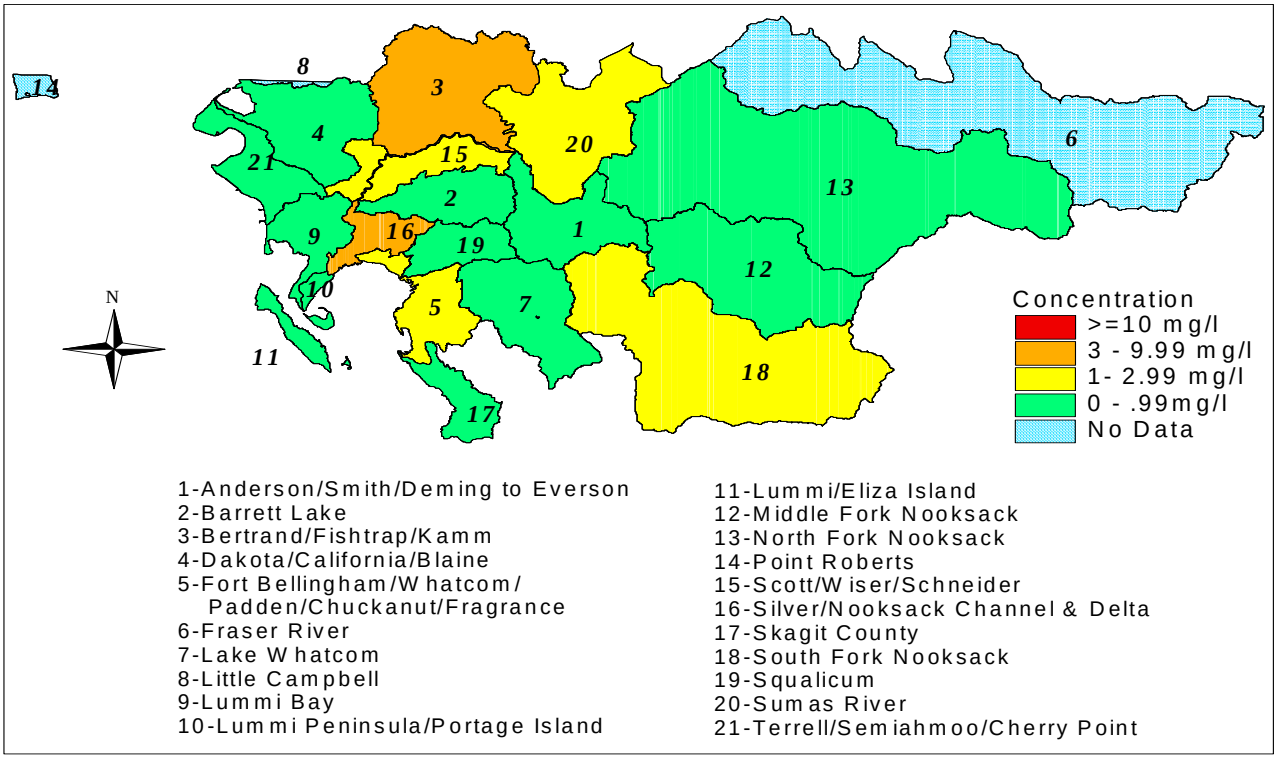

Figure B-1. Median [NO3+NO2] concentration in sub-basins in 1990.

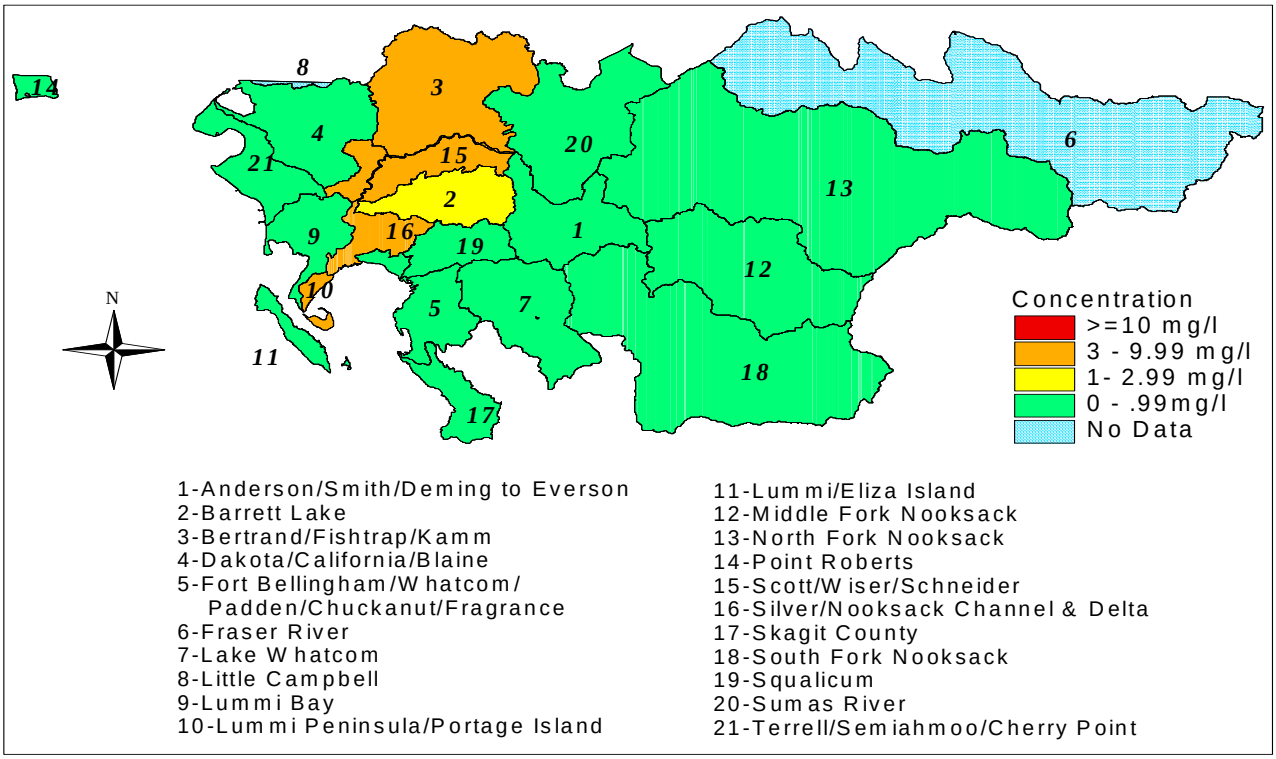

Figure B-2. Median [NO3+NO2] concentration in sub-basins in 1991.

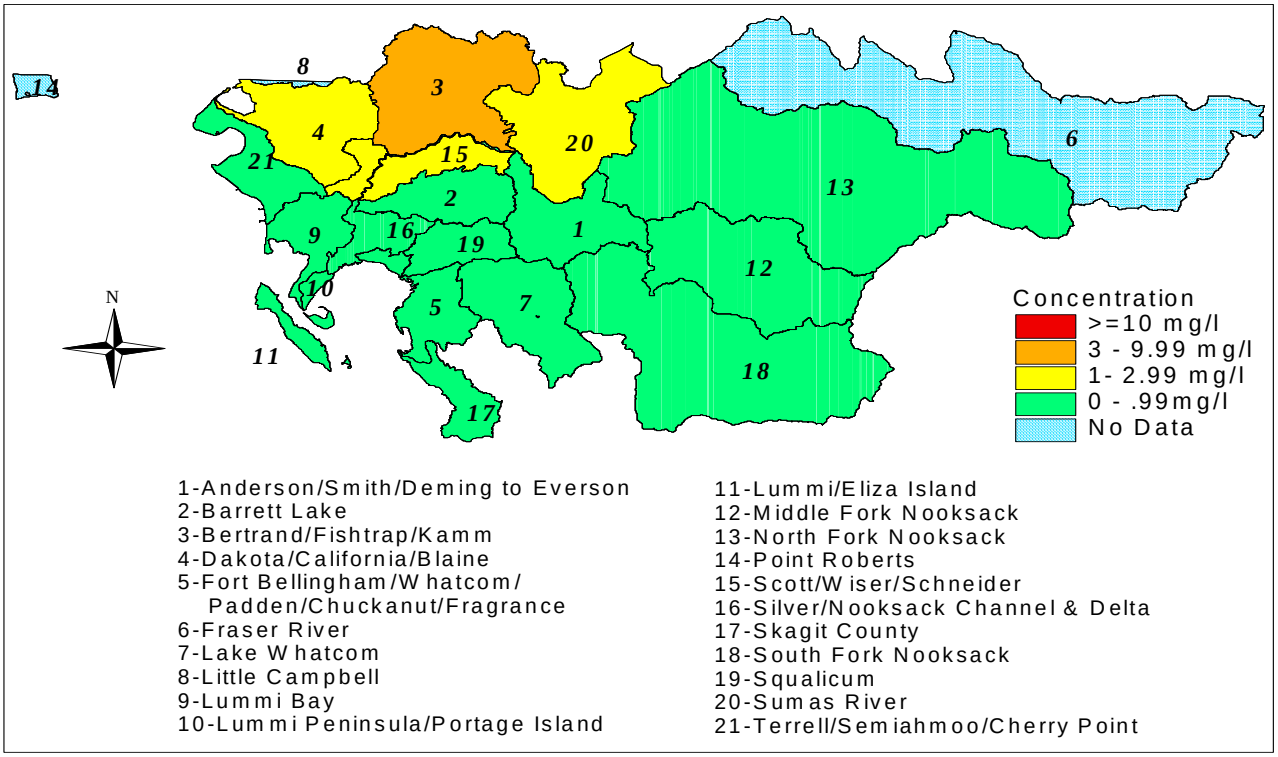

Figure B-3. Median [NO3+NO2] concentration in sub-basins in 1992.

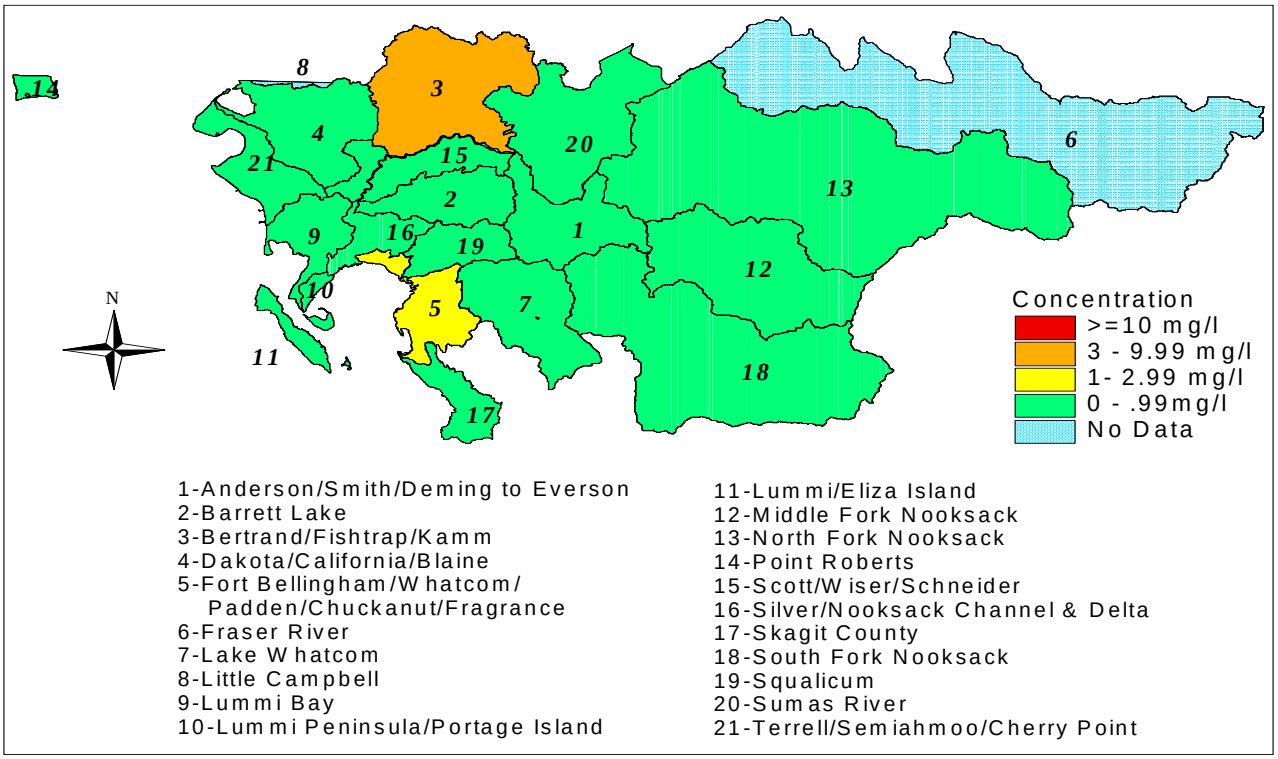

Figure B-4. Median [NO3+NO2] concentration in sub-basins in 1993.

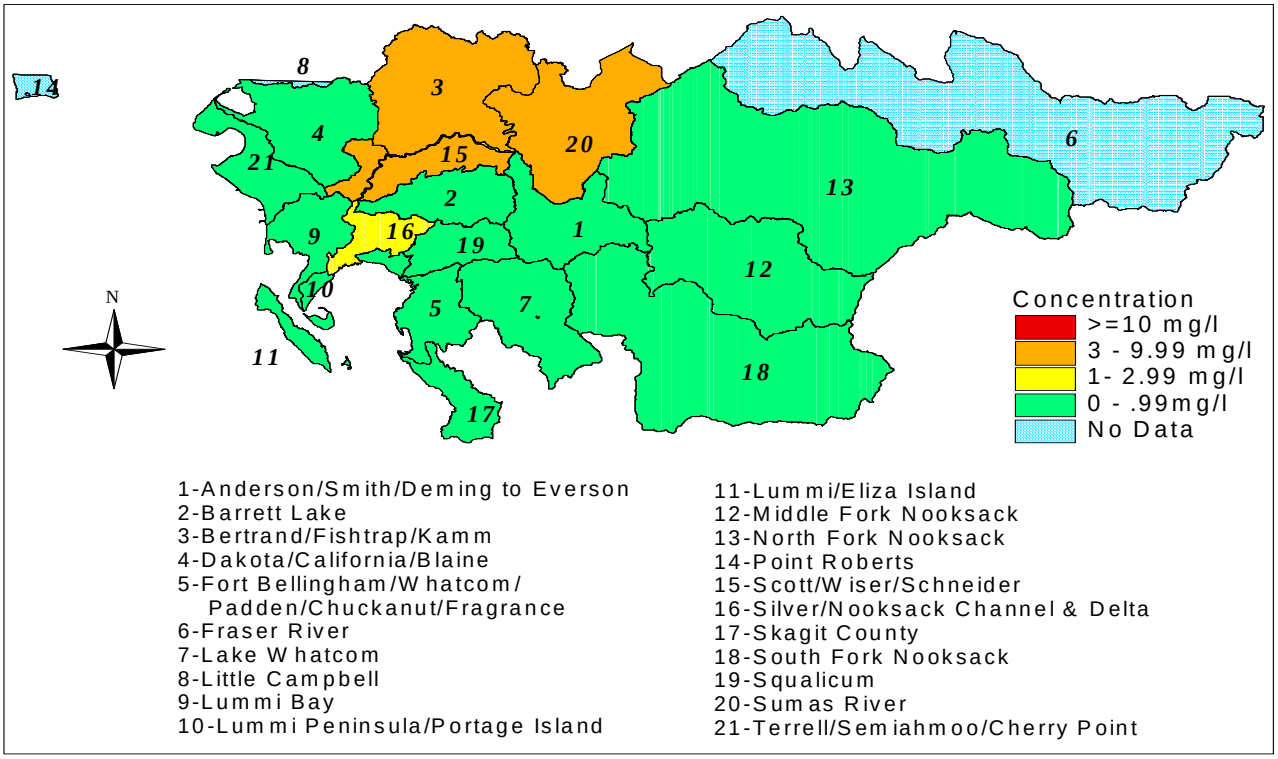

Figure B-5. Median [NO3+NO2] concentration in sub-basins in 1994.

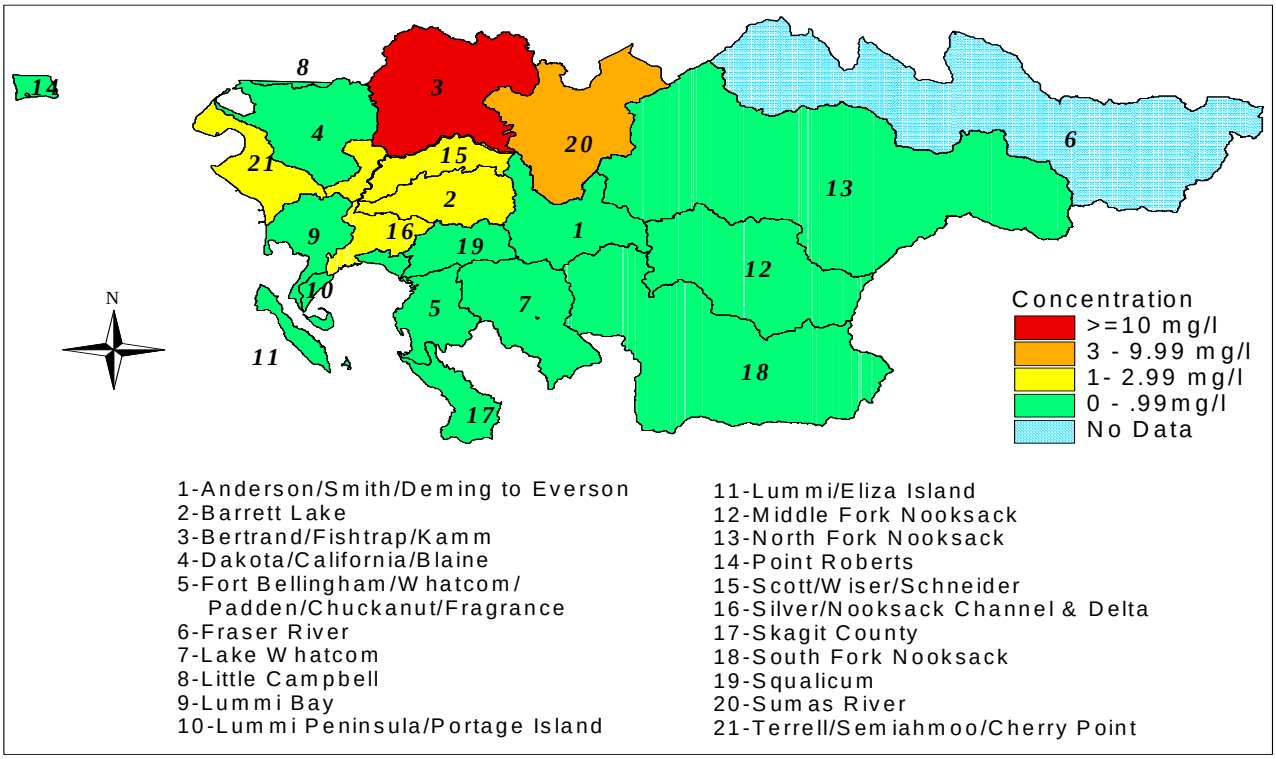

Figure B-6. Median [NO3+NO2] concentration in sub-basins in 1995.

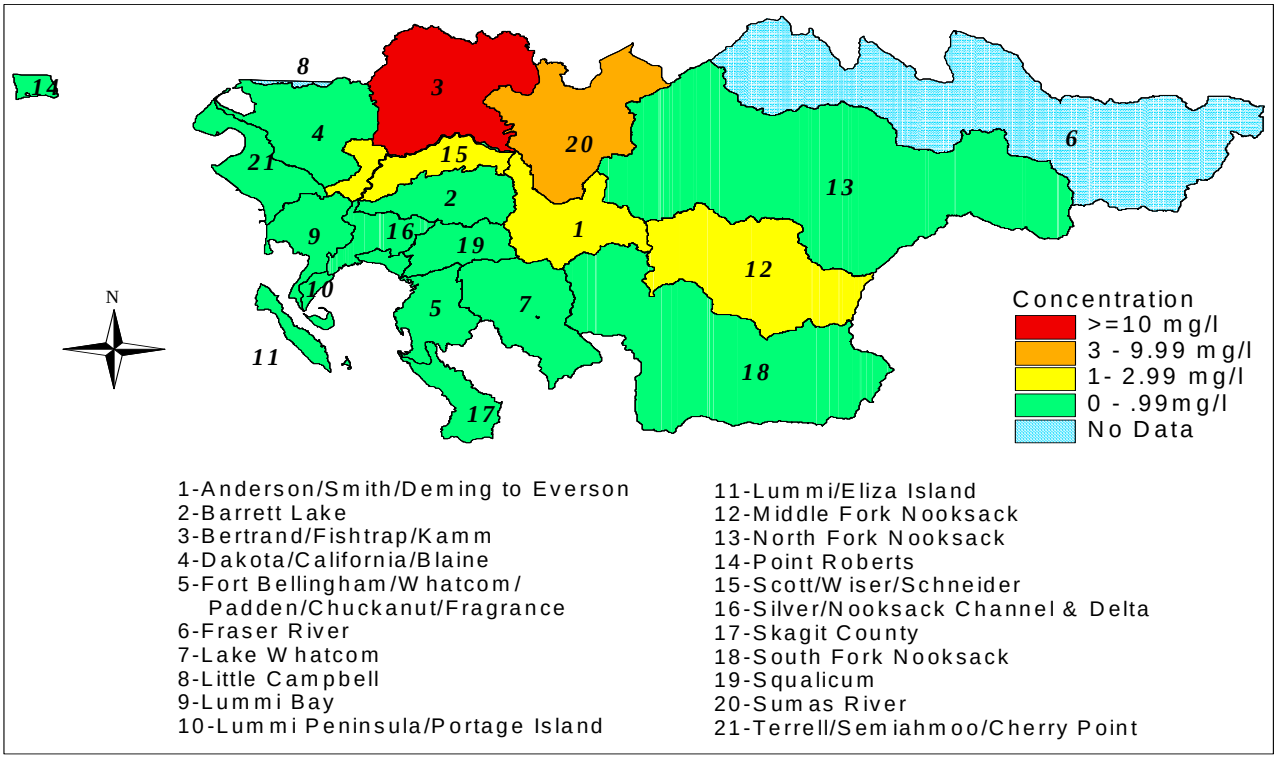

Figure B-7. Median [NO3+NO2] concentration in sub-basins in 1996.

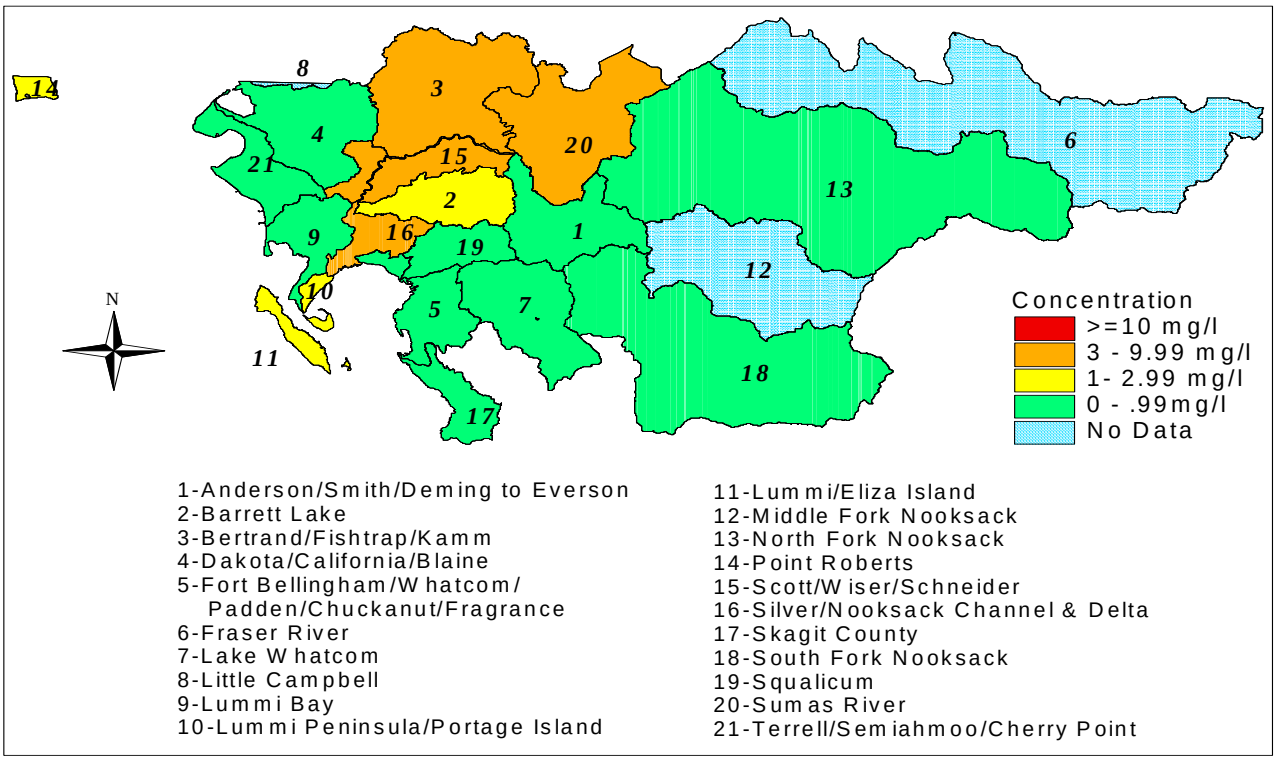

Figure B-8. Median [NO3+NO2] concentration in sub-basins in 1997.

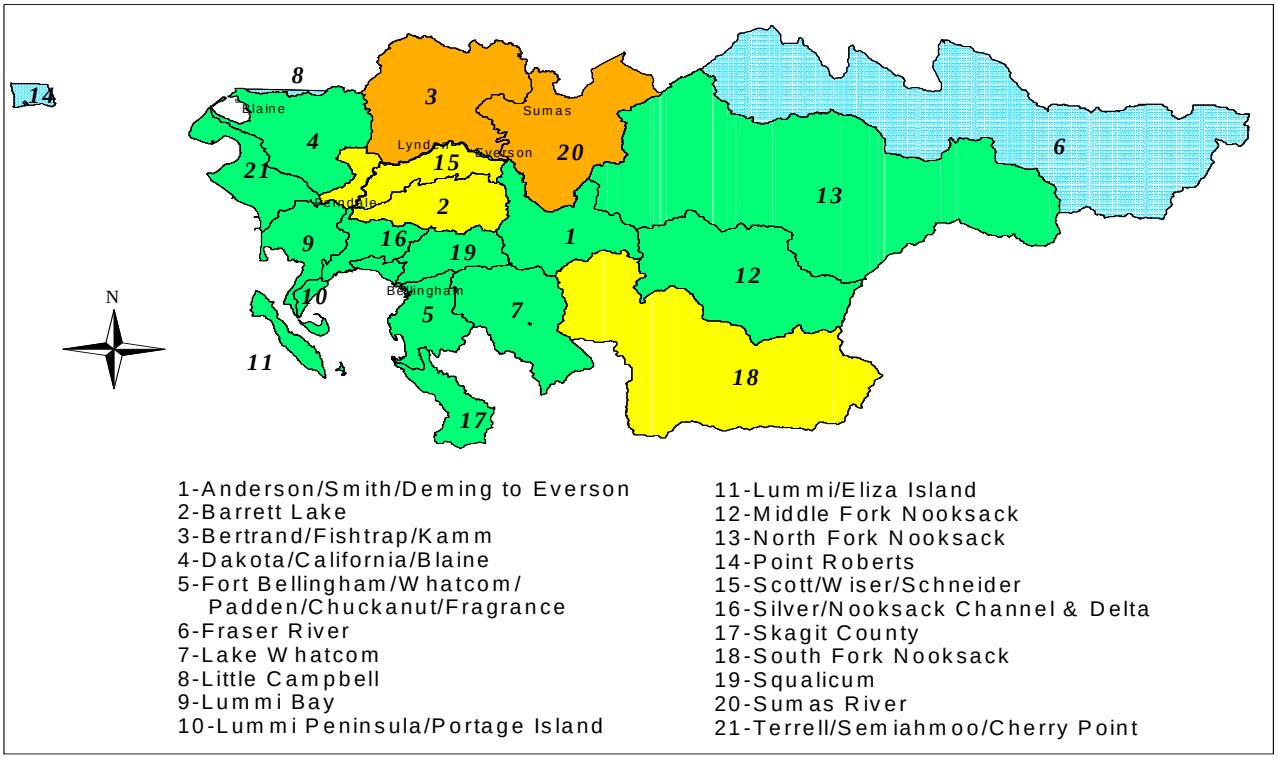

Figure B-9. Median [NO3+NO2] concentration in sub-basins in 1998.

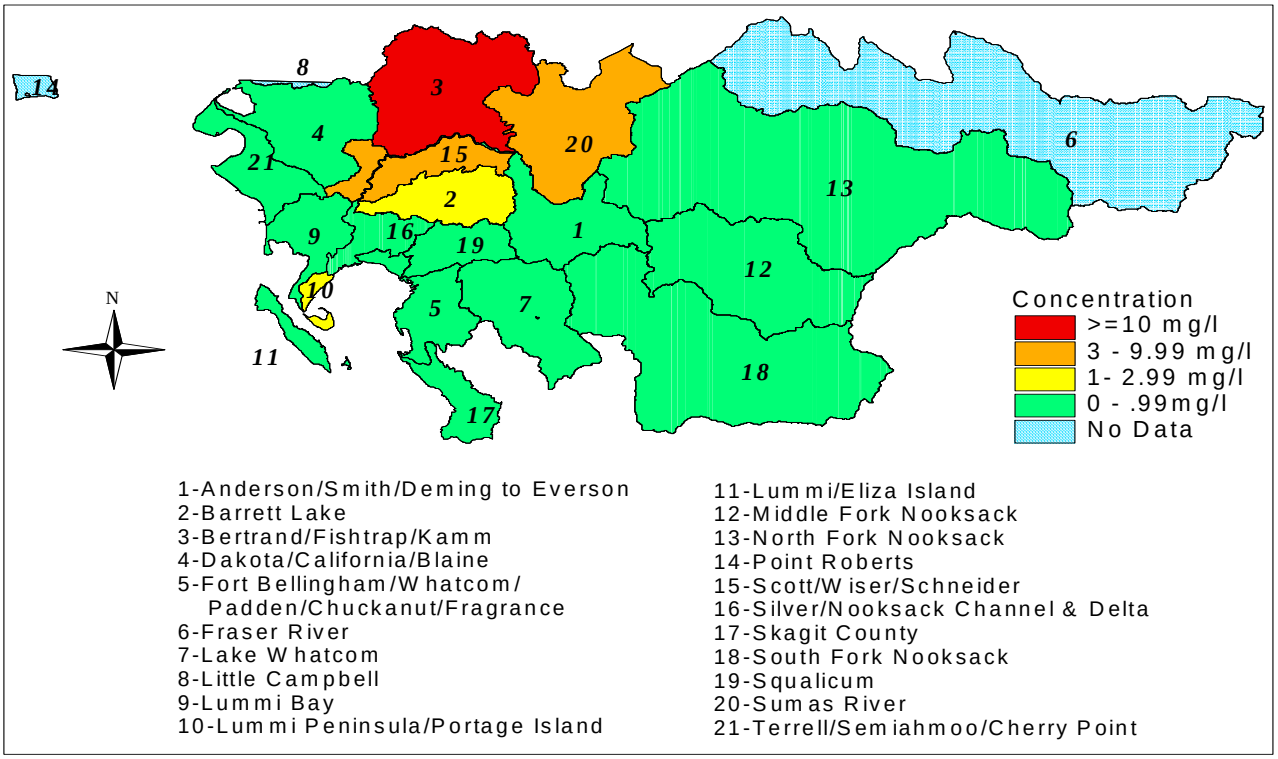

Figure B-10. Median [NO3+NO2] concentration in sub-basins in 1999.

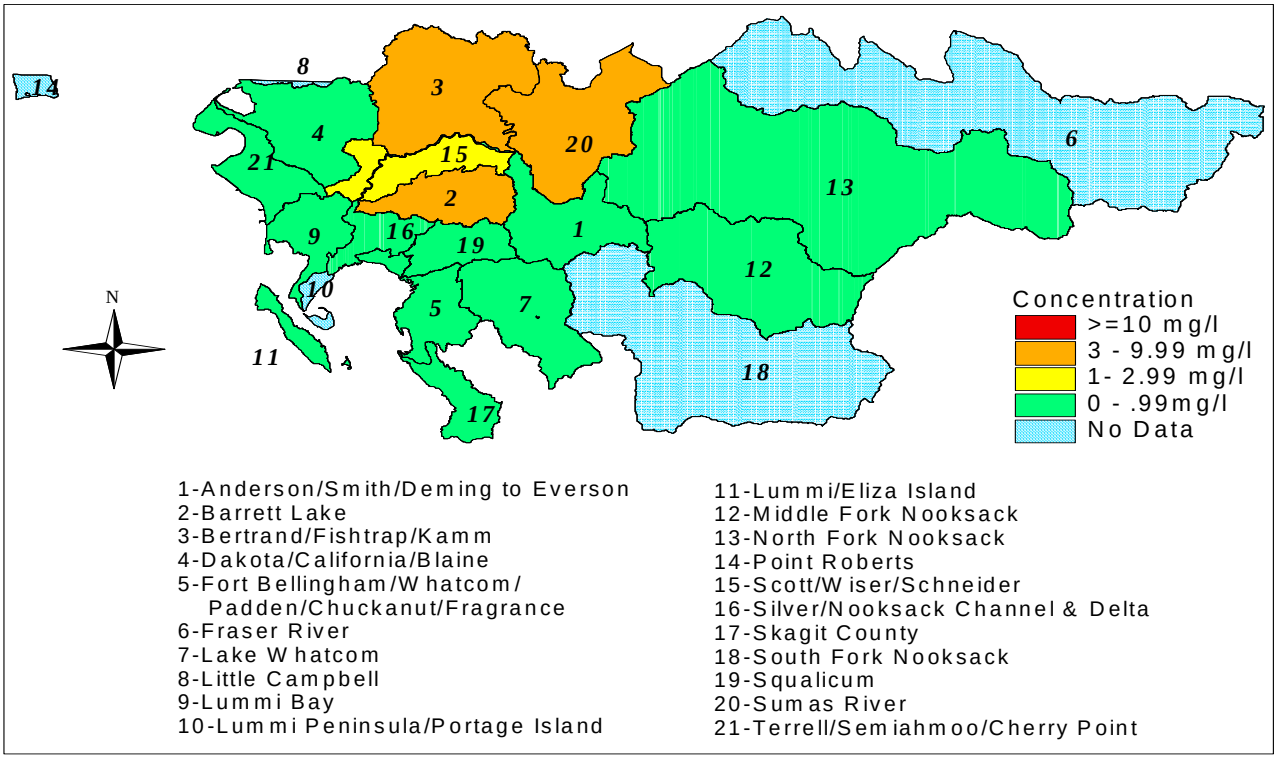

Figure B-11. Median [NO3+NO2] concentration in sub-basins in 2000.

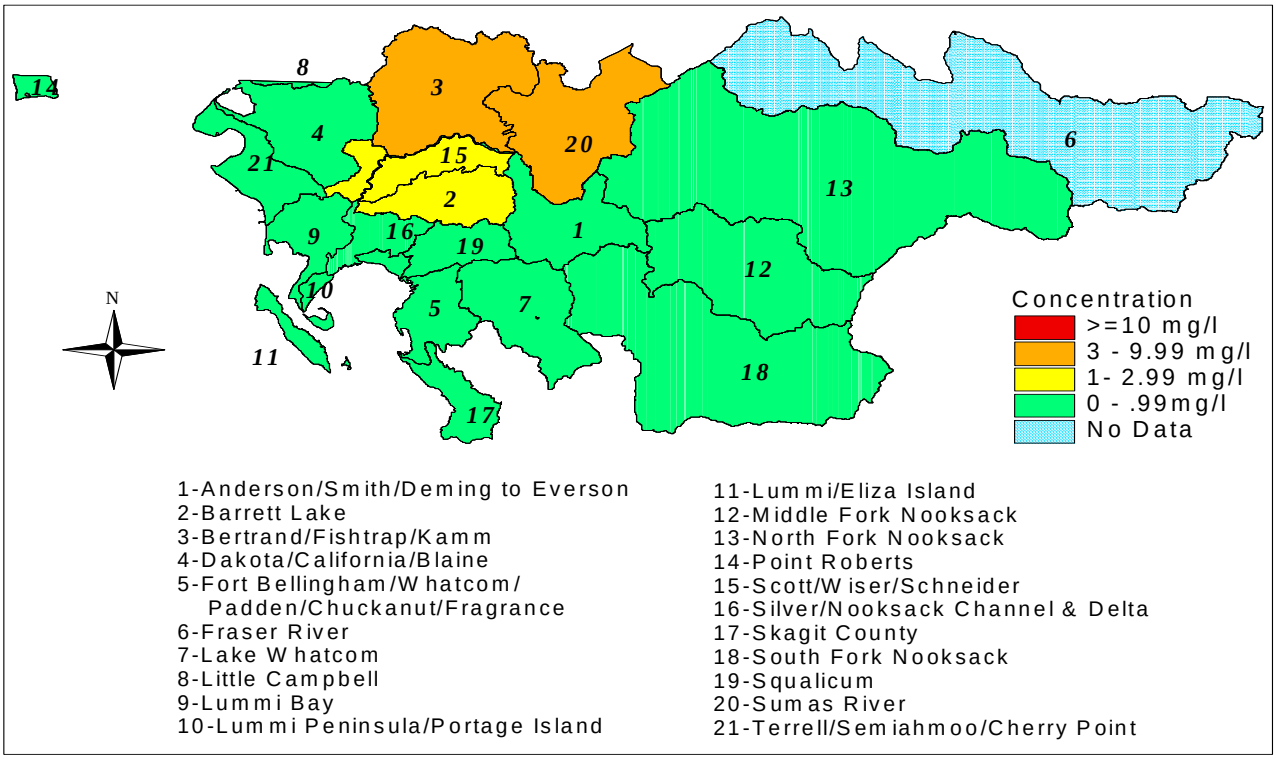

Figure B-12. Median [NO3+NO2] concentration in sub-basins in 1990-2000.

APPENDIX C - AREAL DISTRIBUTION OF ANNUAL MAXIMUM [NO3+NO2] CONCENTRATIONS

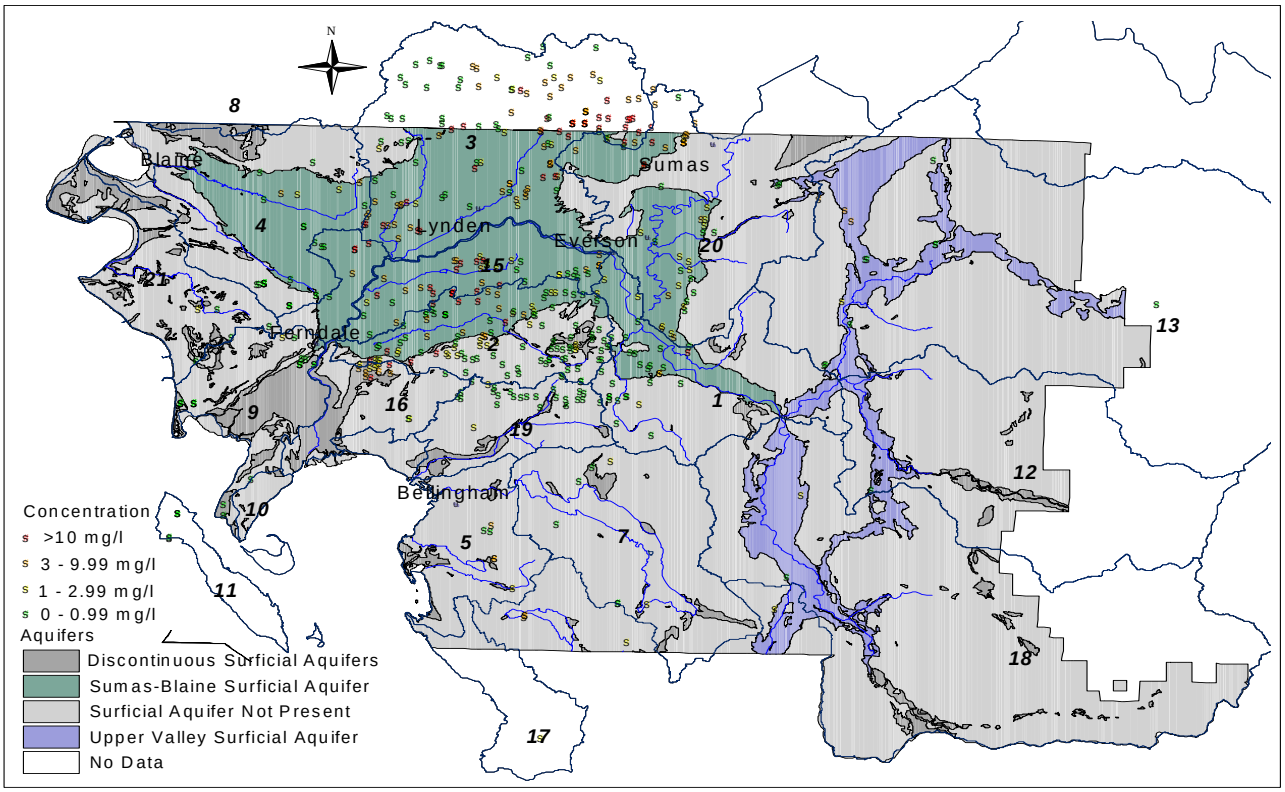

Figure C-1. Areal distribution of maximum [NO3+NO2] concentration for 1990.

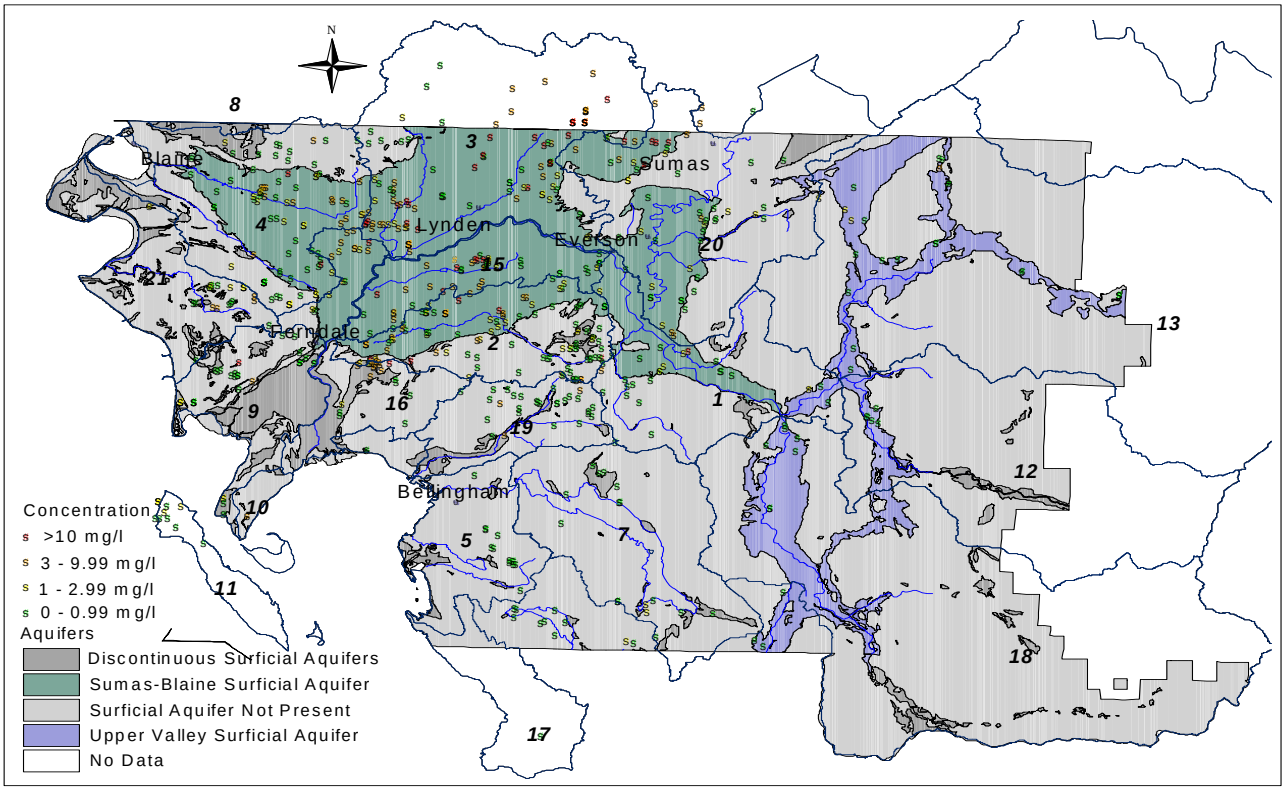

Figure C-2. Areal distribution of maximum [NO3+NO2] concentrations for 1991.

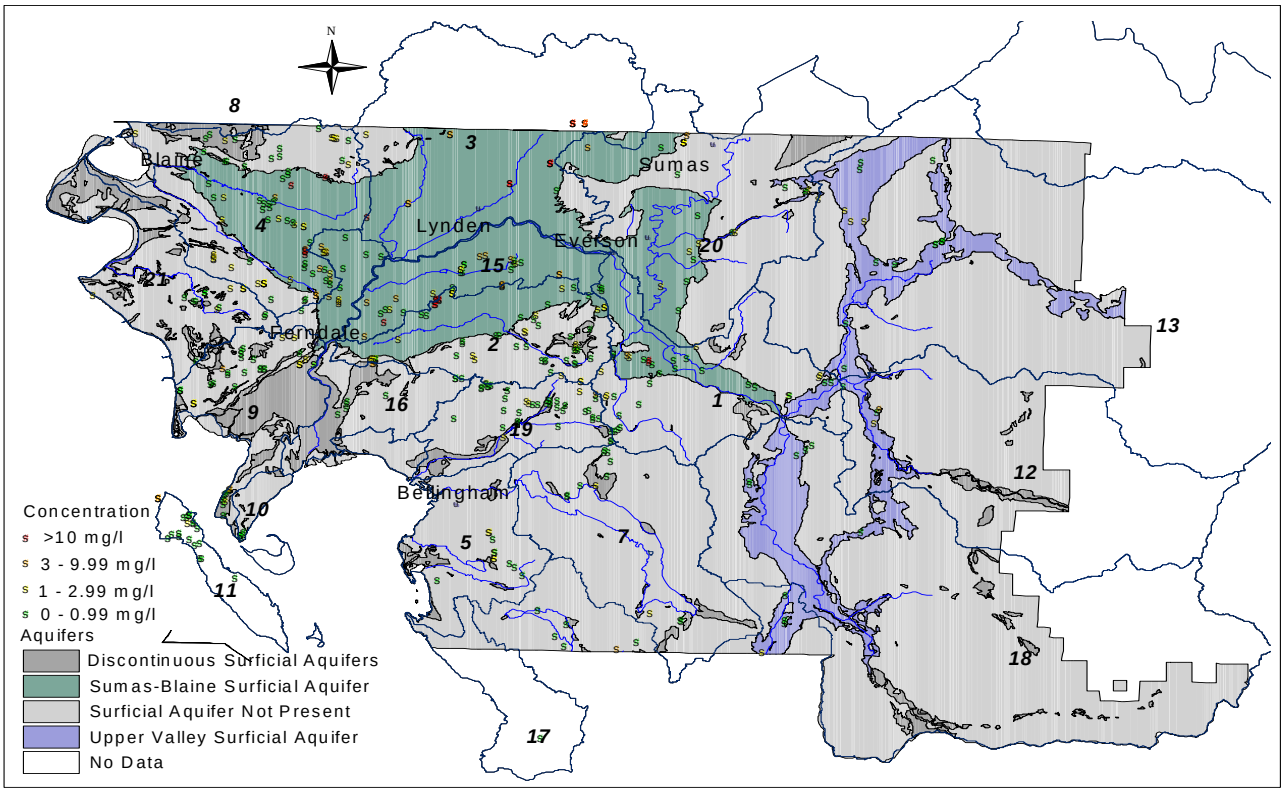

Figure C-3. Areal distribution of maximum [NO3+NO2] concentrations for 1992.

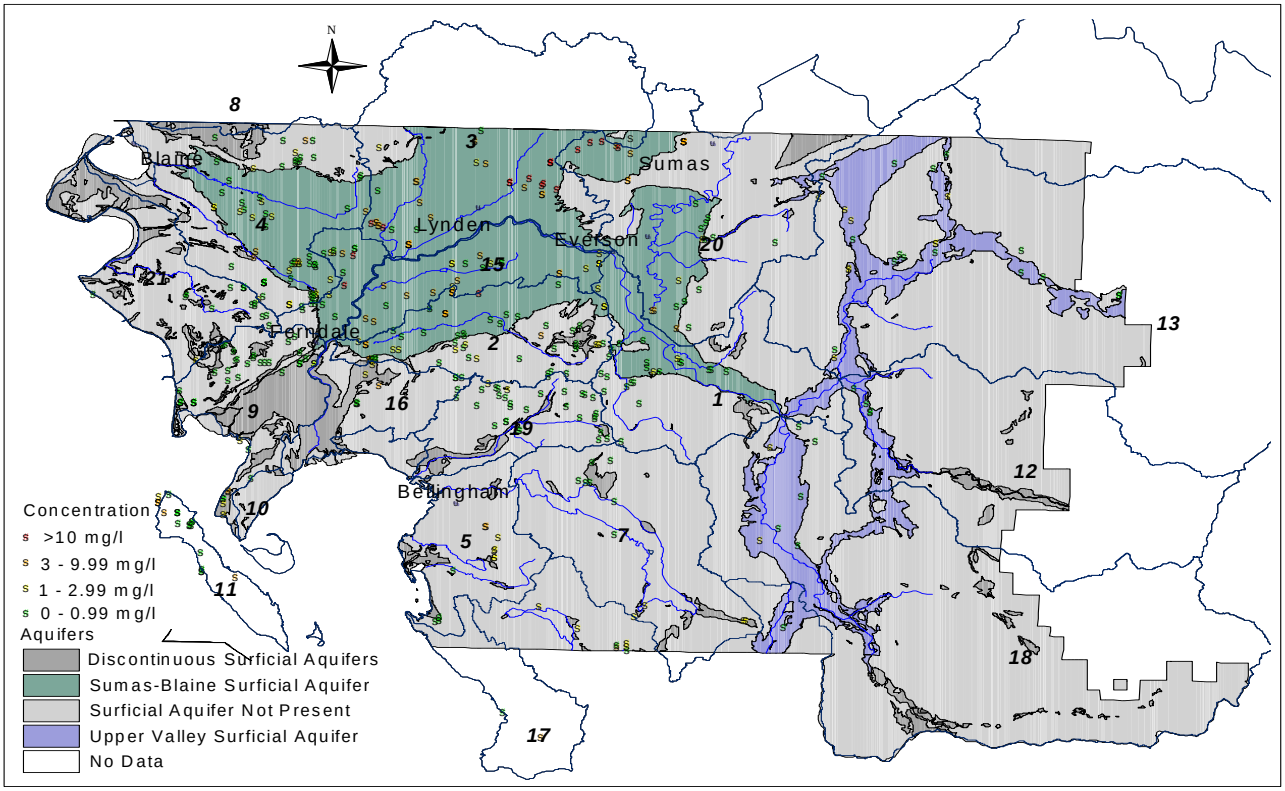

Figure C-4. Areal distribution of maximum [NO3+NO2] concentrations for 1993.

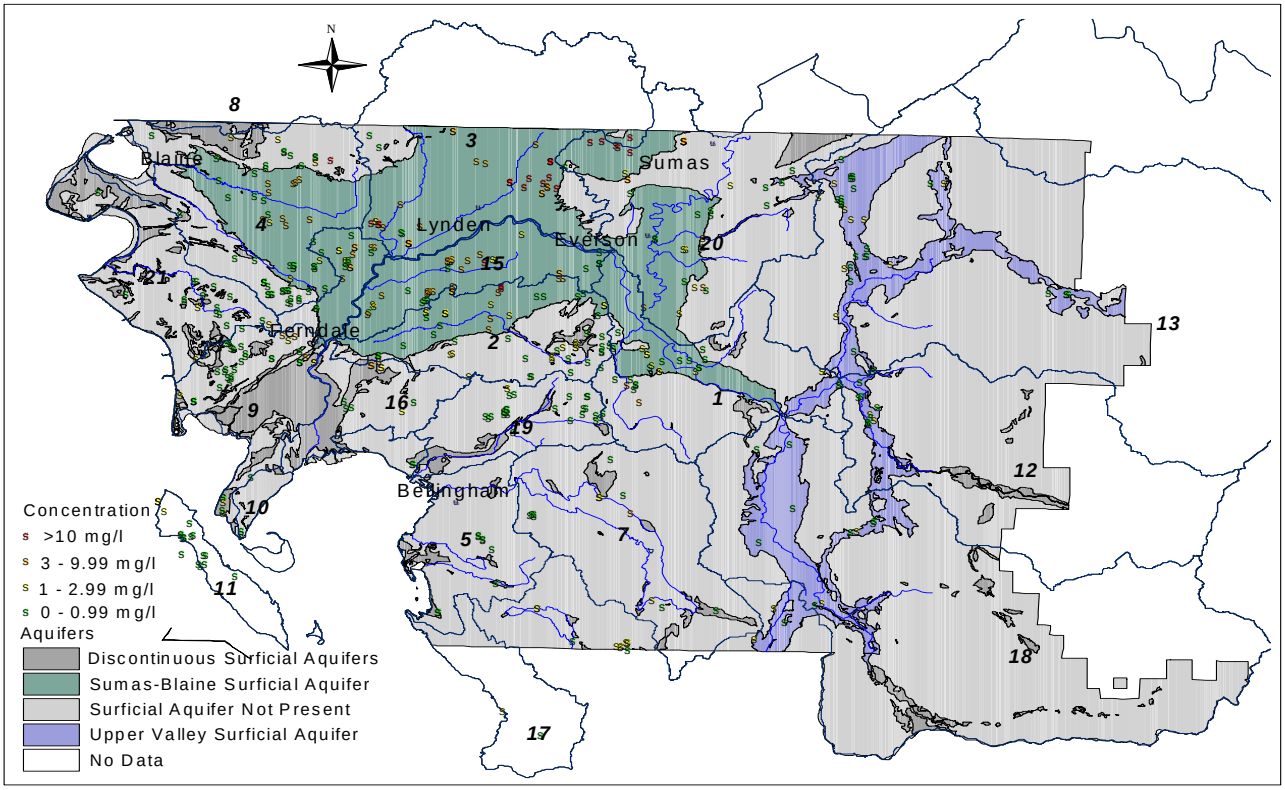

Figure C-5. Areal distribution of maximum [NO3+NO2] concentrations for 1994.

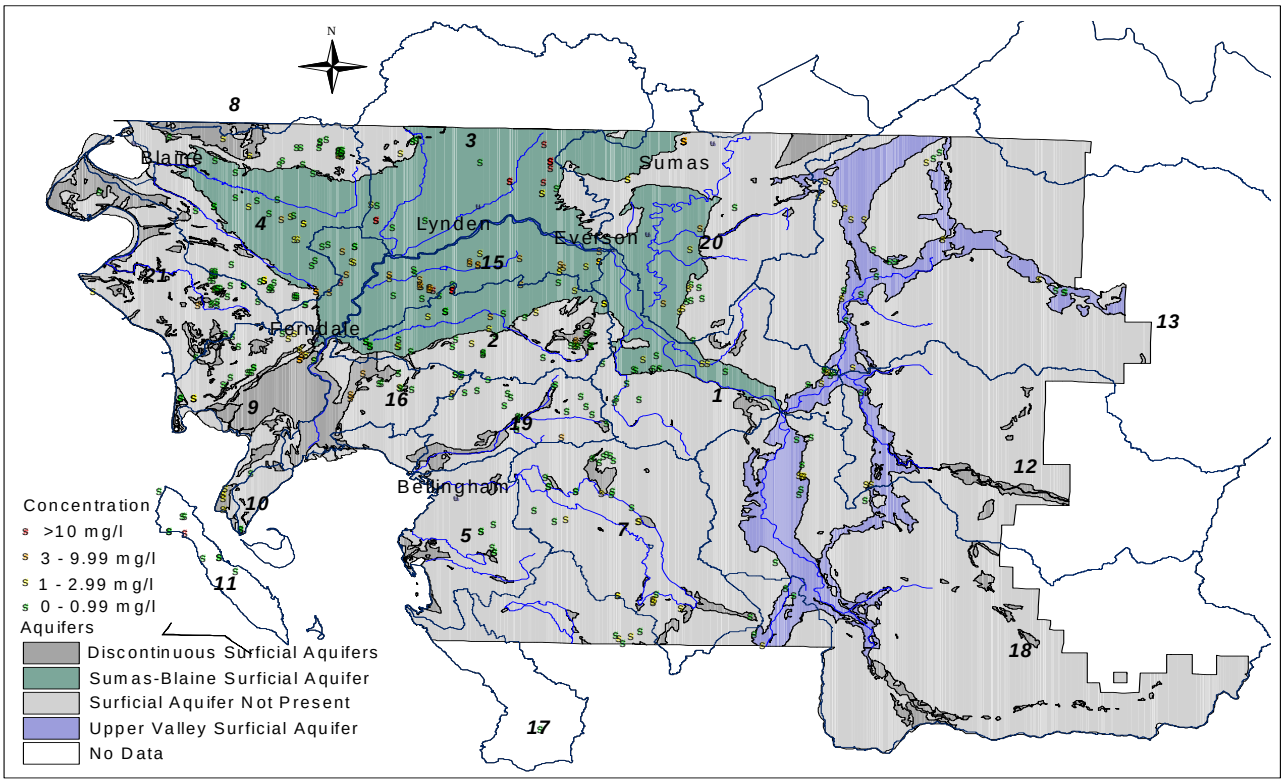

Figure C-6. Areal distribution of maximum [NO3+NO2] concentrations for 1995.

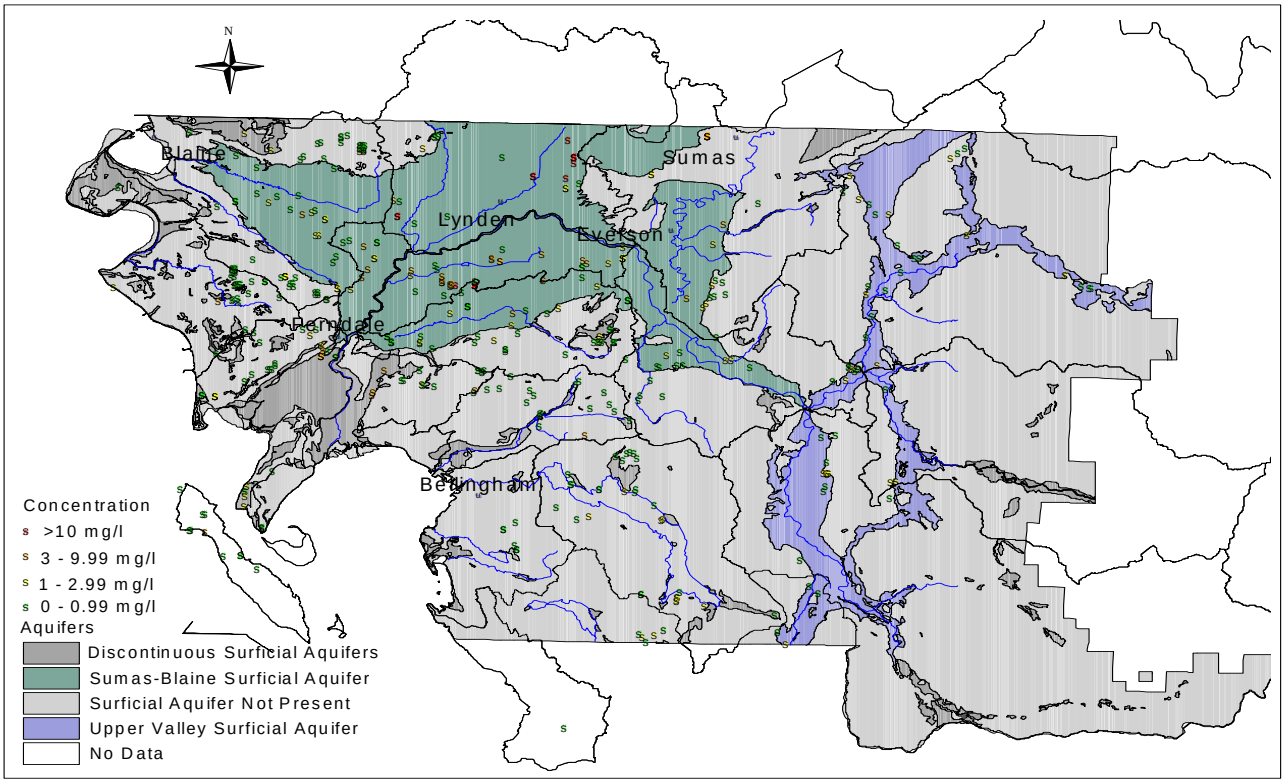

Figure C-7. Areal distribution of maximum [NO3+NO2] concentrations for 1996.

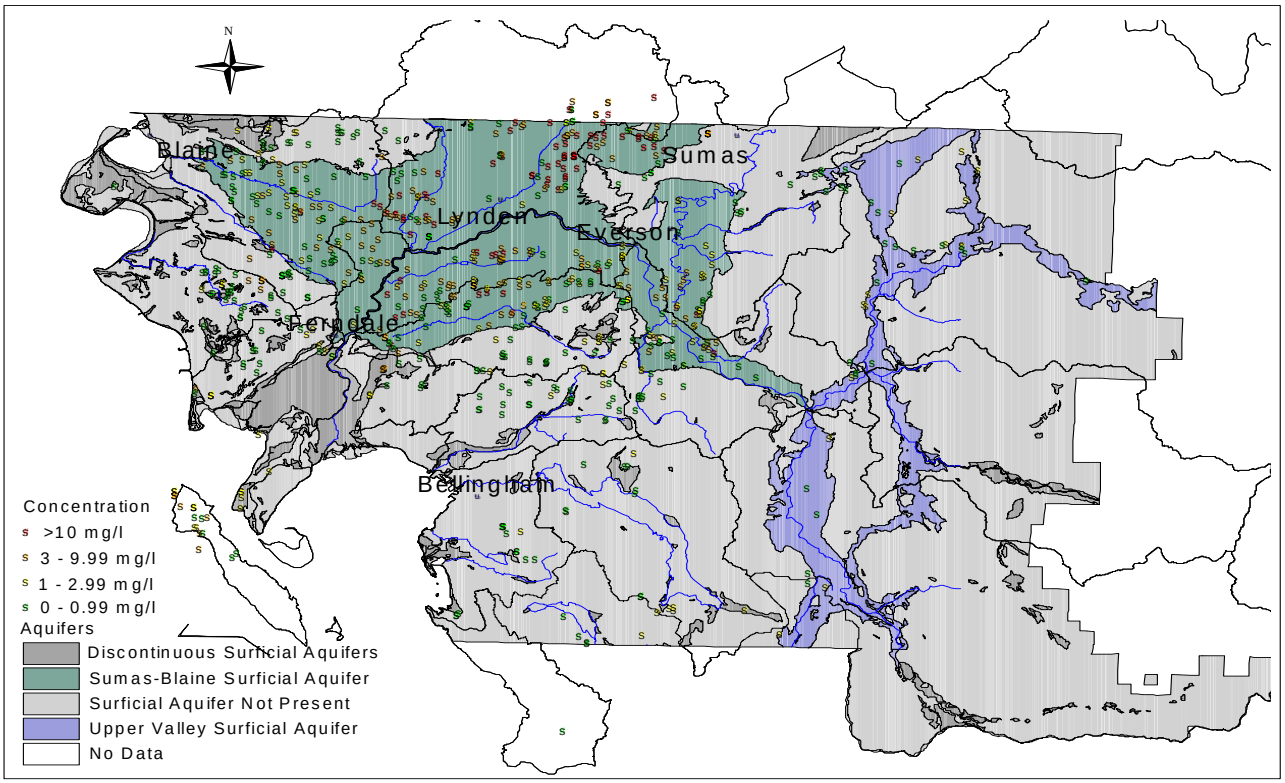

Figure C-8. Areal distribution of maximum [NO3+NO2] concentrations for 1997.

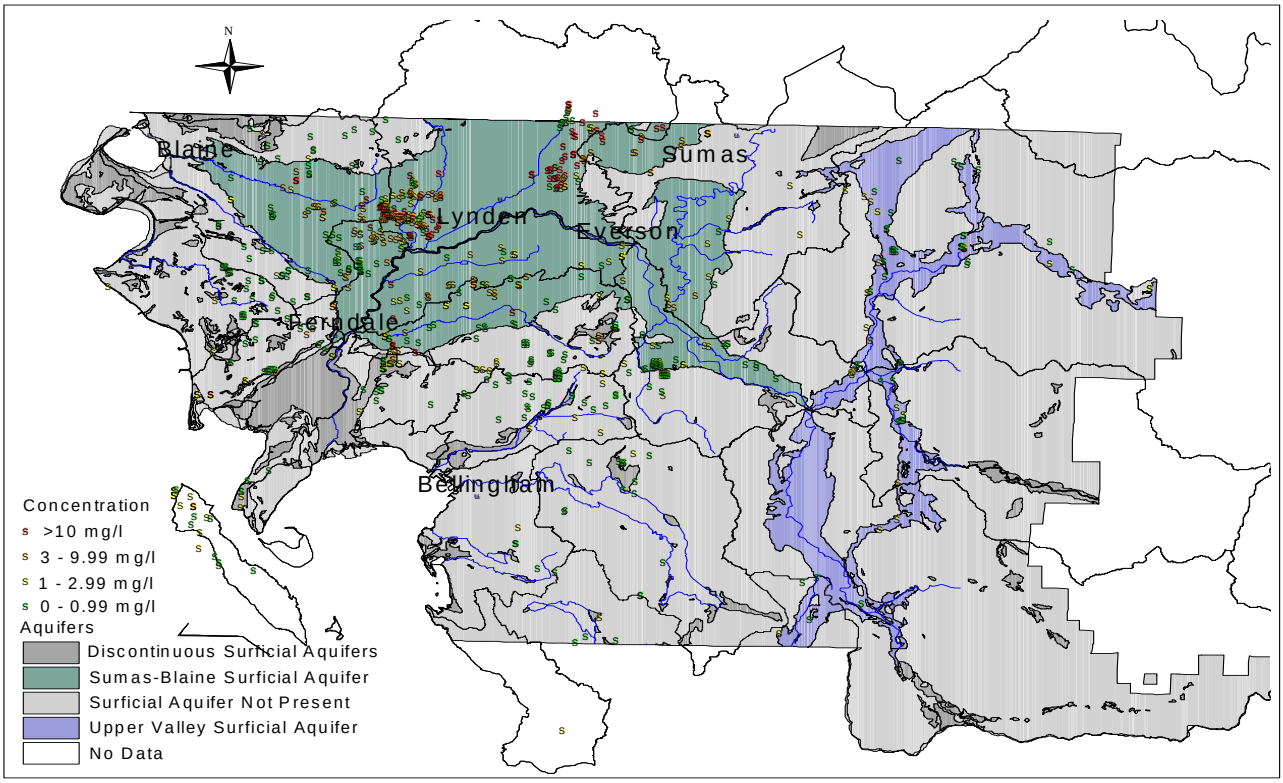

Figure C-9. Areal distribution of maximum [NO3+NO2] concentrations for 1998.

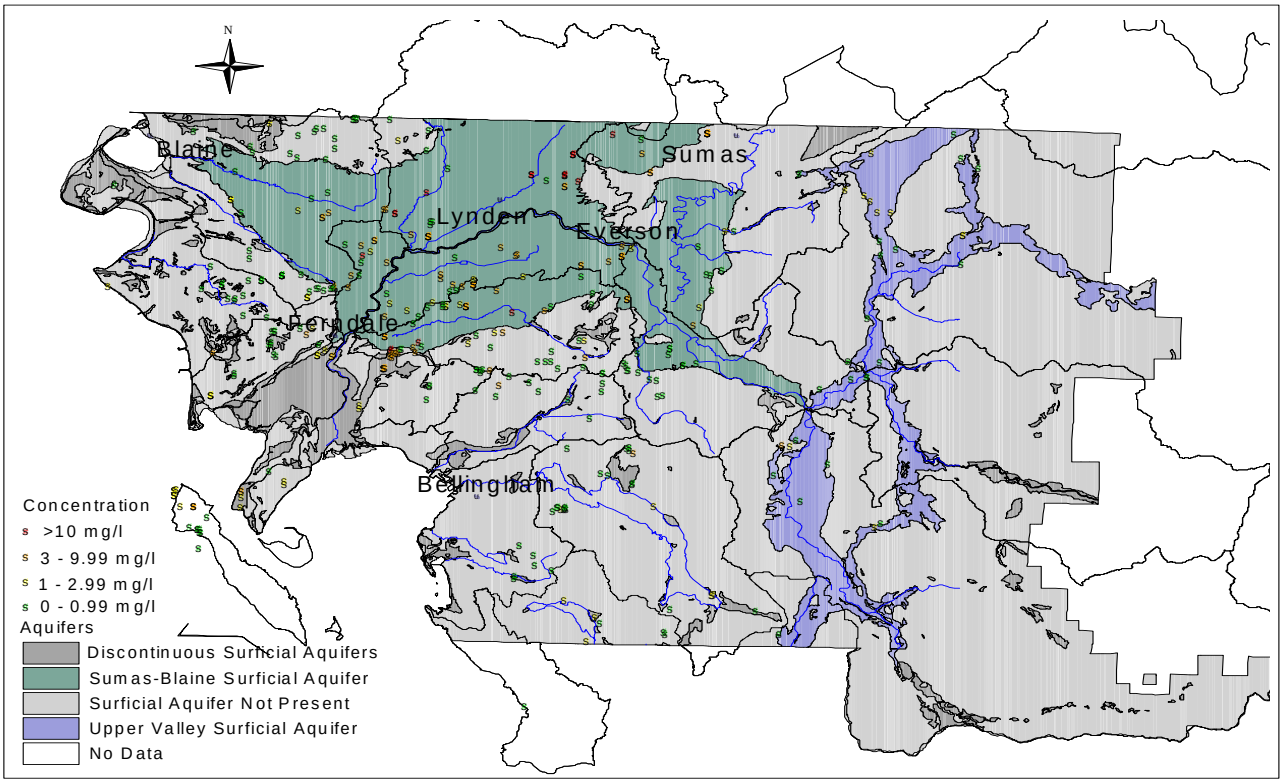

Figure C-10. Areal distribution of maximum [NO3+NO2] concentrations for 1999.

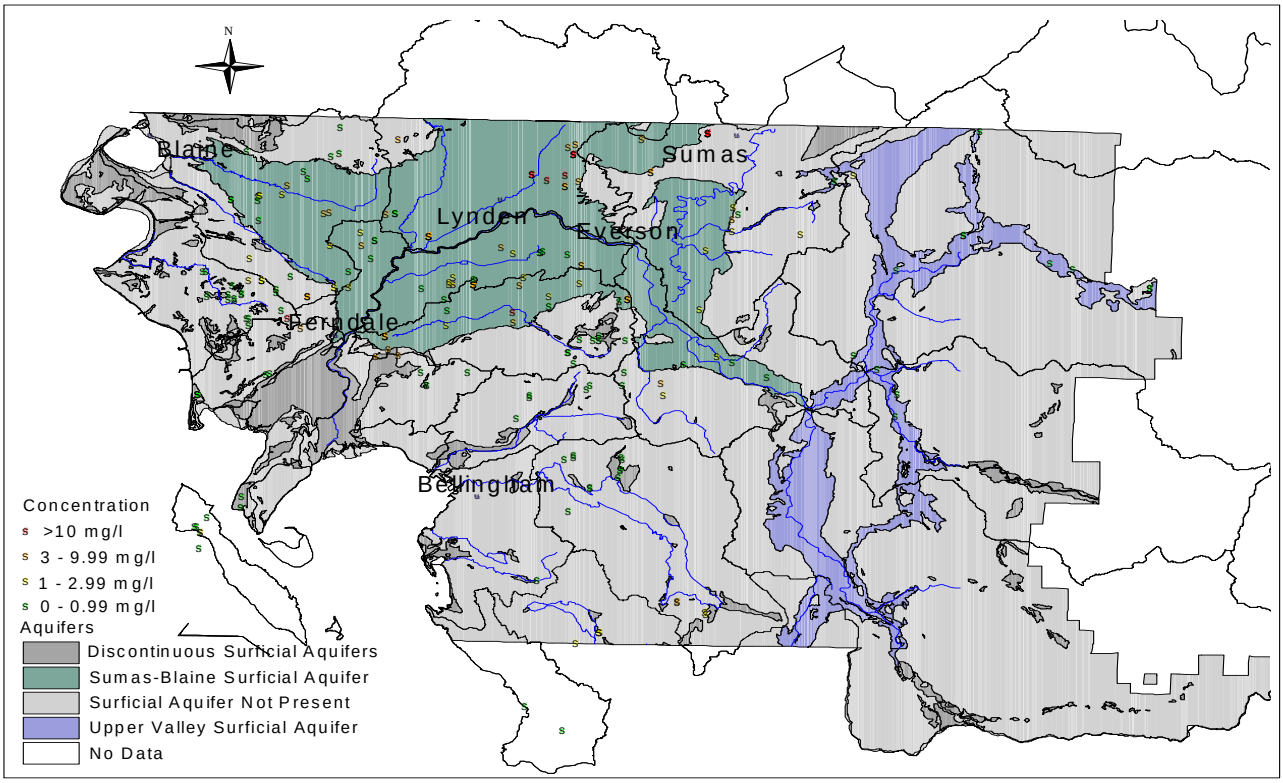

Figure C-11. Areal distribution of maximum [NO3+NO2] concentrations for 2000.

APPENDIX D - AREAL DISTRIBUTION OF ANNUAL MEDIAN [NO3+NO2] CONCENTRATIONS IN WRIA 1

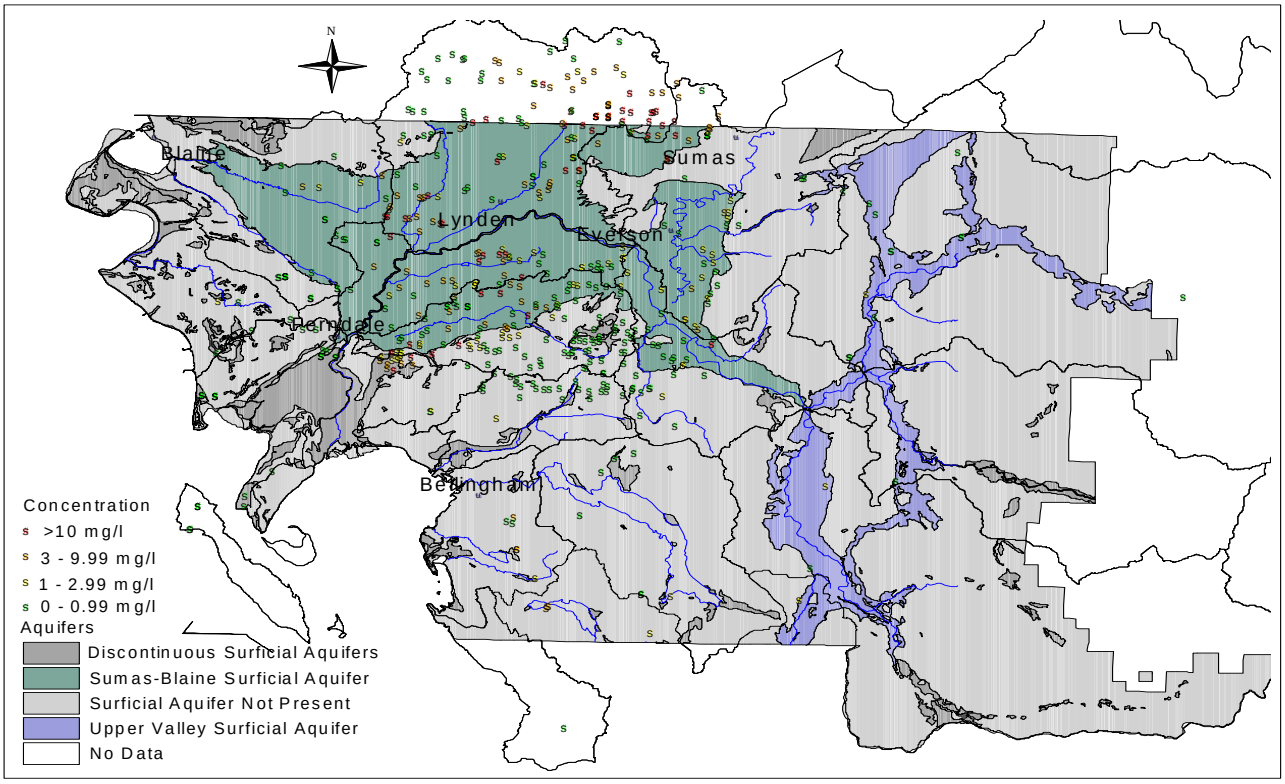

Figure D-1. Areal distribution of median [NO3+NO2] concentrations for 1990.

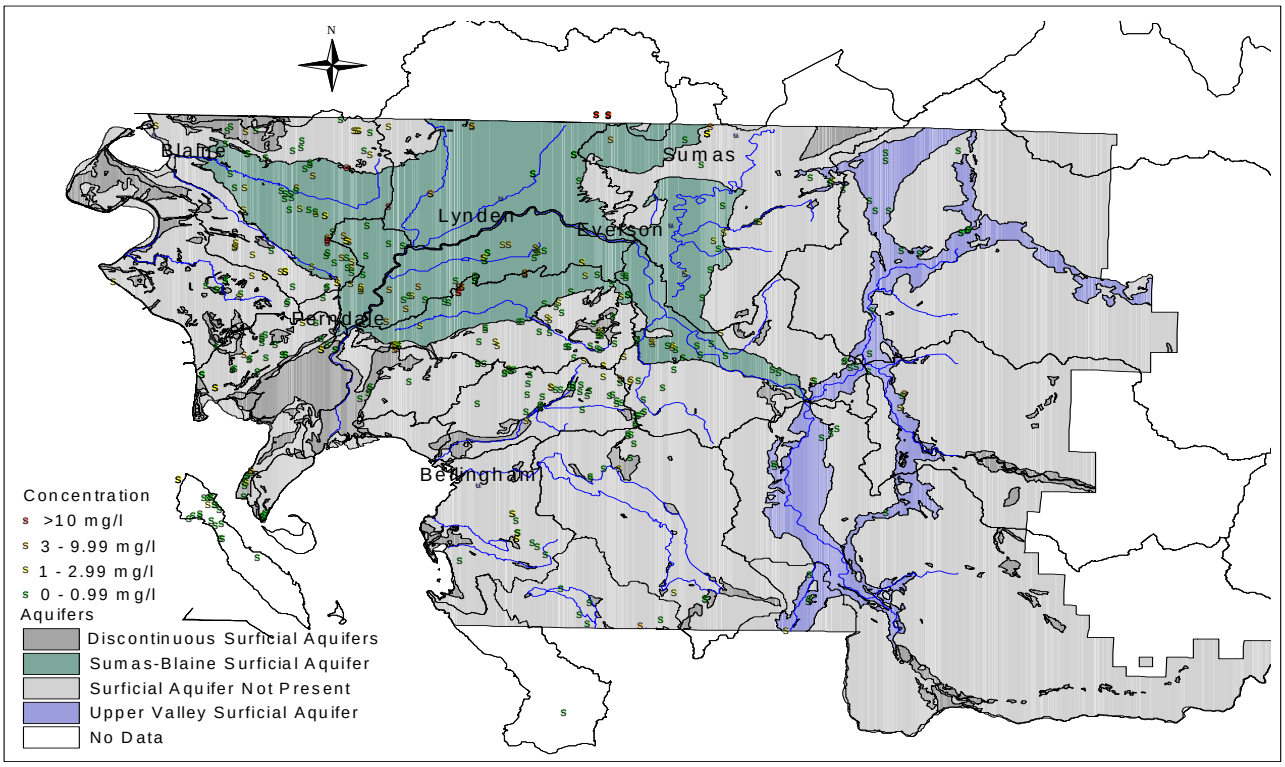

Figure D-2. Areal distribution of median [NO3+NO2] concentrations for 1991.

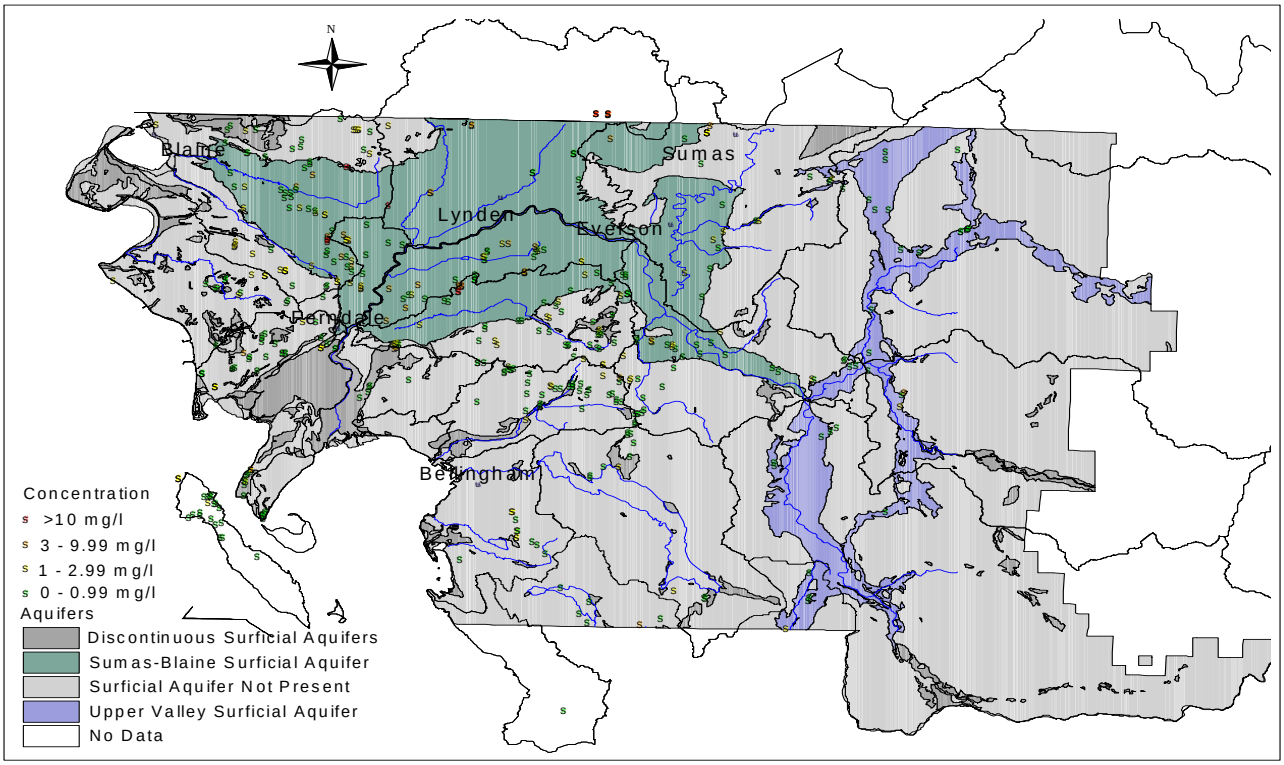

Figure D-3. Areal distribution of median [NO3+NO2] concentrations for 1992.

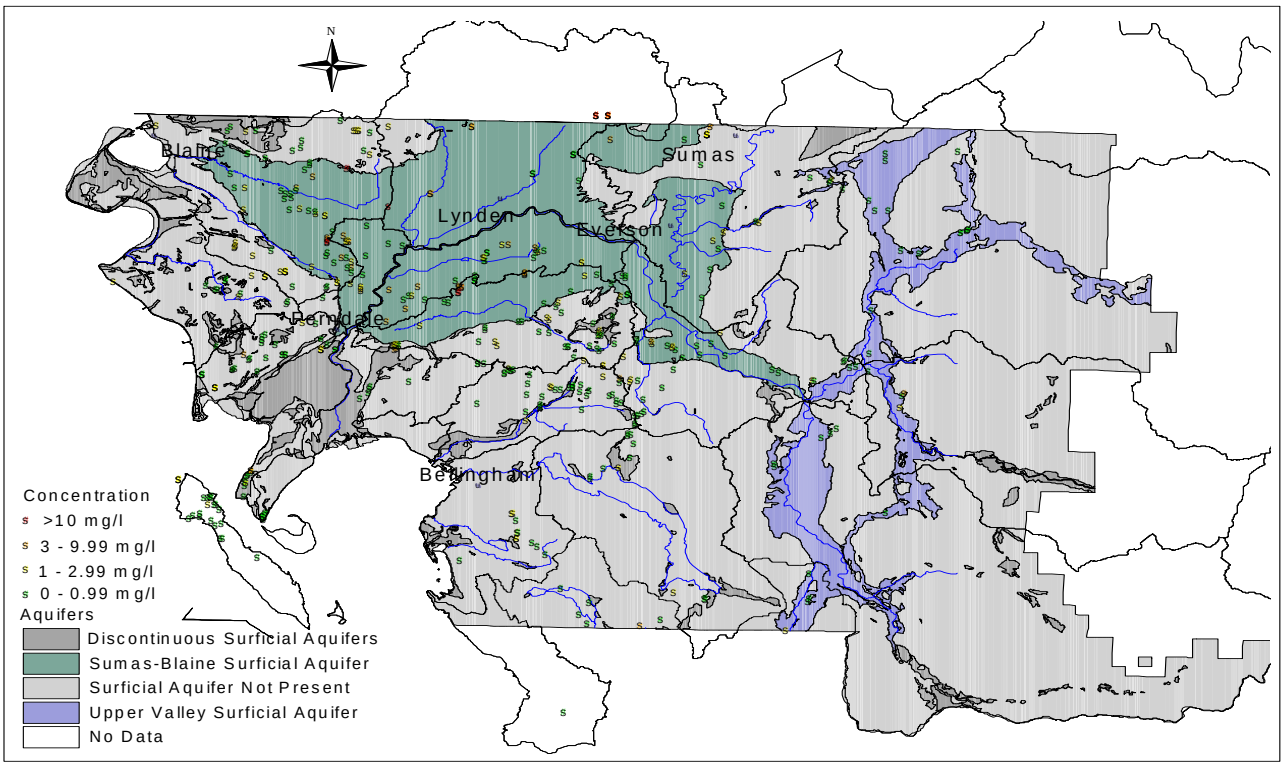

Figure D-4. Areal distribution of median [NO3+NO2] concentrations for 1993.

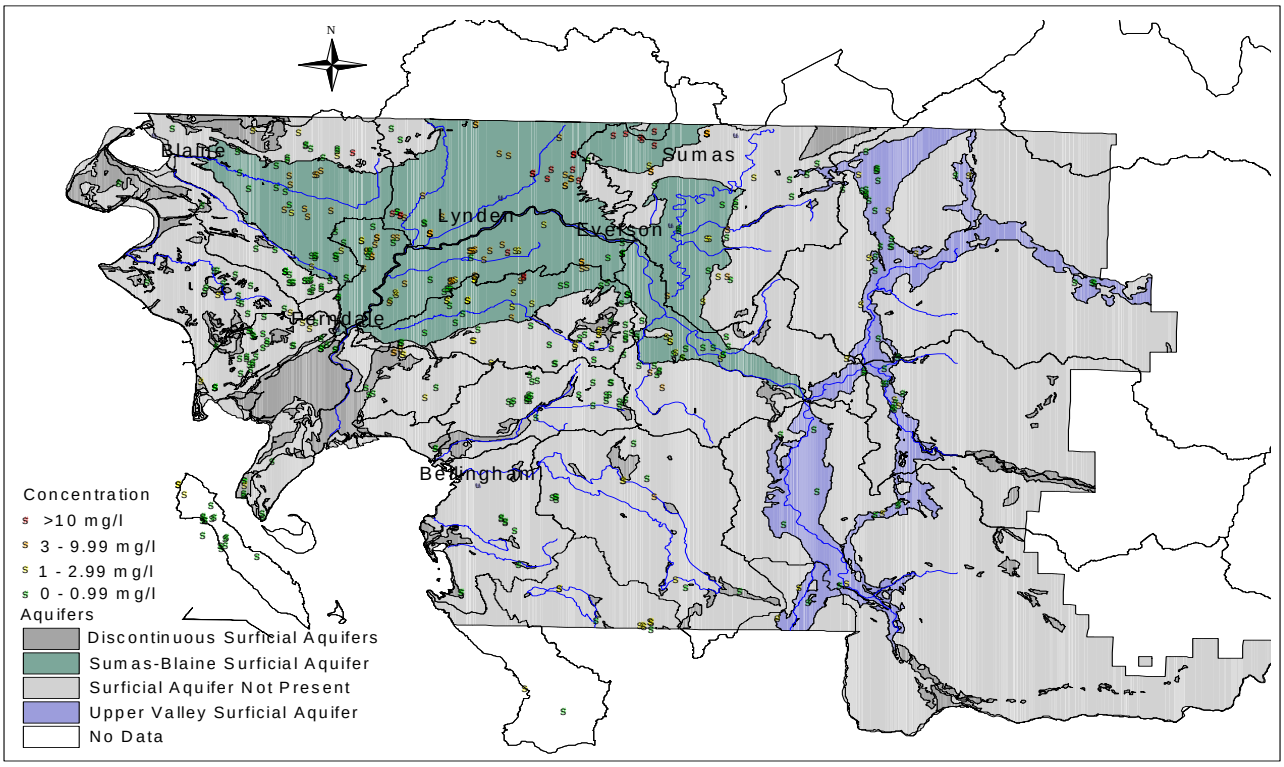

Figure D-5. Areal distribution of median [NO3+NO2] concentrations for 1994.

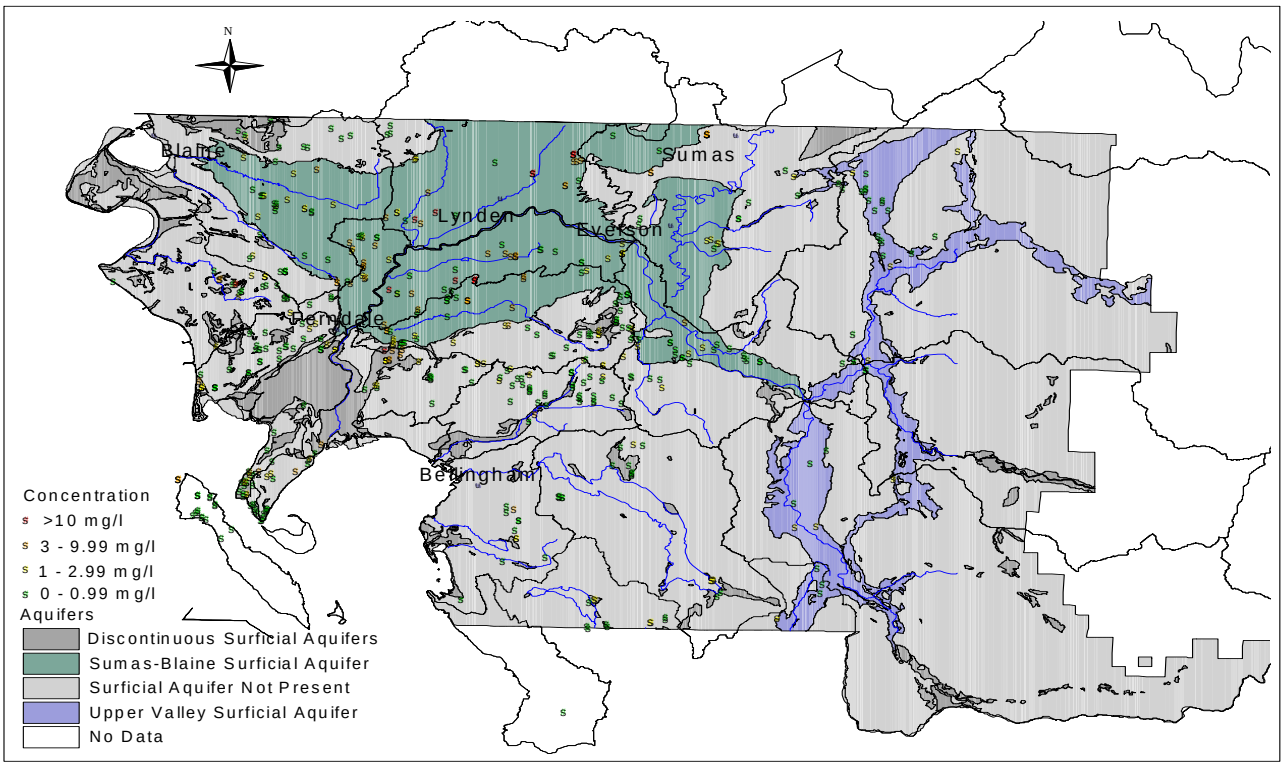

Figure D-6. Areal distribution of median [NO3+NO2] concentrations for 1995.

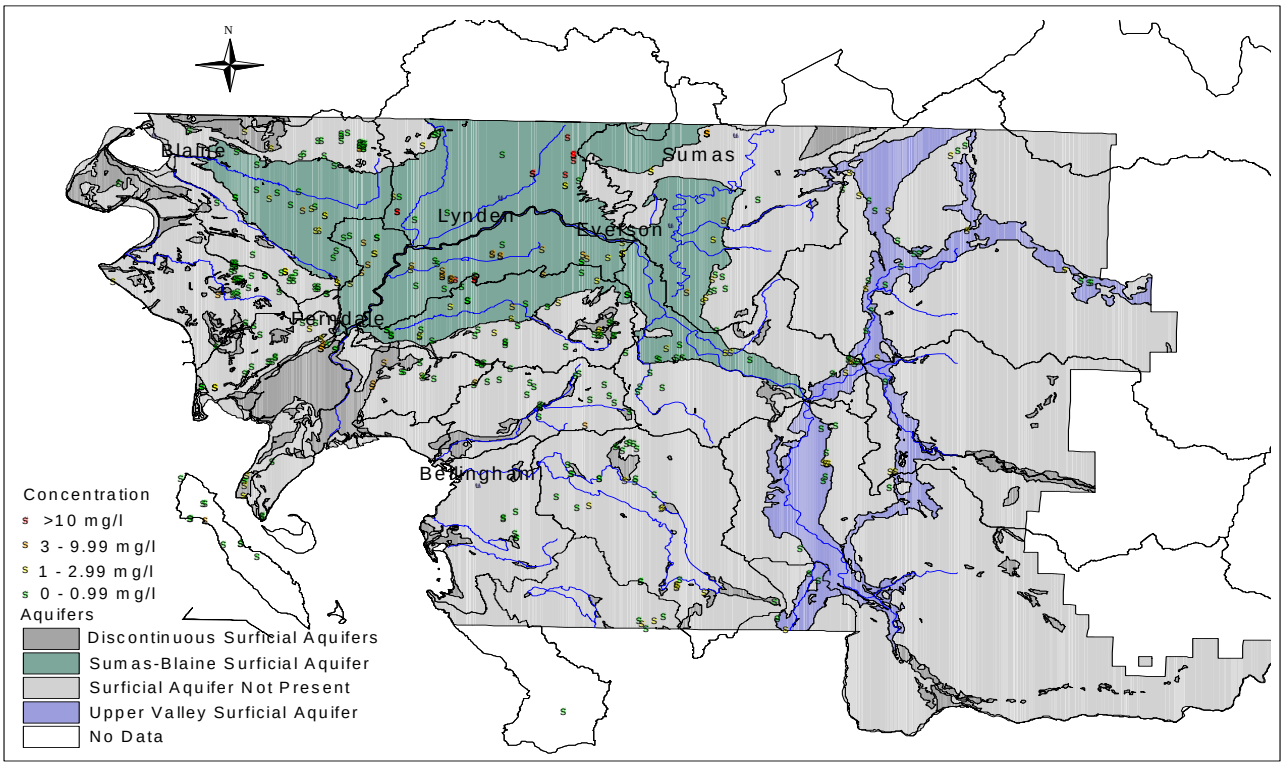

Figure D-7. Areal distribution of median [NO3+NO2] concentrations for 1996.

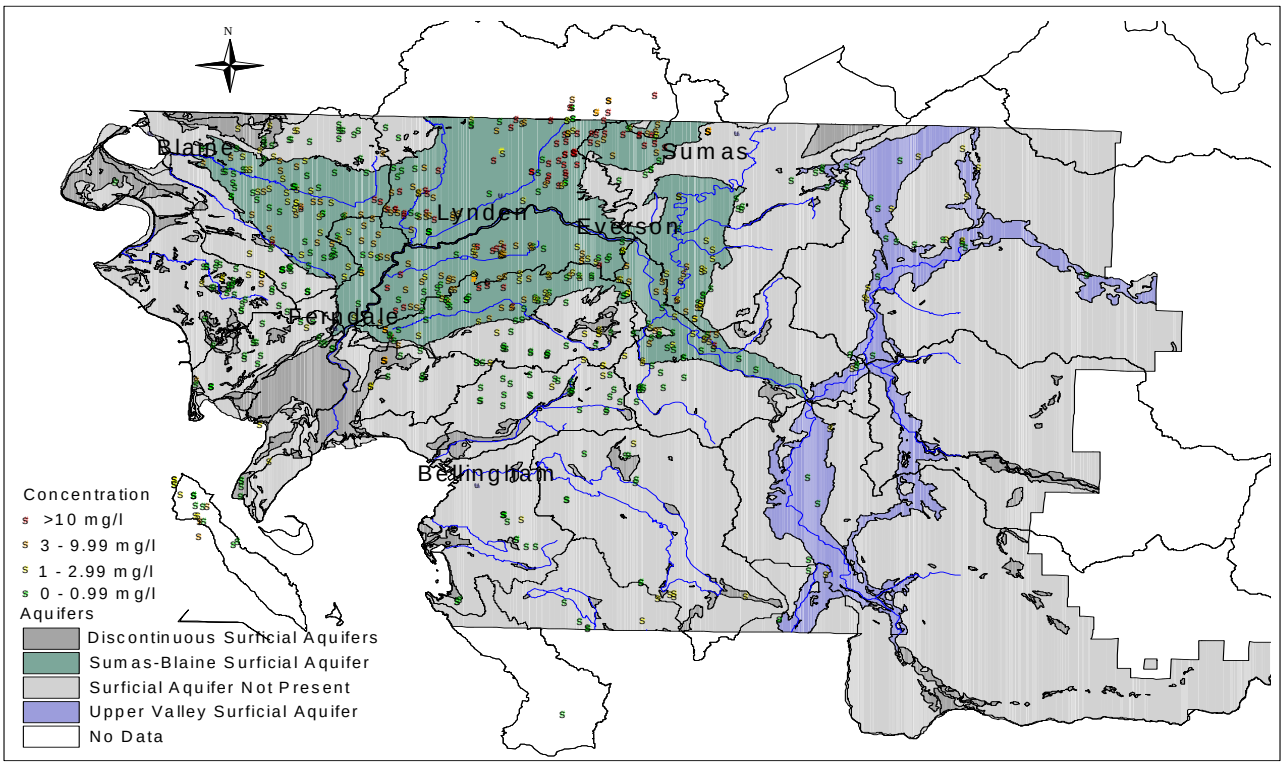

Figure D-8. Areal distribution of median [NO3+NO2] concentrations for 1997.

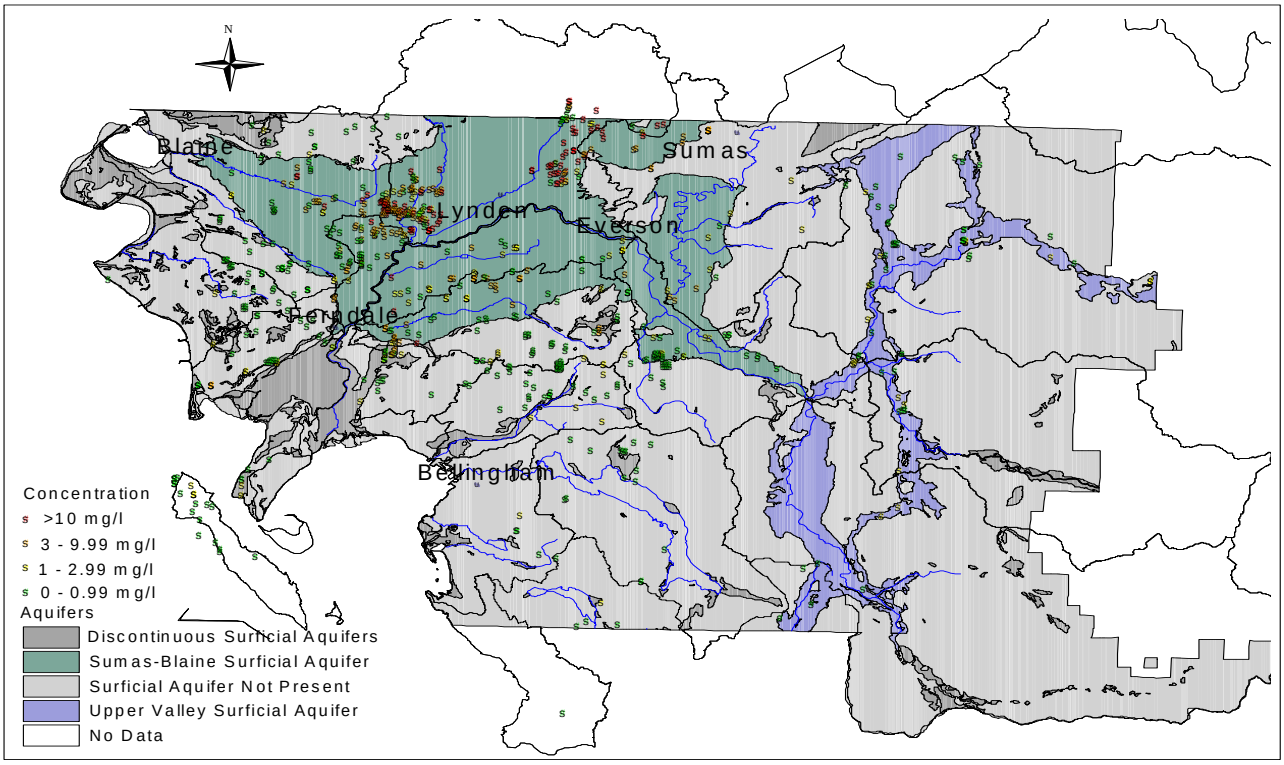

Figure D-9. Areal distribution of median [NO3+NO2] concentrations for 1998.

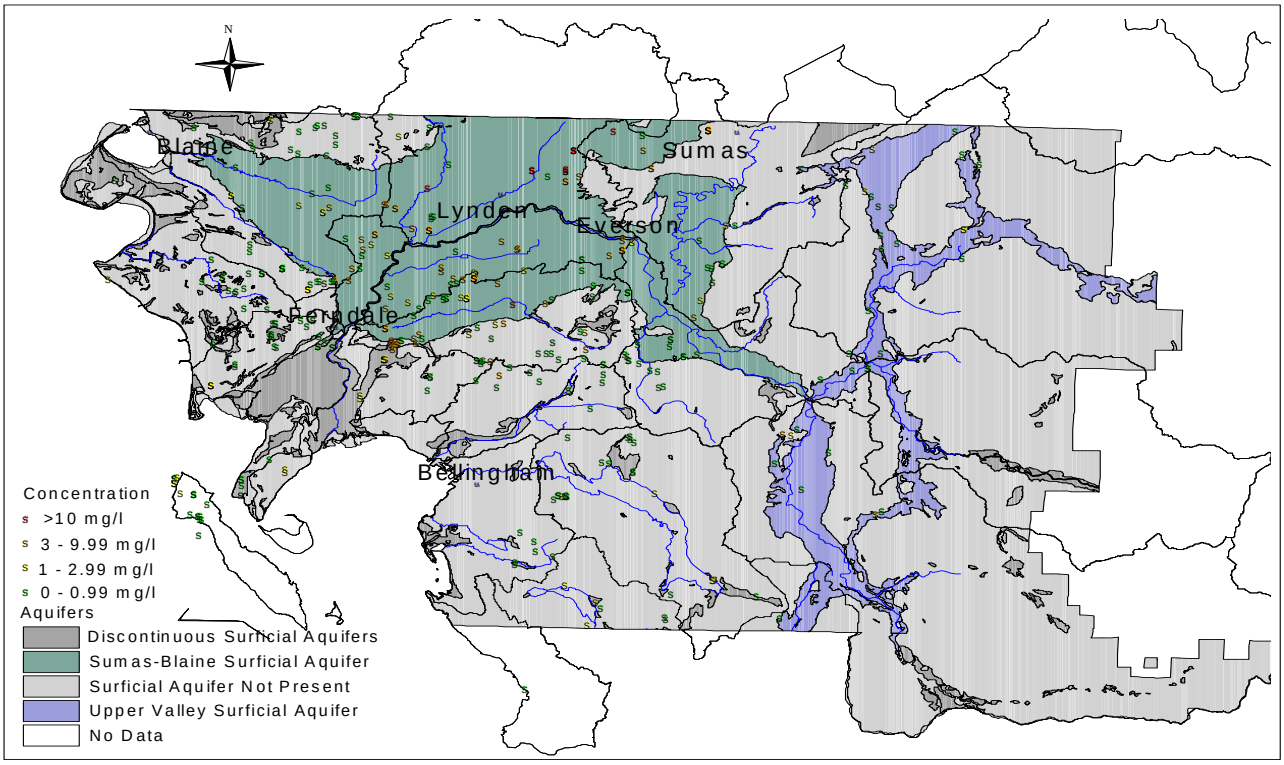

Figure D-10. Areal distribution of median [NO3+NO2] concentrations for 1999.

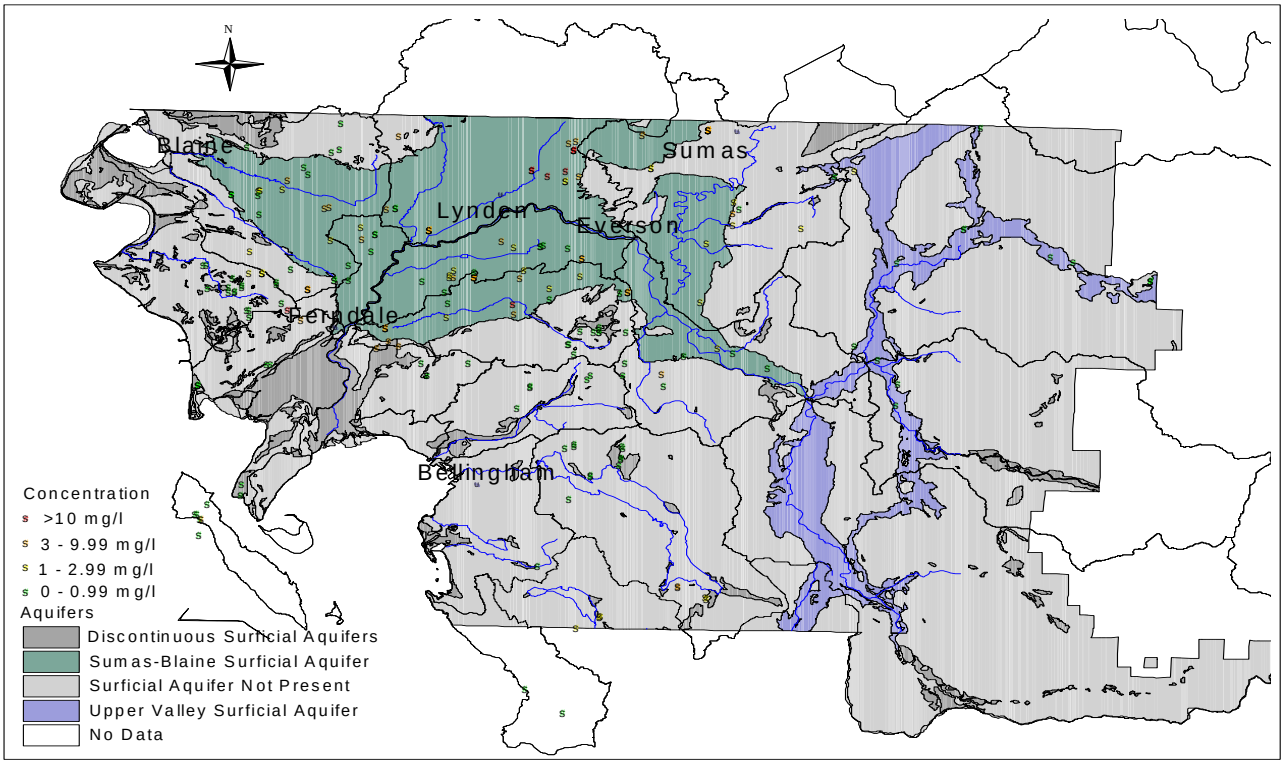

Figure D-11. Areal distribution of median [NO3+NO2] concentrations for 2000.

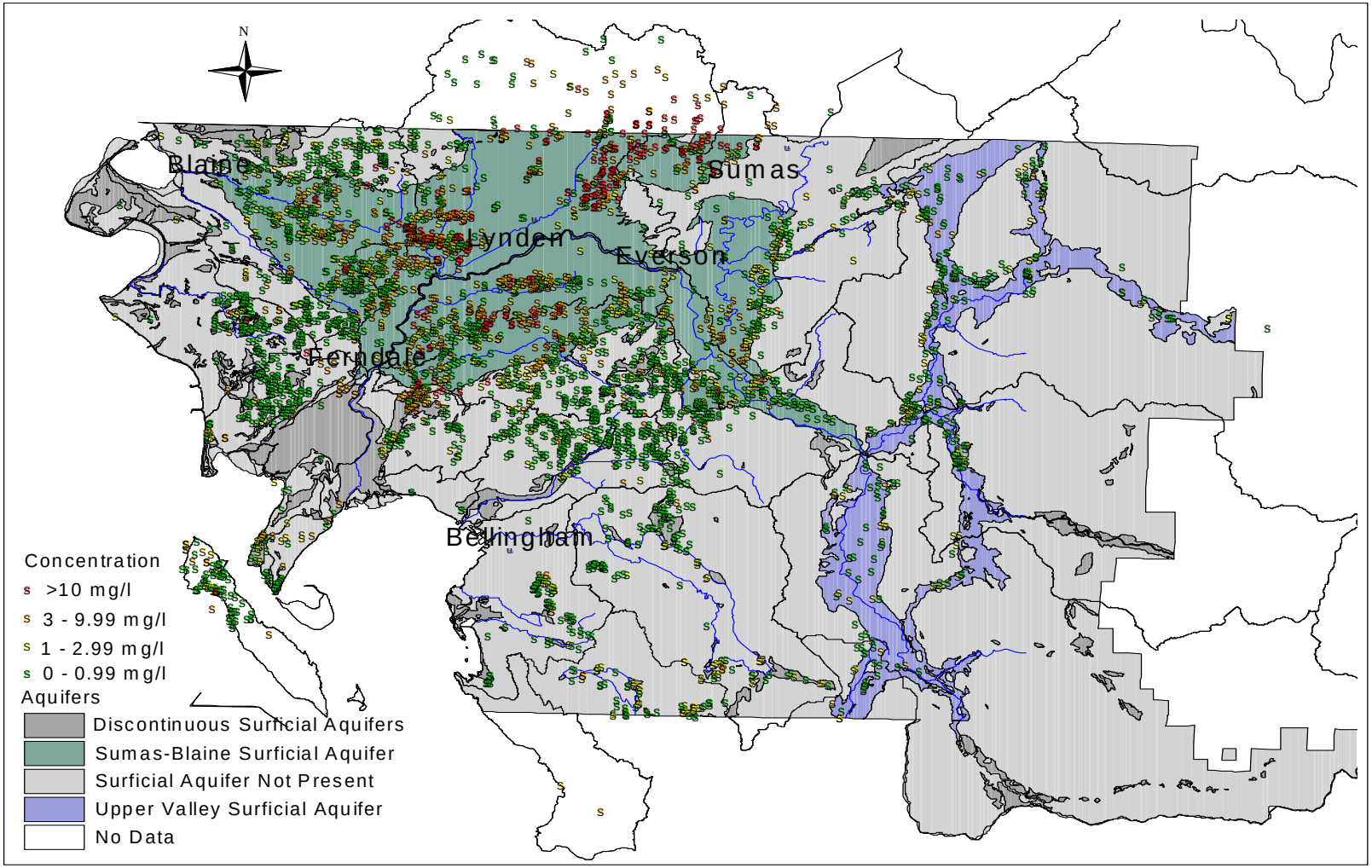

Figure D-12. Areal distribution of median [NO3+NO2] concentrations for the period 1990 - 2000.

APPENDIX E - AREAL DISTRIBUTION OF TOTAL AMMONIA CONCENTRATIONS IN WRIA 1

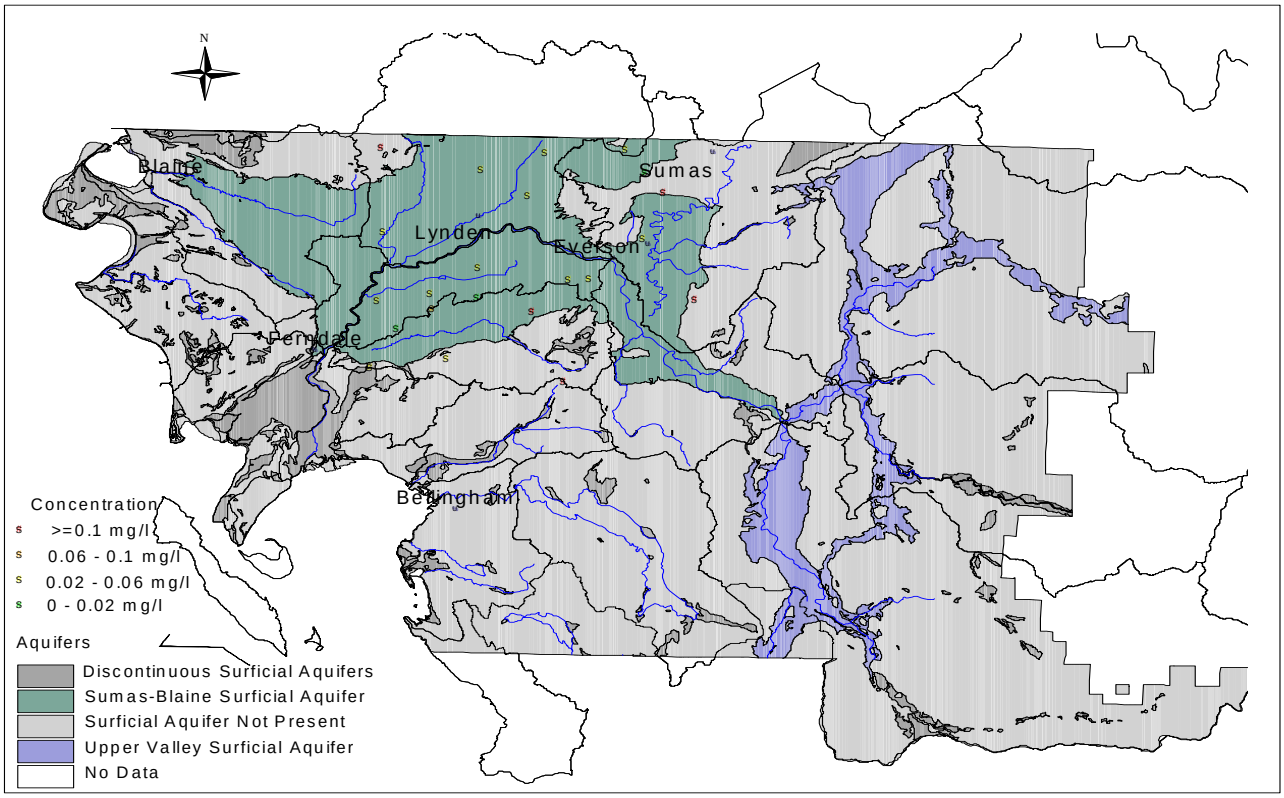

Figure E-1. Areal distribution of mean ammonia concentrations in 1990.

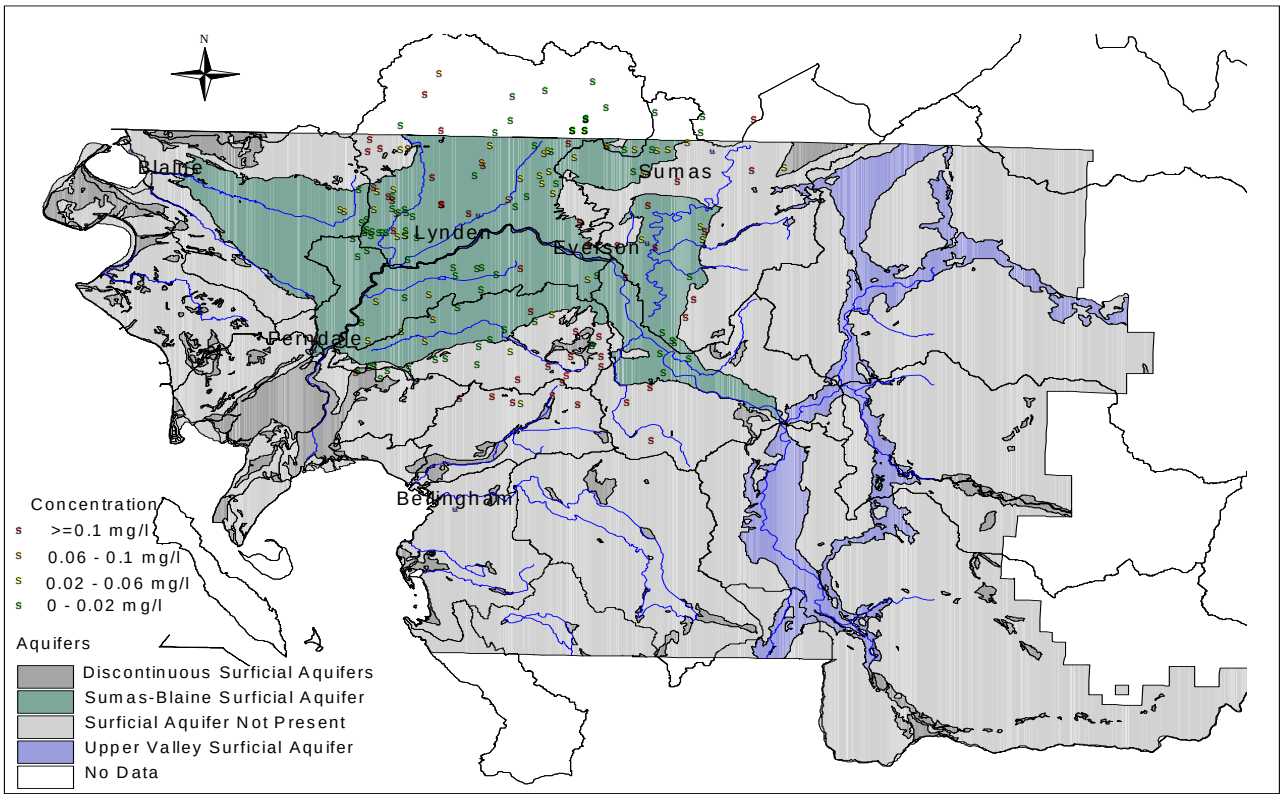

Figure E-2. Areal distribution of mean ammonia concentrations in wells – 1991.

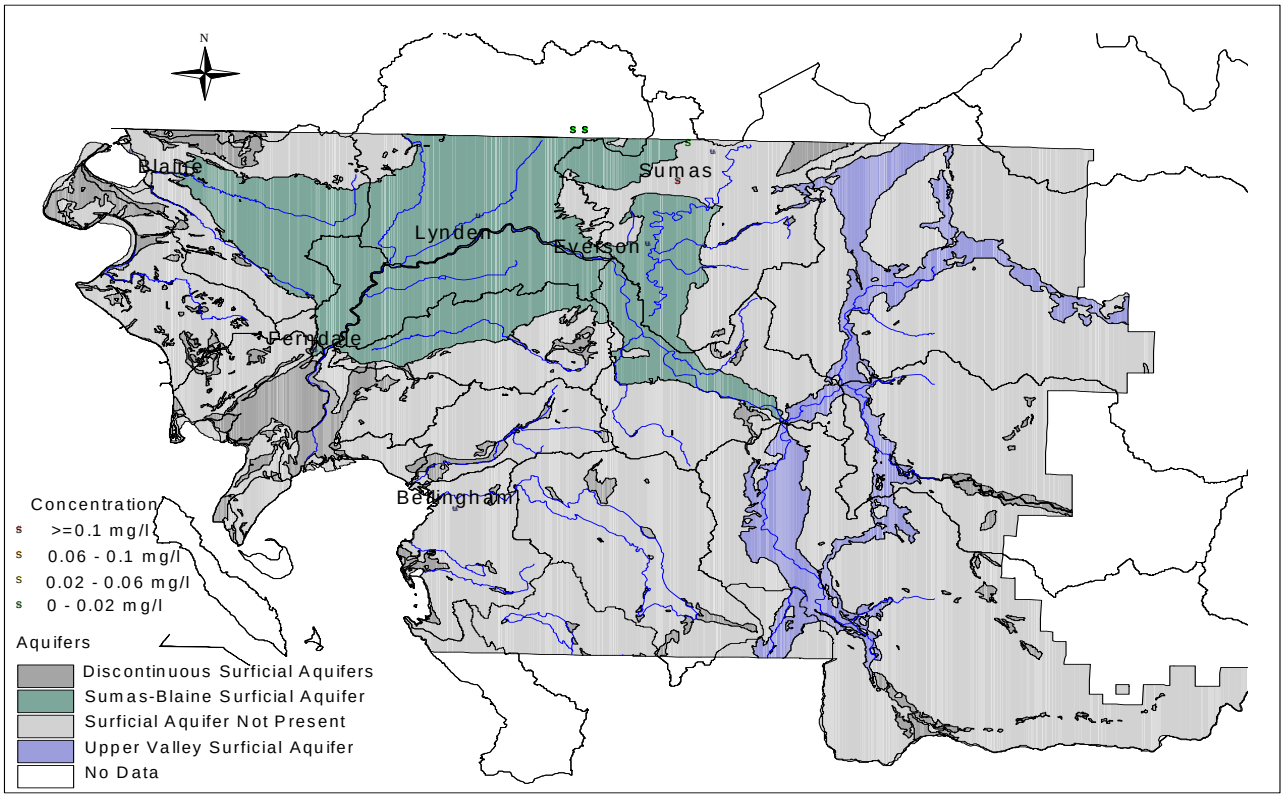

Figure E-3. Areal distribution of mean ammonia concentrations in 1992.

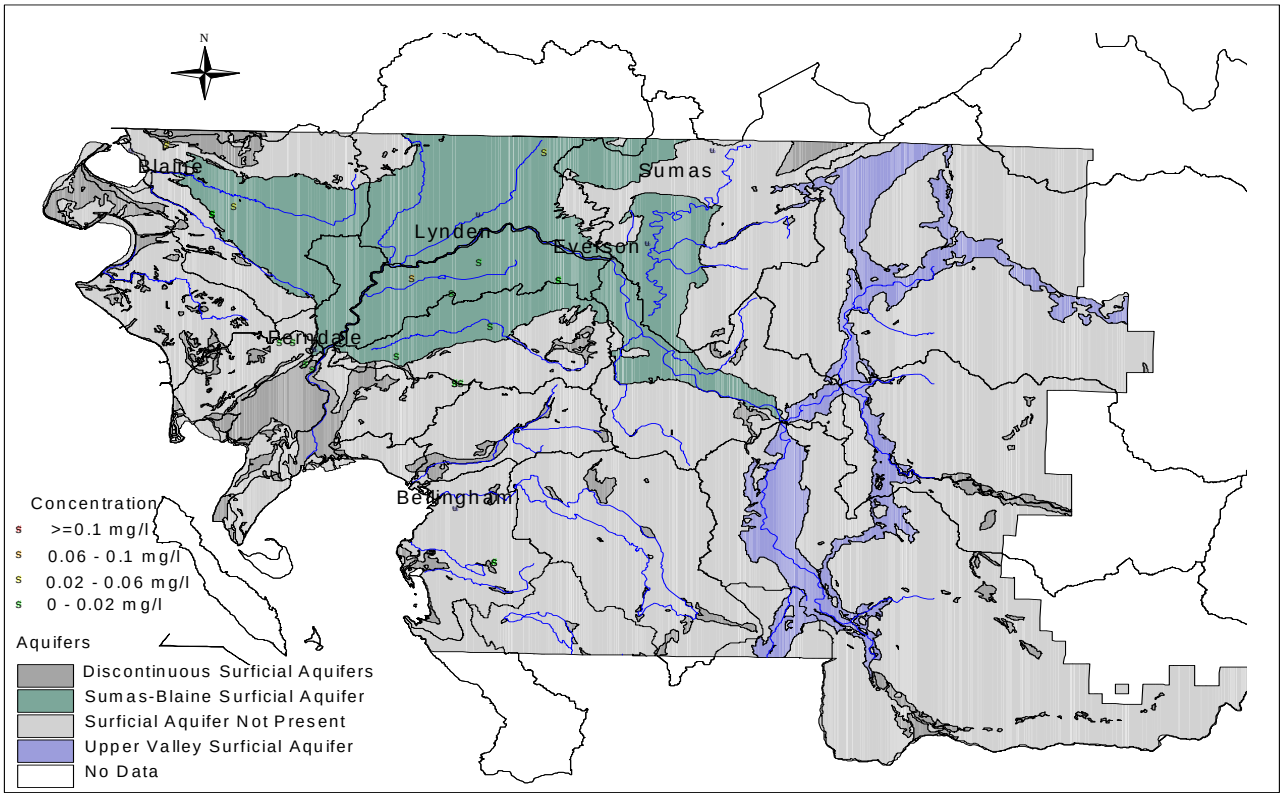

Figure E-4. Areal distribution of mean ammonia concentrations in 1996.

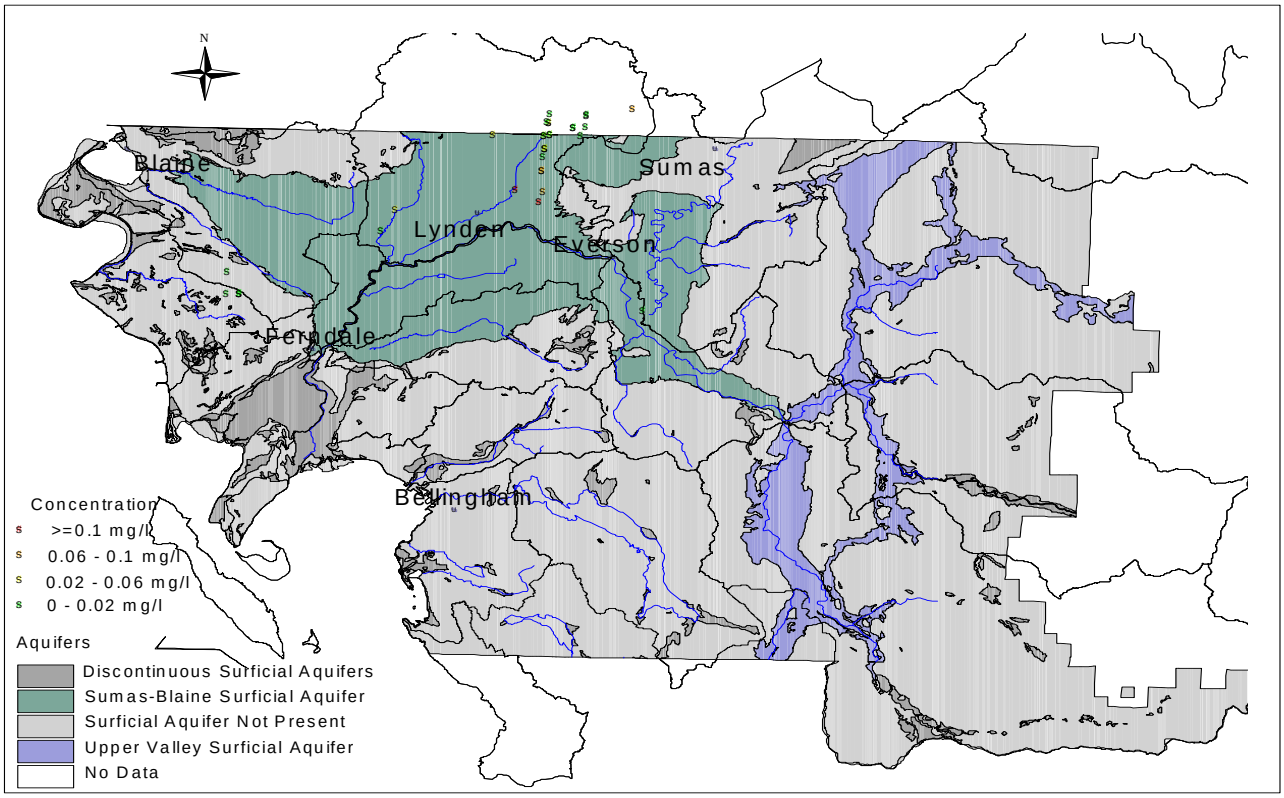

Figure E-5. Areal distribution of mean ammonia concentrations in 1997.

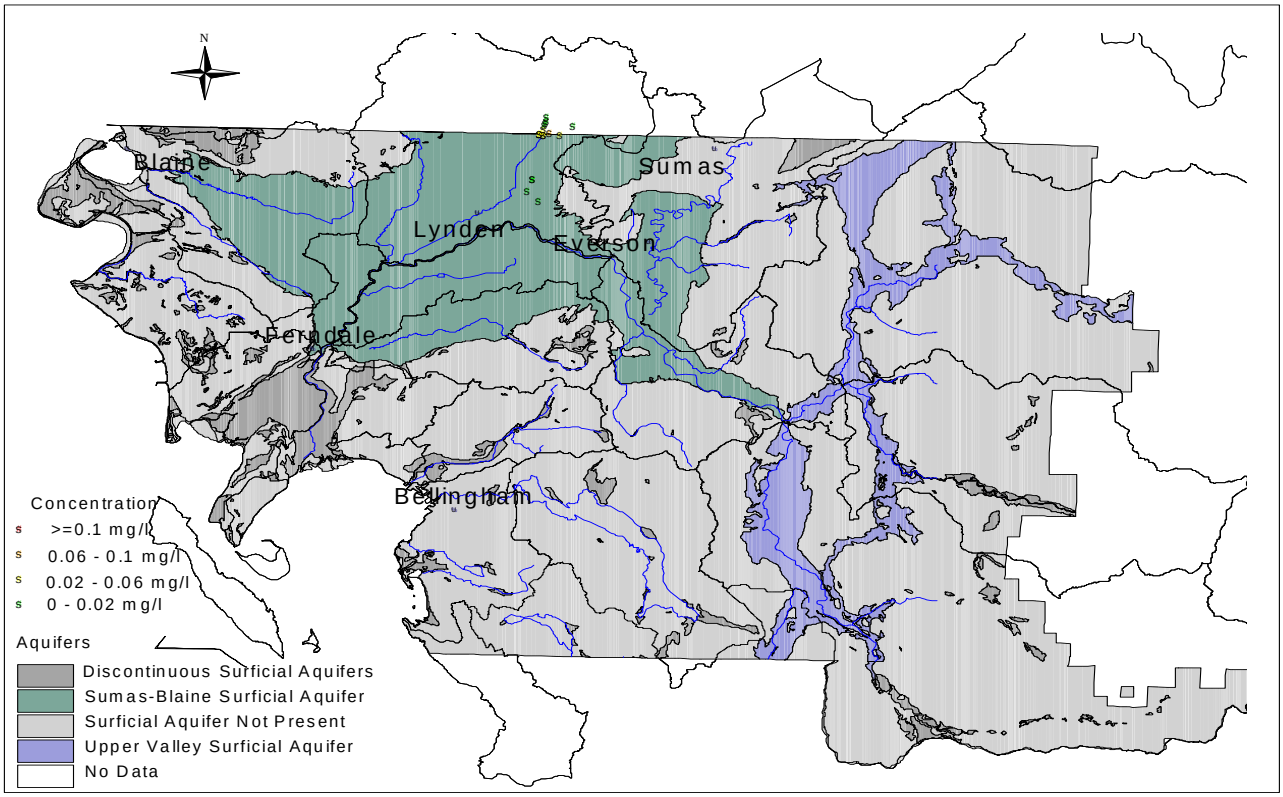

Figure E-6. Areal distribution of mean ammonia concentrations in 1998.

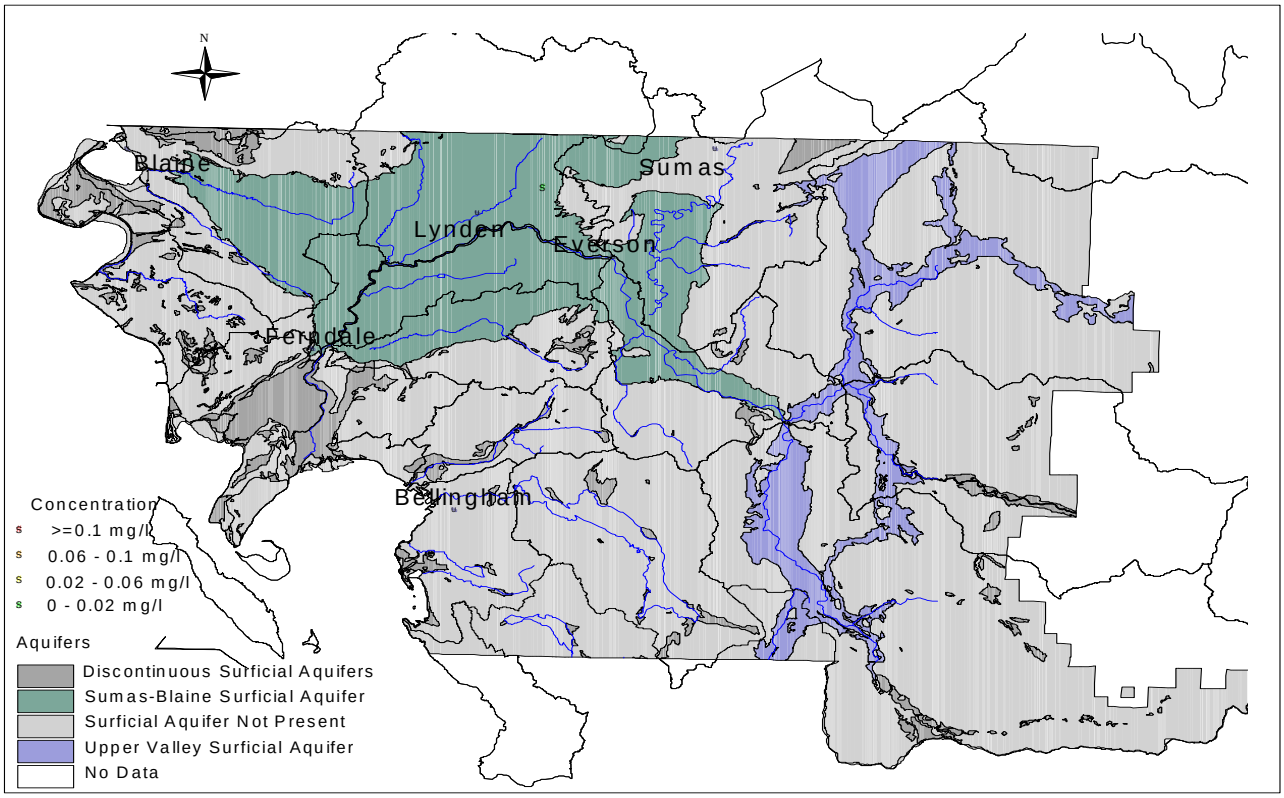

Figure E-7. Areal distribution of mean ammonia concentrations in 1999.

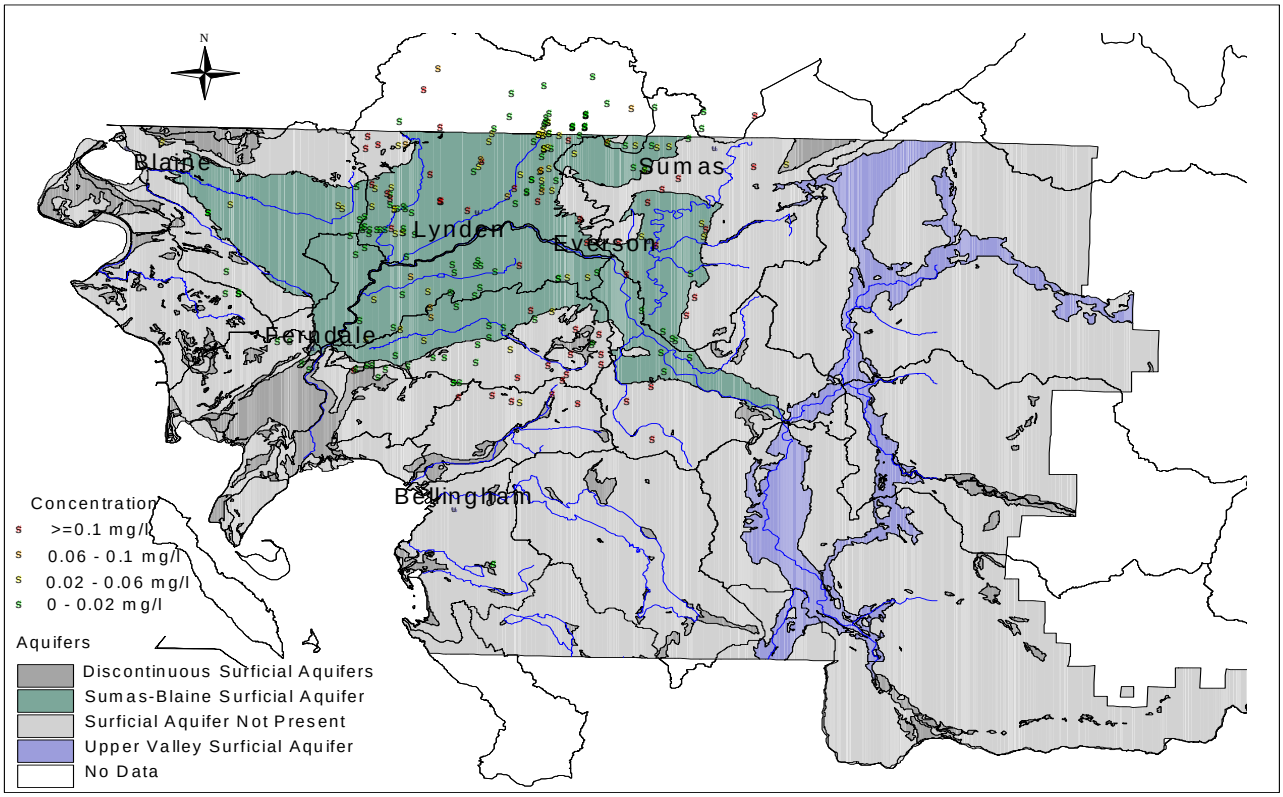

Figure E-8. Areal distribution of mean ammonia concentrations for 1990-2000.

APPENDIX F - AREAL DISTRIBUTION OF ORGANIC NITROGEN CONCENTRATIONS IN WRIA 1

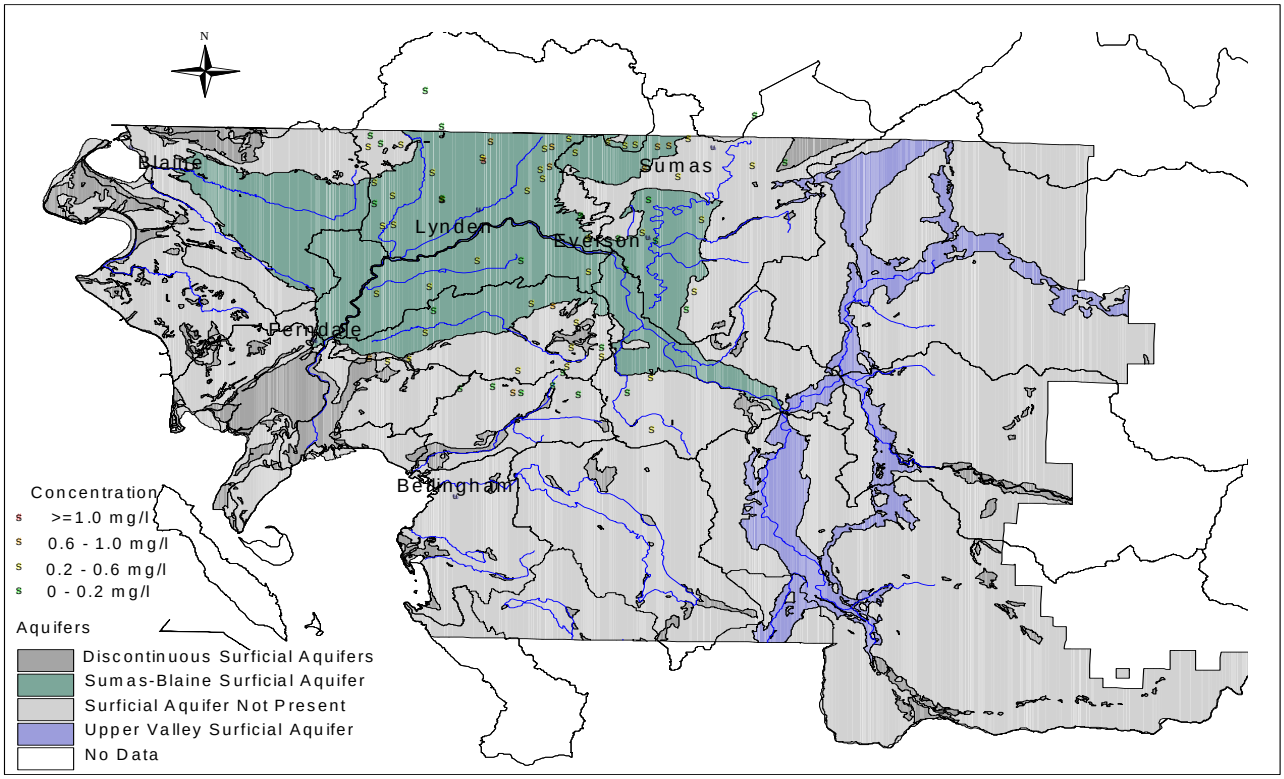

Figure F-1. Areal distribution of median total organic nitrogen concentrations in 1991.

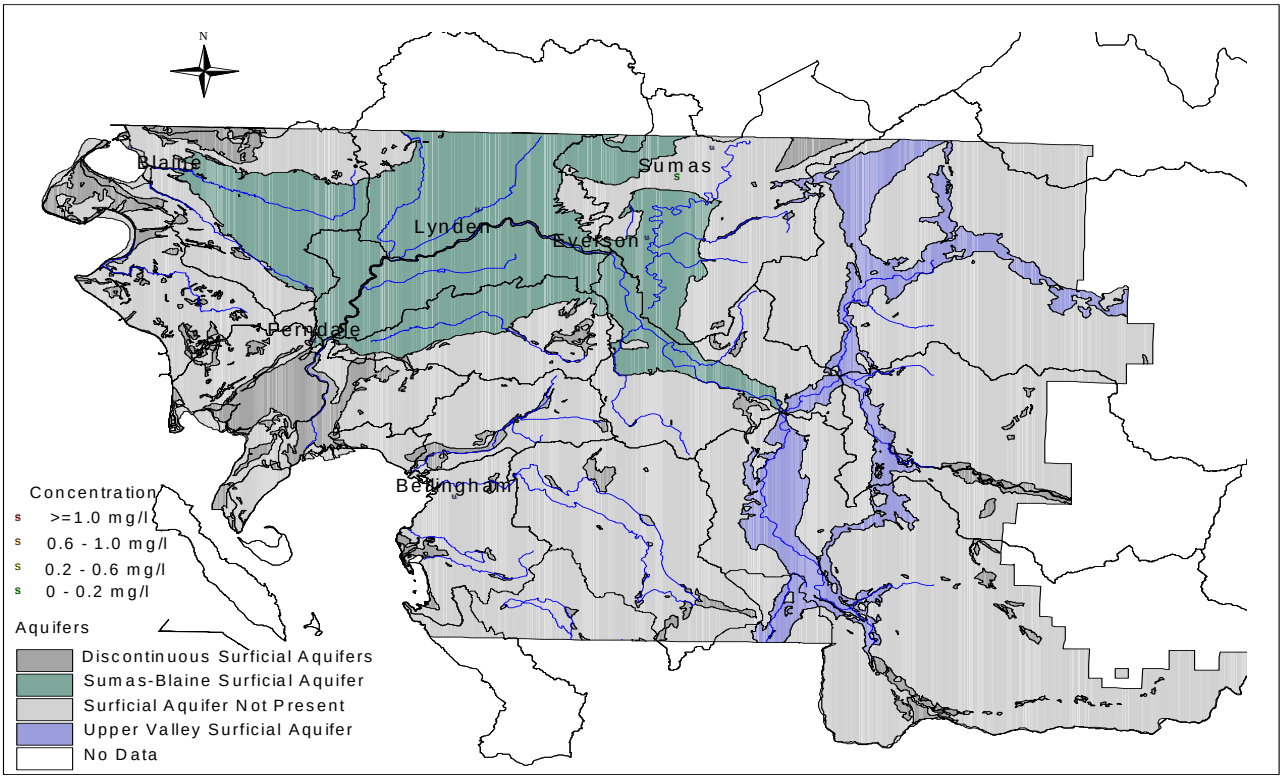

Figure F-2. Areal distribution of median total organic nitrogen concentrations in 1992.

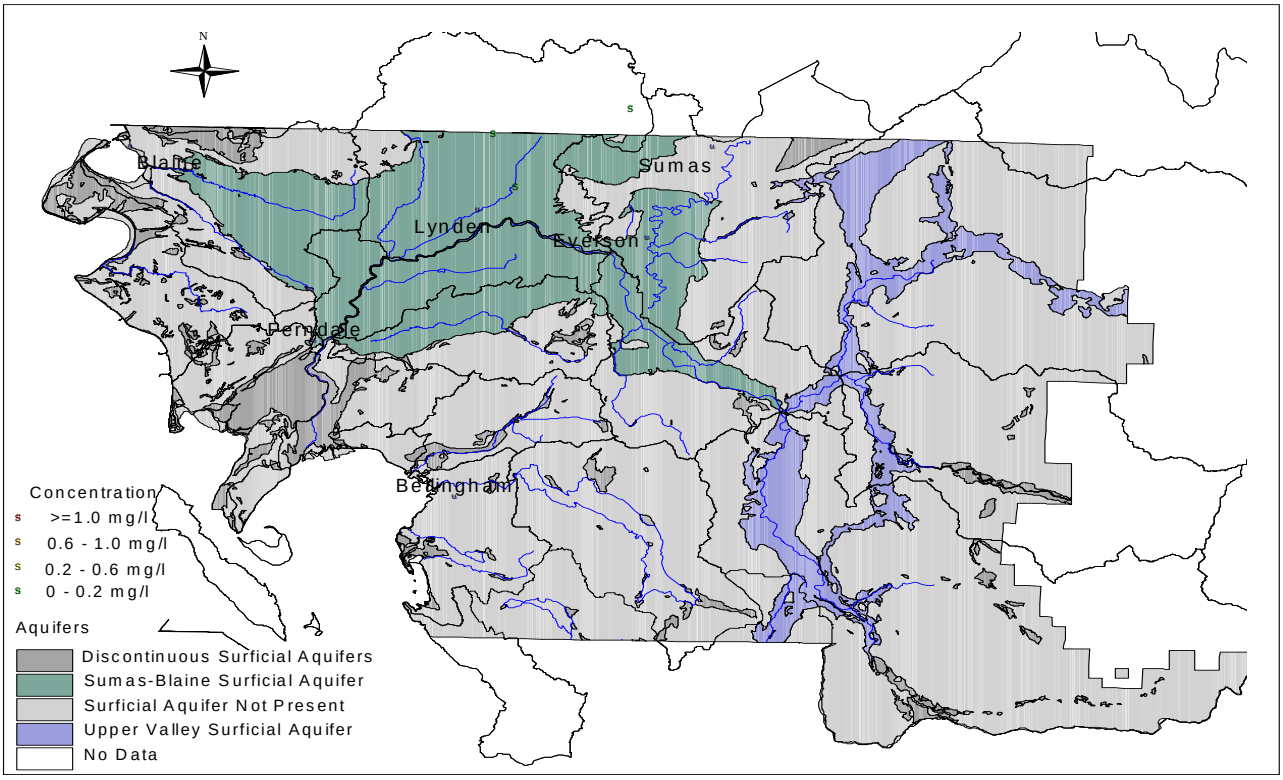

Figure F-3. Areal distribution of median total organic nitrogen concentrations in 1997.

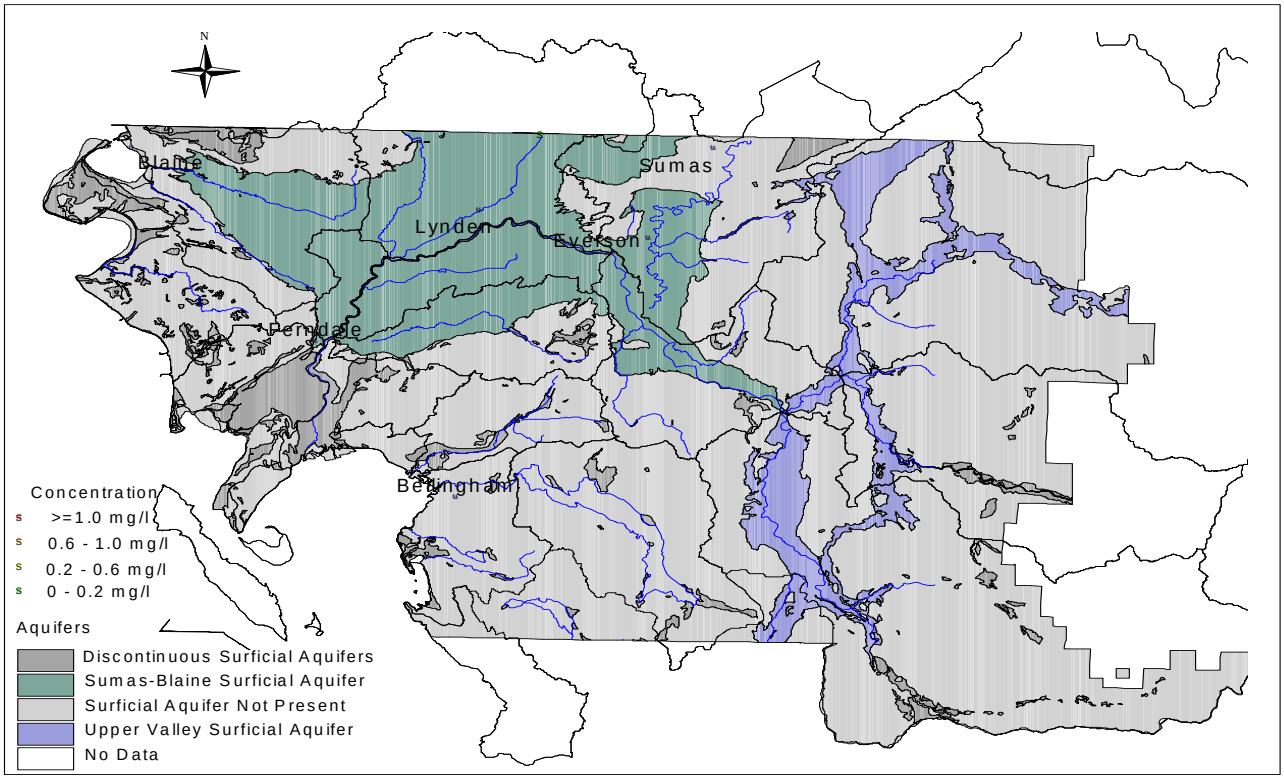

Figure F-4. Areal distribution of median total organic nitrogen concentrations – 1998.

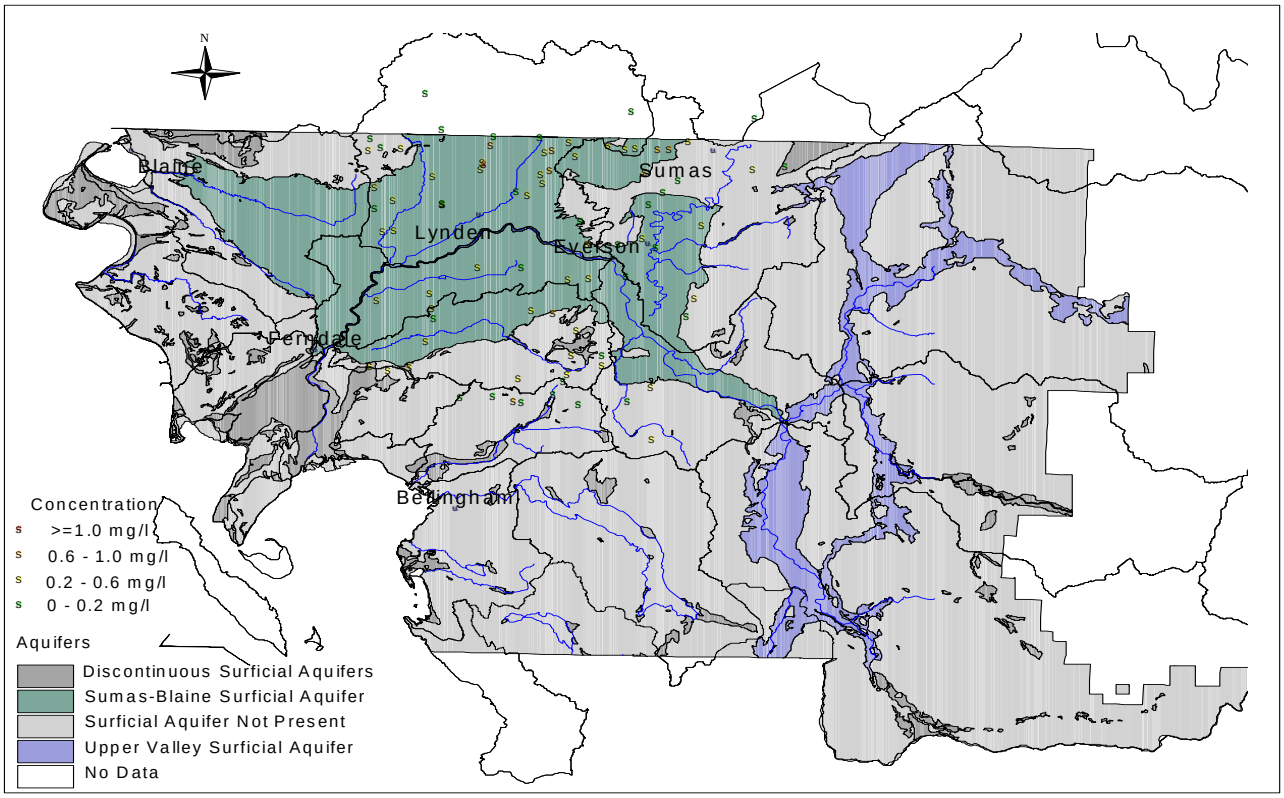

Figure F-5. Areal distribution of median total organic nitrogen concentrations for 1990-2000.

APPENDIX G - TIME SERIES PLOTS [NO3+NO2] FOR SELECTED WELLS WITH MAXIMUM DATA SERIES

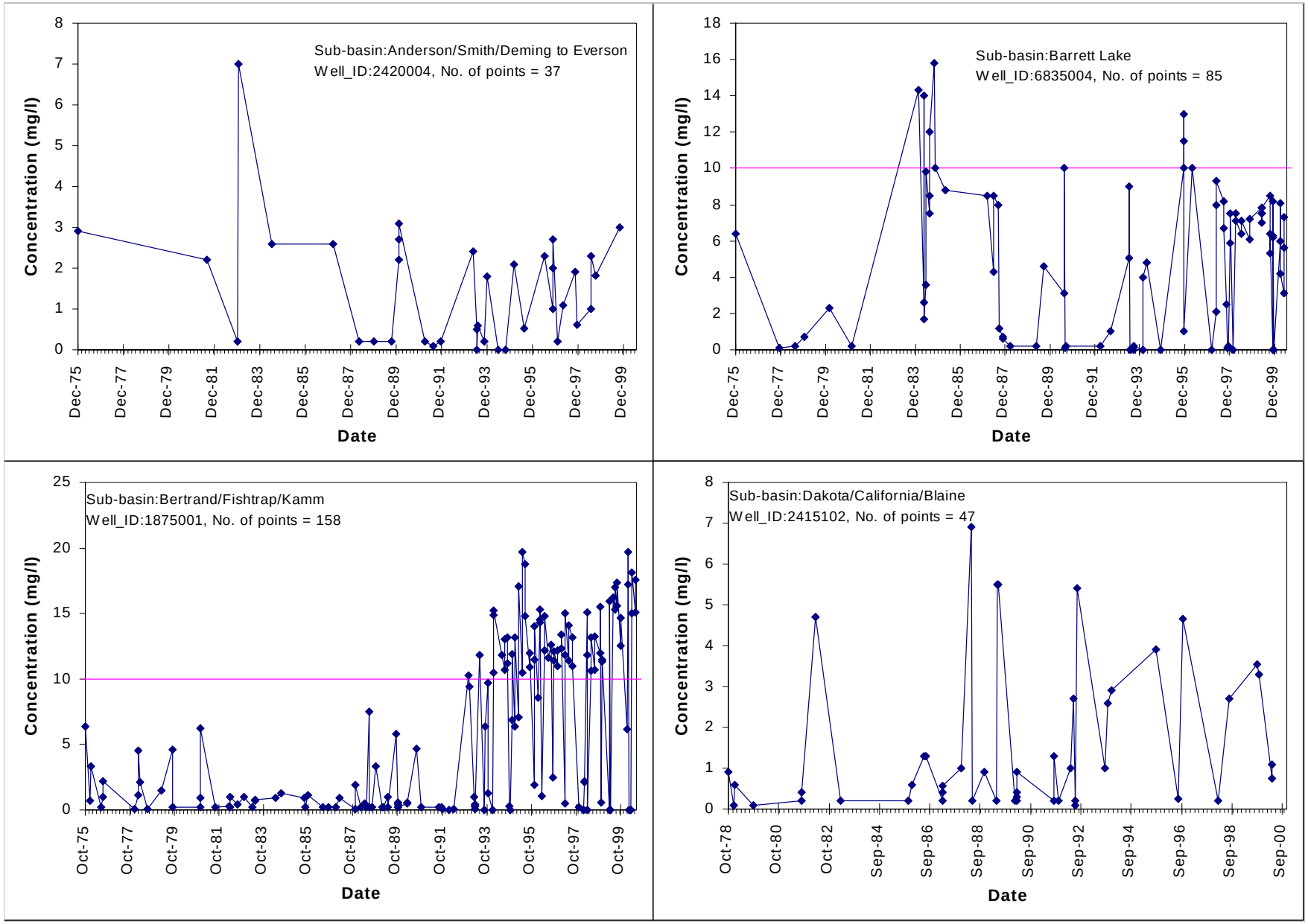

Figure G- 1. Time series of nitrates for wells with maximum number of data in the indicated sub-basins.

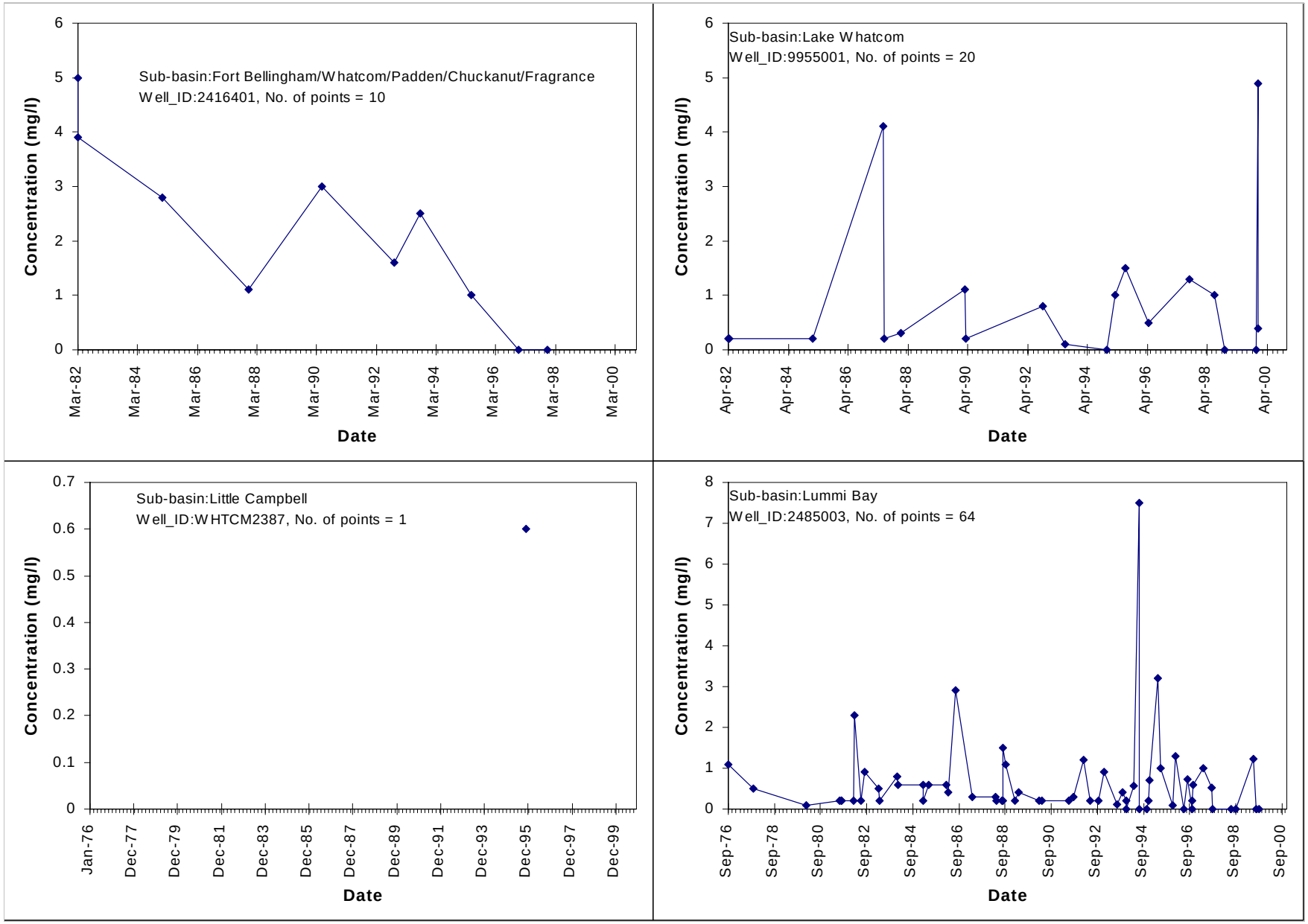

Figure G- 2. Time series plots of nitrates for wells with maximum number of data in the indicated sub-basins.

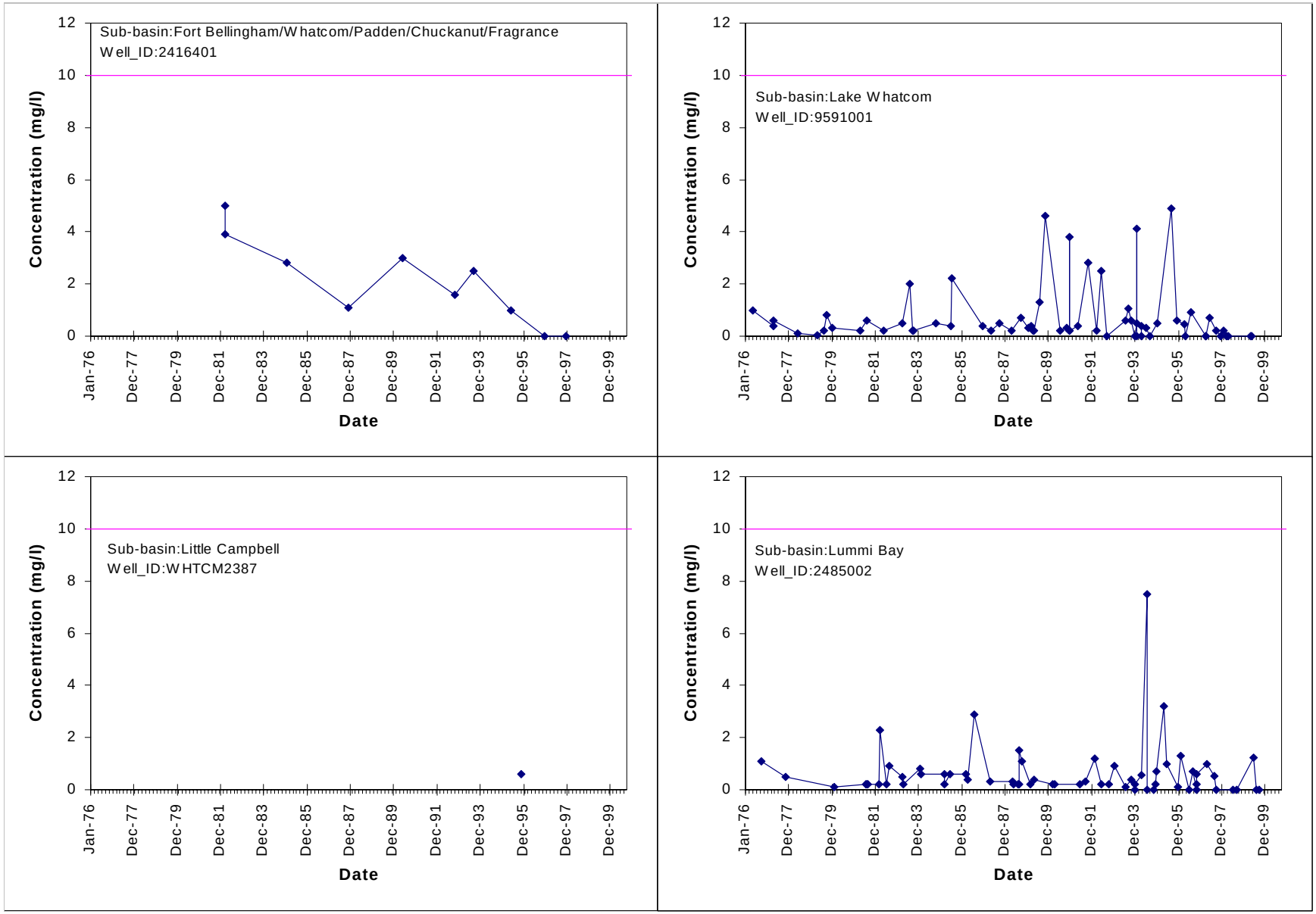

Figure G- 3. Time series of nitrates for wells with maximum number of data in the indicated sub-basins.

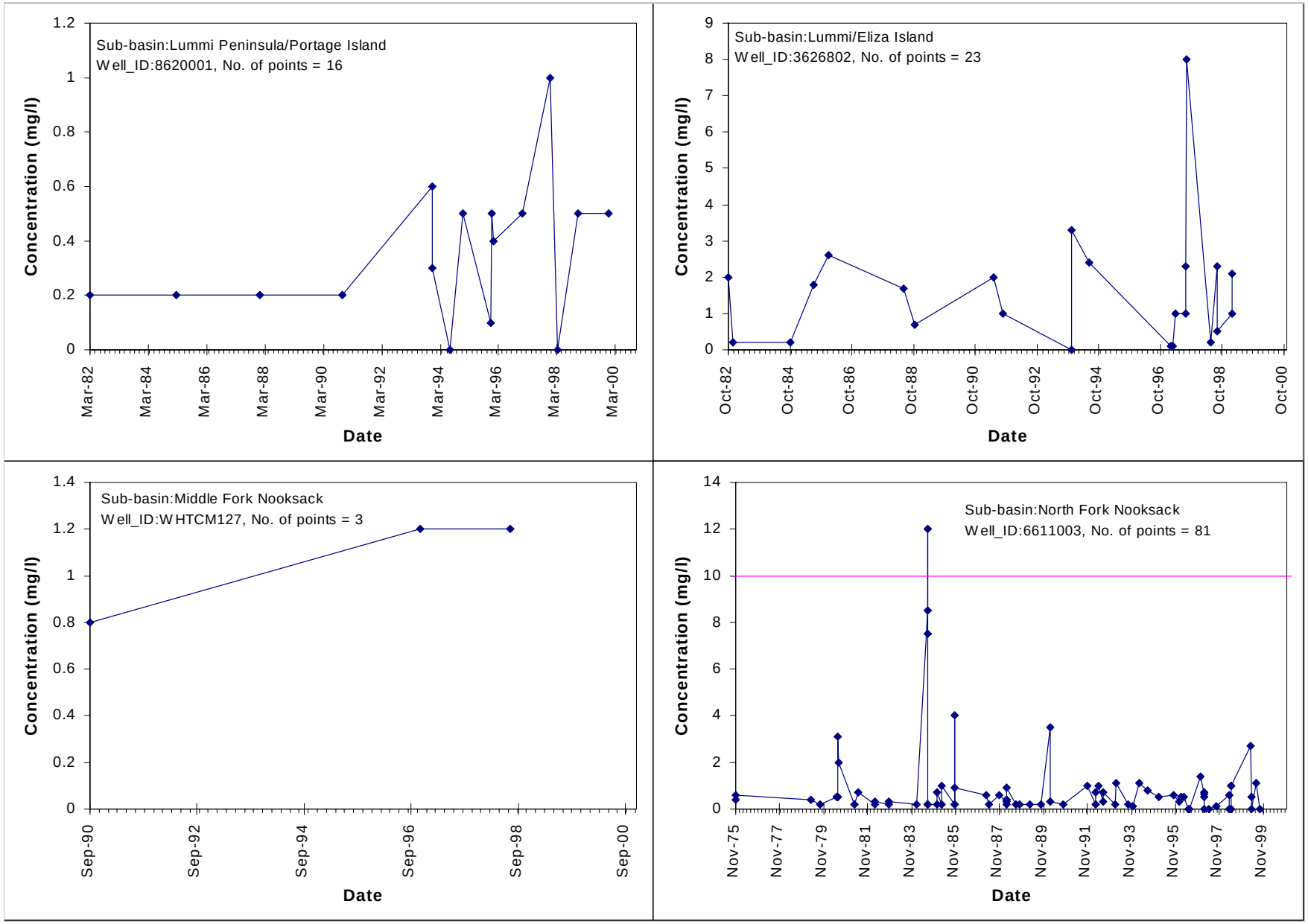

Figure G- 4. Time series of nitrates for wells with maximum number of data in the indicated sub-basins.

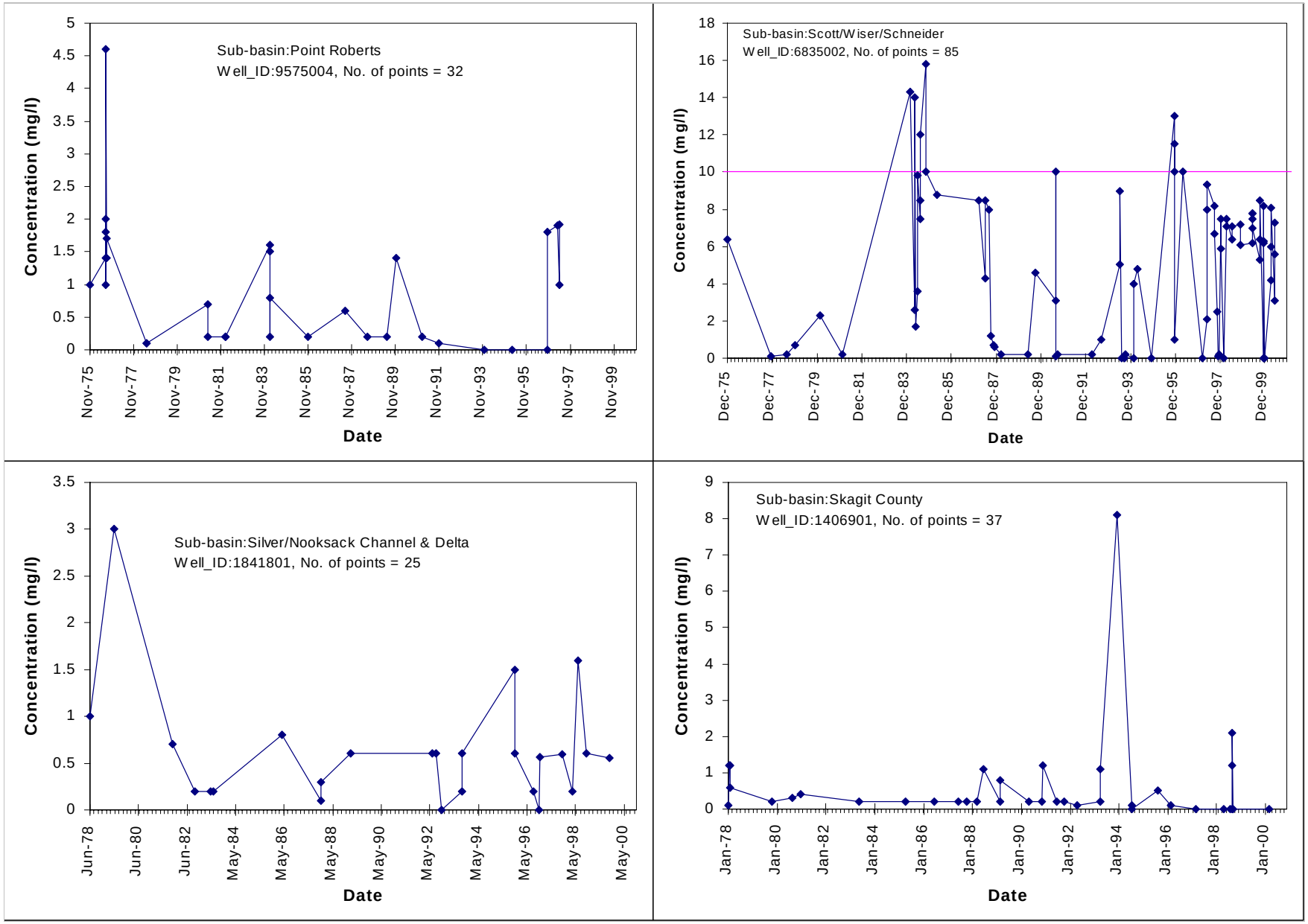

Figure G- 5. Time series of nitrates for wells with maximum number of data in the indicated sub-basins.

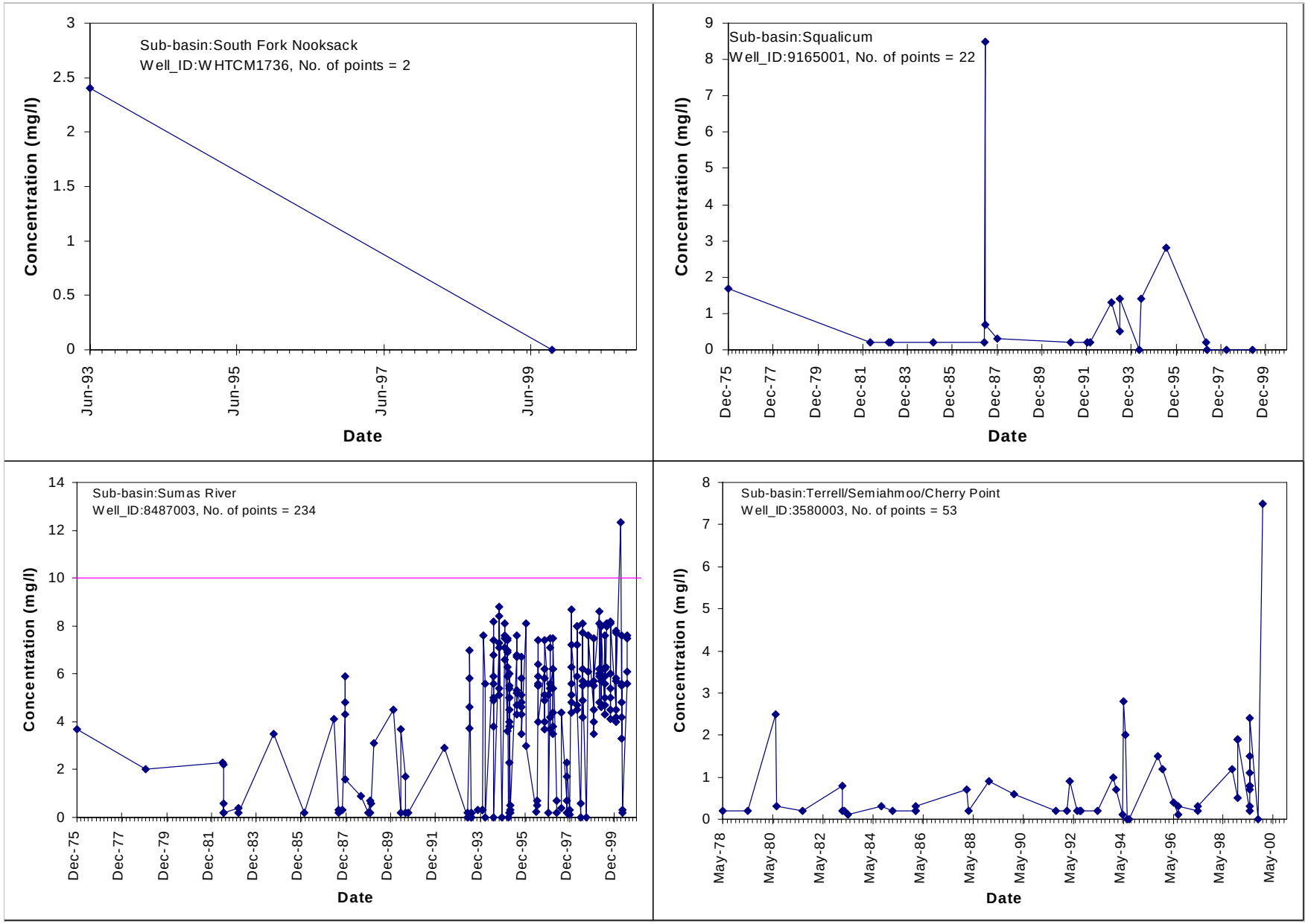

Figure G- 6. Time series of nitrates for wells with maximum number of data in the indicated sub-basins.

APPENDIX H - TIME SERIES PLOTS OF ORGANIC NITROGEN, AMMONIA, AND DISSOLVED OXYGEN

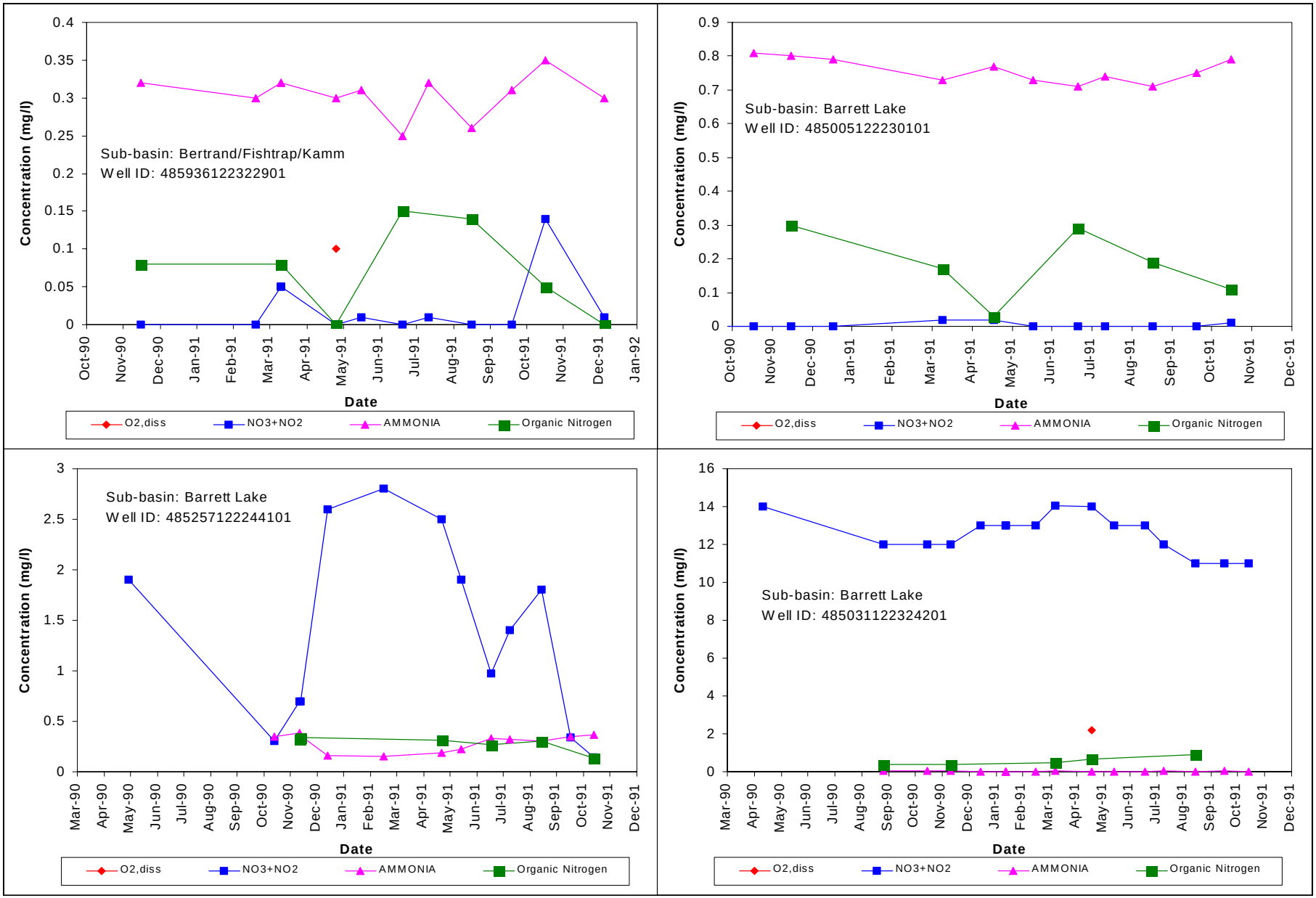

Figure H- 1. Time series of nitrates, ammonia, organic nitrogen and dissolved oxygen for wells with data. .

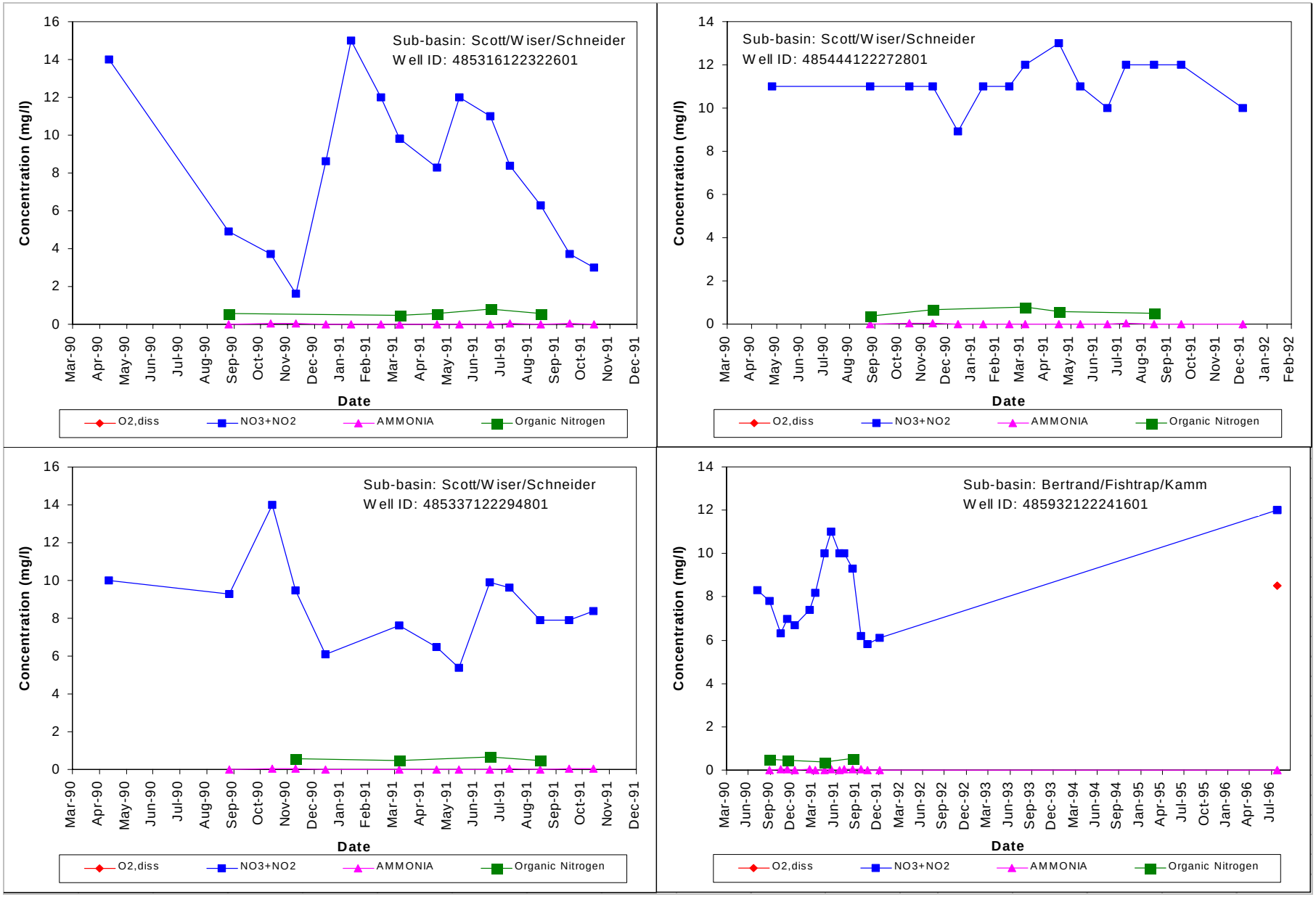

Figure H- 2. Time series of nitrates, ammonia, organic nitrogen and dissolved oxygen for wells with data.

APPENDIX I - SUMMARY DATA OF PESTICIDES EXCEEDING MCL FROM DIFFERENT WELLS IN WRIA 1

Table I- 1. Ground water wells exceeding MCL of EDB in WRIA 1; MCL=0.05 ppb.

Year	No. of Wells	# Wells Exceeding MCL	Well ID	Conc. (ppb)	Location
1984 `	7	4	WHTCM2593	1.25	Meadowdale area
			WHTCM2545	0.266	Bertrand Creek area
			WHTCM86	2.31	Bertrand Creek area
			WHTCM2599	0.065	Bertrand Creek area
1986	36	10	WHTCM73	2.9	Bertrand Creek area
			WHTCM295	5.9	Bertrand Creek area
			WHTCM295	6.1	Bertrand Creek area
			WHTCM47	0.84	Bertrand Creek area
			WHTCM88	0.94	Bertrand Creek area
			WHTCM90	0.4	Bertrand Creek area
			WHTCM46	3.8	Bertrand Creek area
			WHTCM83	0.05	Bertrand Creek area
			WHTCM101	0.056	Bertrand Creek area
			WHTCM2538	0.056	Bertrand Creek area
			WHTCM89	0.67	Bertrand Creek area
			WHTCM295	6.08	Bertrand Creek area
1987	25	5	WHTCM53	0.064	Meadowdale area
			WHTCM73	1.35	Bertrand Creek area
			WHTCM295	3.56	Bertrand Creek area
			WHTCM2594	0.086	Meadowdale area
			WHTCM59	0.057	Bertrand Creek area
			WHTCM73	1.18	Bertrand Creek area
1988	53	7	WHTCM73	1.69	Bertrand Creek area
			WHTCM90	0.3	Bertrand Creek area
			WHTCM295	3.87	Bertrand Creek area
			WHTCM73	1.99	Bertrand Creek area
			WHTCM73	1.45	Bertrand Creek area
			WHTCM73	1.83	Bertrand Creek area
			WHTCM86	0.104	Bertrand Creek area
			WHTCM86	0.101	Bertrand Creek area
			WHTCM2593	0.345	Meadowdale area
			WHTCM2596	0.54	Close to Sumas city
			WHTCM73	2.16	Bertrand Creek area
			WHTCM73	1.92	Bertrand Creek area
			WHTCM73	1.92	Bertrand Creek area
			WHWS24	2.95	Bertrand Creek area
			WHTCM73	1.92	Bertrand Creek area
			WHTCM73	1.59	Bertrand Creek area

			WHTCM73	1.72	Bertrand Creek area
			WHTCM73	1.64	Bertrand Creek area
1989	20	4	WHTCM73	2.29	Bertrand Creek area
			WHTCM73	2.02	Bertrand Creek area
			WHTCM46	5.79	Bertrand Creek area
			WHTCM73	2.13	Bertrand Creek area
			WHTCM295	3.3	Bertrand Creek area
			WHTCM73	1.6	Bertrand Creek area
			WHWS24	1.5	Bertrand Creek area
			WHWS24	1.52	Bertrand Creek area
			WHWS24	1.72	Bertrand Creek area
			WHTCM46	4.65	Bertrand Creek area
			WHTCM73	1.46	Bertrand Creek area
			WHTCM295	2.83	Bertrand Creek area
			WHTCM73	1.69	Bertrand Creek area
			WHTCM73	1.62	Bertrand Creek area
			WHTCM73	1.51	Bertrand Creek area
			WHTCM73	1.48	Bertrand Creek area
			WHTCM73	1.5	Bertrand Creek area
			WHTCM73	1.58	Bertrand Creek area
			WHTCM73	1.18	Bertrand Creek area
			WHTCM73	1.48	Bertrand Creek area
1990	5	2	WHTCM73	1.26	Bertrand Creek area
			WHTCM73	1.33	Bertrand Creek area
			WHTCM73	1.23	Bertrand Creek area
			WHTCM2576	0.17	Meadowdale area
			WHTCM73	0.94	Bertrand Creek area
			WHTCM73	1.38	Bertrand Creek area
1991	10	6	WHTCM46	2.4	Bertrand Creek area
			WHTCM47	0.6	Bertrand Creek area
			WHTCM47	0.62	Bertrand Creek area
			WHTCM73	0.83	Bertrand Creek area
			WHTCM87	0.3	Bertrand Creek area
			WHTCM90	0.24	Bertrand Creek area
			WHTCM295	2.1	Bertrand Creek area
1992	10	6	WHTCM46	2.32	Bertrand Creek area
			WHTCM47	0.65	Bertrand Creek area
			WHTCM73	1.22	Bertrand Creek area
			WHTCM87	0.22	Bertrand Creek area
			WHTCM90	0.14	Bertrand Creek area
			WHTCM295	1.95	Bertrand Creek area
1993	10	4	WHTCM46	2.86	Bertrand Creek area
			WHTCM53	0.28	Meadowdale area

			WHTCM87	0.31	Bertrand Creek area
			WHTCM295	1.55	Bertrand Creek area
1994	15	12	WHTCM46	3.13	Bertrand Creek area
			WHTCM87	0.32	Bertrand Creek area
			WHTCM295	2.39	Bertrand Creek area
			8487001	0.06	East of Sumas City
			8487002	0.06	East of Sumas City
			8487003	0.06	East of Sumas City
			8487004	0.06	East of Sumas City
			8487005	0.06	East of Sumas City
			8487006	0.06	East of Sumas City
			8487007	0.06	East of Sumas City
			8487008	0.06	East of Sumas City
			WHTCM320	0.05	Meadowdale area
			WHTCM320	0.05	Meadowdale area
			8487001	0.5	East of Sumas City
			8487002	0.5	East of Sumas City
			8487003	0.5	East of Sumas City
			8487004	0.5	East of Sumas City
			8487005	0.5	East of Sumas City
			8487006	0.5	East of Sumas City
			8487007	0.5	East of Sumas City
			8487008	0.5	East of Sumas City
1998	10	11	WHTCM2558	0.68	Bertrand Creek area
			WHTCM2558	0.68	Bertrand Creek area
			WHTCM46	0.11	Bertrand Creek area
			WHTCM47	0.24	Bertrand Creek area
			WHTCM73	0.13	Bertrand Creek area
			WHTCM86	0.12	Bertrand Creek area
			WHTCM87	0.085	Bertrand Creek area
			WHTCM88	0.24	Bertrand Creek area
			WHTCM409	0.052	Bertrand Creek area
			WHTCM410	0.07	Bertrand Creek area
			WHTCM73	0.406	Bertrand Creek area
			WHTCM46	0.54	Bertrand Creek area
			WHTCM47	0.37	Bertrand Creek area
			WHTCM295	0.37	Bertrand Creek area
			WHTCM2540	0.37	Bertrand Creek area
1999	3	1	WHTCM2642	0.186	Bertrand Creek area