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Manure and Groundwater Quality

Literature Review

by

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Abstract

Agriculture is important to the vitality of Washington State. There are concerns about impacts to groundwater quality from manure generated at animal production facilities (concentrated animal feeding operations; CAFOs).

The purpose of this report is to (1) review current scientific information on manure management and present strategies for measuring the effectiveness of these management practices and (2) identify practices and treatment technologies that assure that groundwater quality will be maintained and protected.

Land application of manure is a waste management tool that uses land treatment to beneficially reuse the nutrients to grow a crop. If manure is not properly managed, it can adversely affect groundwater quality. Research indicates that this typically occurs when (1) manure is applied in amounts greater than crops can use, (2) manure is applied when crops are not growing, (3) manured fields are over-irrigated, or (4) manure is stored in lagoons not constructed to a recognized standard.

This report summarizes information presented in the literature in the following areas:

- Documented impacts to groundwater quality.
- Effectiveness of manure management practices including application rates, crop uptake, timing of application, soil mechanics, irrigation water management, and storage lagoon design.
- Sampling protocols for crops, manure, soil, groundwater, and mass balance calculations.
- Manure management practices that adversely impact groundwater.
- Tools to assess manure management practices, quantify available nitrogen, and determine application rates and times.
- Evaluating impacts to groundwater quality using site-specific soil nitrate threshold limits and groundwater monitoring.

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Executive Summary

This literature review compiles existing studies and describes what is known about impacts to groundwater quality resulting from manure management at concentrated animal feeding operations (CAFOs). The intent is provide the scientific basis to support and guide our continued conversations with the agricultural community to work together to better protect groundwater quality in Washington State as well as foster sustainable agricultural practices. Based on the outcomes of these conversations, changes to Washington's CAFO general water quality discharge permit will be proposed.

Manure management is a key component to assuring that groundwater is protected and that farmers are in compliance with groundwater quality standards. CAFOs land-apply manure as a fertilizer to grow crops. By beneficially re-using their manure, CAFOs treat and manage their wastes and grow crops. When manure is appropriately land-applied in the right amounts and at the right times, the nutrients will be taken up by plants and will not seep into groundwater. Constructed ponds, called lagoons, are a common way manure is stored, allowing producers to contain manure until it can be applied to crops during the growing season.

Washington State groundwater quality standards are in place to protect all of the benefits groundwater provides, including the most common use of drinking water. Yet, there are documented adverse impacts to groundwater quality from some Washington CAFO manure management practices. Research indicates that this typically occurs when too much manure is applied, manure is applied during the wrong time, or manure is stored in a lagoon that is not constructed to a recognized standard.

The purpose of this report is to review current scientific information on manure management and present strategies for measuring the effectiveness of these management practices. The report summarizes the most current findings from more than 170 published research documents. These research documents are from local, state, national, and international sources including government agencies, scientific research groups, and academic institutions.

Primary Questions This Literature Review Attempts to Answer

- Is manure management impacting water quality?
- Which management practices are important to protect water quality?
- What kind of monitoring is most effective for evaluating management practices?
- Do storage lagoons leak? Are they causing problems? How should they be constructed?
- How do we know when a facility is operating in a way that protects water quality?
- What are the common conclusions and recommendations made by researchers?

Key Findings

- CAFOs and other animal operations can impact groundwater quality.
- There are documented impacts to groundwater quality in Washington State from CAFO manure management practices. Impacts to groundwater quality from other areas are also summarized.
- Manure management strategies that are critical to protecting groundwater quality are summarized. These management practices include:
 - Application rates – applying the right amount of nutrients to maintain a viable crop while minimizing impacts to water quality.
 - Timing – applying manure only during the growing season when the crop can use the nutrients. Three tools are cited to provide producers methods for determining the growing season based on their location and the crop grown.
 - Storing manure in a lagoon (constructed to recognized standards) during the non-growing season.
 - Managing irrigation water to prevent leaching of nitrate to groundwater.
- Monitoring is important to assess how well a facility is operating.
 - Spring and fall soil monitoring is recommended. Soil monitoring should be conducted in the spring to determine how much soil nitrate is present and how much additional nutrients are needed to maintain a viable crop. Fall soil monitoring should be conducted to assess the effectiveness of the previous year's manure management.
 - Literature offers recommendations on appropriate soil nitrate. Researchers recommend that spring soil nitrate should range from 16 to 30 ppm, and fall soil nitrate should range from 5 to 24 ppm. Researchers also provide recommended actions depending on different soil nitrate concentrations.
 - A tool was developed to assist in quantifying the potential nitrate leaching to groundwater based on the soil nitrate concentration and other site-specific factors.
 - Problems are noted with soil storage of nutrients during the non-growing season and year-round application of manure.
 - Groundwater monitoring is identified as the only way to measure impacts to groundwater quality.
- Typical nitrogen loss estimates are summarized for nitrogen volatilization, mineralization, nitrification, and denitrification.
- Success stories show benefits to CAFOs and groundwater quality.

Introduction

Agriculture has a strong historical presence in Washington State and provides many commodities and economic benefits. One segment of the agricultural industry is the production of animals and animal products. This industry segment is composed of animal feeding operations (AFOs), concentrated animal feeding operations (CAFOs), and other pasture and rangeland operations. The waste generated from animal operations is required to be managed to assure that water quality is protected. This literature review consolidates and summarizes information from relevant literature to provide a scientific basis for assessing manure management practices.

AFOs and CAFOs use collection systems to consolidate manure and other wastes from the facility production areas where animals are housed and products are produced. This collected waste is typically land-applied to provide nutrients for crops, or it is stored in a lagoon until crops can utilize the nutrients. Manure is a waste or by-product from animal production and is rich in nutrients that crops need, especially nitrogen. Manure can be beneficially used as a resource to grow crops. However, if manure is not managed properly it can adversely impact groundwater quality.

When nitrogen inputs to the soil system exceed crop needs, excess nitrate could enter groundwater. Adverse impacts to water quality can be minimized through management practices. Understanding the processes that affect nitrogen transformation and mobilization in the environment can help in efficiently managing nitrogen in land treatment systems.

Report Goals

The purpose of this report is to review current scientific information on manure management and present strategies for measuring the effectiveness of these management practices. The goal is to identify practices and treatment technologies that assure that groundwater quality will be maintained and protected. This report primarily focuses on the potential adverse impacts of animal wastes on groundwater quality and how these impacts can be monitored, assessed, and mitigated. This report summarizes the current scientific information presented in the literature in the following areas:

- Documented manure impacts to groundwater quality.
- Manure management related to groundwater impacts.
- Effectiveness of manure management practices including application rates, crop uptake, timing of application, soil mechanics, storage lagoon design and management, vegetative buffers and setbacks to surface waters.
- Use of mass balance calculations.
- Sampling protocols for crops, manure, soil, and groundwater.
- Tools to assess manure management effectiveness, quantify available nitrogen, and determine application rates and times.
- Strategies for evaluating impacts to groundwater quality using site-specific soil nitrate thresholds and groundwater monitoring.

Methods

The intent of this review is to include only peer-reviewed published work found in journals, technical reports generated by government agencies, scientific research groups, academic institutions, and other similar reputable scientific sources. Standard widely accepted practices are also included in some instances, when it is the basis for treatment practices or monitoring protocols.

Preliminary findings, raw data, work that is in progress, or work that has not been peer reviewed were not considered adequate to meet the goals of this report. Additionally, personal opinion and biased work were not included.

There is a tremendous amount of literature on agricultural practices. An effort was made to find the most current and relevant literature that addressed the report goals and met the above criteria. However, this review is not exhaustive.

Background

Nitrate is one of the most prevalent groundwater contaminants in the world (Nolan and Stoner, 2000; Rosenstock et al., 2014), and concentrations are increasing (Spalding and Exner, 1993). The National Academy of Engineering (2008) identified nitrogen management as one of the grand challenges facing the United States.

Numerous studies document that agricultural activities are a source of elevated nitrate in groundwater: Almasri, and Kaluarachchi, 2004; Hudak, 2002; Harter et al., 2002; Lindsey et al., 1998; Dzurella et al., 2012, Sánchez-Pérez et al., 2003; Yin et al., 2007; Erickson and Norton, 1990; Zebarth et al., 1998; Carey and Harrison, 2014; EPA, 2013a and 2013b; Burkholder et al., 2007; Sajil Kumar et al., 2013; Ju et al., 2006; Nolan and Stoner, 2000; Rosenstock et al., 2014; and Olson et al., 2009.

Nolan and Stoner (2000) concluded that in agricultural areas within the United States that 19% of the shallow groundwater wells do not meet the groundwater standard of 10 mg N/L. They also concluded that groundwater nitrate concentrations are higher in agricultural areas than in urban areas.

Previous Literature Reviews

Washington State University (Hermanson et al., 2000) conducted a literature search titled *Nitrogen Use by Crops and the Fate of Nitrogen in the Soil and Vadose Zone*. The primary goal of this review was to provide to the Washington State Department of Ecology (Ecology) information for management of process wastewater for land application, including sources from municipal, food processing, and livestock manure. This report focuses on nitrogen use by crops and the interactions between soil, water, and nitrogen.

Hermanson et al. (2000) provided the following general principles and recommendations drawn from their comprehensive review:

- The estimation of agronomic rate for a crop must factor in all sources of nitrogen available during the growing season. This includes mineralization, residual inorganic nitrogen, and contribution from irrigation water. Agronomic rate is defined as the recommended rate of nitrogen addition to the soil that is needed to produce an expected yield, while minimizing adverse environmental effects.
- In waste management scenarios, agronomic rate and the application rate may be different. When the application rate is in excess of the agronomic rate, close attention must be given to the environmental consequences of this practice.
- All nitrogen applied to the soil, that is not volatilized, will eventually convert to nitrate. The total transformation to nitrate may take a few weeks to a few years, depending on the nature of the organic waste.
- Nitrate moves readily with water in the soil profile and can reach groundwater if not taken up by the crops, denitrified, or volatilized. Other forms of nitrogen are less mobile.
- Soil nitrogen that moves below the root zone will eventually leach to groundwater as nitrate. Steps should be taken to minimize movement of nitrogen below the root zone during the growing and non-growing seasons.
- Denitrification may reduce nitrate loading to groundwater under some conditions, though it is of little importance in well-drained soils.
- Nitrogen applied at the time and in the amounts needed by the crop will minimize the buildup of soil nitrogen.
- Wastes applied substantially before or after maximum crop demand may result in the buildup of inorganic soil nitrogen that will subsequently be susceptible to nitrate leaching.
- Use of winter cover crops can minimize movement of nitrogen deeper into the soil profile by utilizing the nitrogen in the root zone, storing it in the plant tissue, and ultimately returning it to the soil surface after the cover crop dies. Cover crops temporarily store nitrogen removed from the root zone.
- Winter cover crops are not a reason to over-apply nitrogen. If excess nitrogen is applied in one growing season, it must be offset by decreased nitrogen application the following season to avoid residual nitrogen buildup and subsequent nitrogen leaching.
- Poor irrigation management will prevent efficient nitrogen management and recovery.
- The nitrogen composition of the manure should be determined before application because it will affect the timing of nitrogen availability and the susceptibility to nitrate leaching.
- Maximizing nitrogen removal by crops (by attempting to maximize crop production and increase nitrogen uptake) will generally increase the risk of nitrate accumulation in the soil.
- Organic wastes applied during the non-growing season will partially or totally convert to nitrate before the next growing season. The fraction mineralized will depend on the manure composition, the soil temperature and moisture conditions. The depth that nitrates will travel in the soil before the next growing season will depend on the soil hydraulic properties and the volume of recharge (precipitation and irrigation).

- Nitrate leached beyond the root depth of the crops to be grown during the following season will be susceptible for transport to groundwater.
- Steps should be taken to minimize movement of nitrogen below the root zone during the growing and non-growing season.
- Applying organic wastes during the non-growing season has an inherent risk in terms of leaching nitrogen to groundwater.
- The use of storage facilities to minimize waste applications during the non-growing season is a safe alternative.

This literature review builds on the work completed by Hermanson et al. (2000) and focuses on current environmental issues related to manure management with a specific focus on groundwater.

AFOs, CAFOs, and Dairies

Understanding the regulatory authority and history is helpful to the development of manure management practices within Washington State. Authority to regulate manure and provide technical assistance is divided between several agencies. Ecology has authority to issue general permits¹ as well as protect groundwater and surface water quality. Ecology has developed technical guidance documents for other similar types of land application discharges. A brief overview of the regulatory authority and related guidance is contained in Appendix B.

CAFOs generally have not been required to demonstrate compliance with groundwater quality standards. This is the case for both permitted and unpermitted facilities. Within Ecology's CAFO general permit; there is no requirement to monitor groundwater. Therefore, documenting a discharge and the associated impacts to groundwater (a water of the State) has been problematic. If a CAFO is not managing its manure properly (e.g., applying too much, applying at the wrong time, over-irrigating, or storing manure in a lagoon not constructed to a recognized standard), groundwater impacts are the likely result.

Within the framework of a water quality permit, disposal is not the goal; waste treatment and water quality protection are the goal. Manure management requires beneficially reusing nutrients in a manner that will not promote leaching. Maximizing crop yield is not equivalent to maximizing crop uptake since maximizing crop yield often entails applying excessive nutrients. Maximizing crop yield is not the goal of a land treatment system; the goal is to maintain a viable crop which utilizes the nutrients applied (Ecology, 2004a).

Dairies

In 1993 the Washington State legislature passed the Dairy Nutrient Management Act (Chapter 90.64 RCW). Dairies are one type of AFO. AFOs include CAFOs and the entire animal agricultural industry as a whole (e.g., beef, poultry, dairy, swine, horses), not just dairy farms.

¹ General permits are a type of permit that covers a category of discharger where there is a group of facilities that have similar discharge characteristics and utilize similar treatment mechanisms.

As a result of the Dairy Nutrient Management Program, the number of dairies and their locations are documented, while the same level of information has not been compiled for other types of AFOs. Because information on dairies is readily available, dairies have generally become the focal point of AFO-related discussions.

Dairies produce milk, cheese, butter, yogurt and other dairy products. Milk is Washington State's second most valuable agricultural commodity, second only to apples. Washington State ranked 10th nationally, producing 690 million gallons of milk in 2010. (WSDA, 2011)

In 2010, Washington had 443 commercial dairy farms with approximately 250,000 mature cows. WSDA (2011) notes that while the number of dairy farms has dropped over the years, the number of dairy cows has remained relatively constant.

Dairy farms are located in 28 of the 39 Washington counties. Table 1 describes the dairy size classifications (small, medium, and large) based on the U.S. Environmental Protection Agency (EPA) AFO definition. Figure 1 illustrates the distribution of classes within the state. Whatcom County is home to the most dairies with 125 farms, which collectively house a total 46,588 mature cows. Yakima County has the highest number of mature cows with 93,606 animals housed on 67 farms. (WSDA, 2011)

Table 1. Dairy size and distribution (WSDA, 2011).

Dairy Size	Number of Cows	Percent
Small	1 – 199	40%
Medium	200 – 699	37%
Large	700 +	23%
	>2,500	4% (16 of the large dairies)

Seventy percent of Washington's large dairies are located in Yakima County and the Columbia Basin. The majority of small and medium sized dairies are located in western Washington (WSDA, 2011). Figure 1 illustrates the locations of dairies.

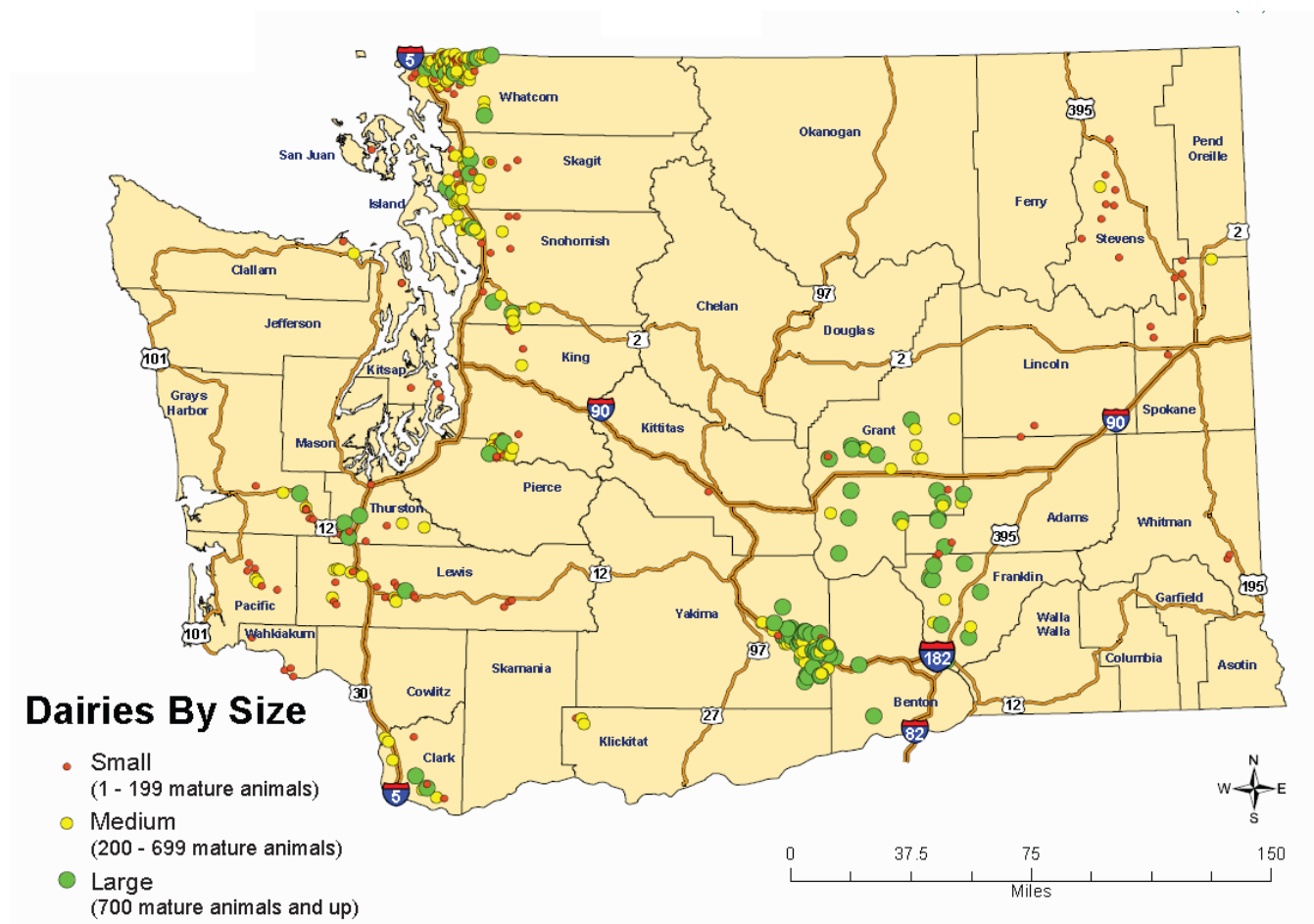
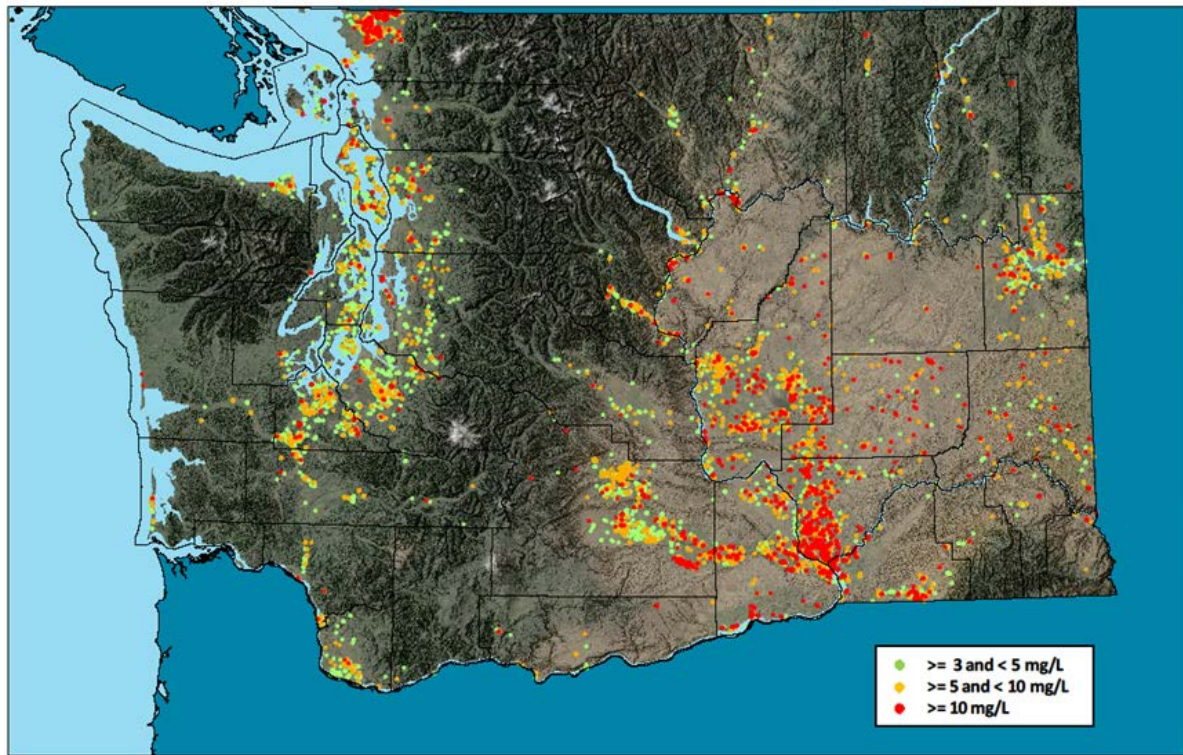


Figure 1. Dairy distribution in Washington State by size (WSDA, 2011).

Nitrate Impacts to Groundwater in Washington State

In Washington State, nitrate has contaminated public and private drinking water supplies. Groundwater is considered contaminated when it does not meet (exceeds) a drinking water standard. The drinking water standard set by EPA for nitrate is 10 mg N/L. Figure 2 highlights locations where elevated nitrate levels in groundwater have been documented. Nitrate groundwater contamination is a persistent widespread issue in the Lower Yakima Valley, the Lower Columbia Basin, and the Sumas-Blaine area in Whatcom County.

It is difficult to assess the direct impacts of AFOs on groundwater quality in Washington State, since few AFOs are required to monitor groundwater. In addition, there are only a few groundwater studies that assess impacts of AFOs in Washington State. EPA (2012 and 2013) recently conducted a groundwater investigation in the Lower Yakima Valley which linked CAFO discharges to groundwater contamination.



Above 3 mg/L indicates a human source
 Above 5 mg/L increases monitoring and work for Public Water Supplies
 Above 10 mg/L, exceeds the health limit (EPA MCL)

Figure 2. Groundwater nitrate occurrences in Washington State (Morgan, 2012).

Based on data from Washington State Department of Health Public Water Supply (2000-2011), Washington State Department of Ecology EIM (1982-2012), and U.S. Geological Survey (USGS) National Water Information System (1970-2011).

Nitrogen Loading by Sources

Nitrogen is a natural element that is concentrated through many sources and activities. It is present in human and animal wastes, plants, fertilizers, and precipitation. Nitrogen exists in different forms and behaves differently depending upon its form. Nitrate is the most prevalent form in groundwater and is the most mobile form. The primary forms of nitrogen in manure are organic nitrogen and ammonium. Plant-available nitrogen includes ammonium and nitrate. Appendix A describes the nitrogen cycle in more detail.

EPA (1999) summarized national data from animal operations. They calculated that the amount of manure produced by AFOs was 13 times greater than the sewage waste generated by humans in 1992 (on a dry weight basis). These AFOs produced 2.07 trillion pounds of manure in the U.S. and contained 11.6 billion pounds of nitrogen and 3.5 billion pounds of phosphorous.

Recent studies in the Lower Yakima Valley (EPA, 2013a) and the Sumas-Blaine aquifer in Whatcom County (Almasri and Kaluarachchi, 2004) assessed nitrogen contributions from various nitrogen sources. These two areas in Washington State have high groundwater nitrate concentrations. These studies assessed nitrogen sources in slightly different ways; however, the results described below, in percentages, are very similar.

The Lower Yakima Valley nitrogen sources are illustrated in Figure 3 and Table 2 (EPA, 2012). These percentages are based on the amount of nitrogen generated by the activity and the potential loading to the ground; they do not represent loadings to groundwater.

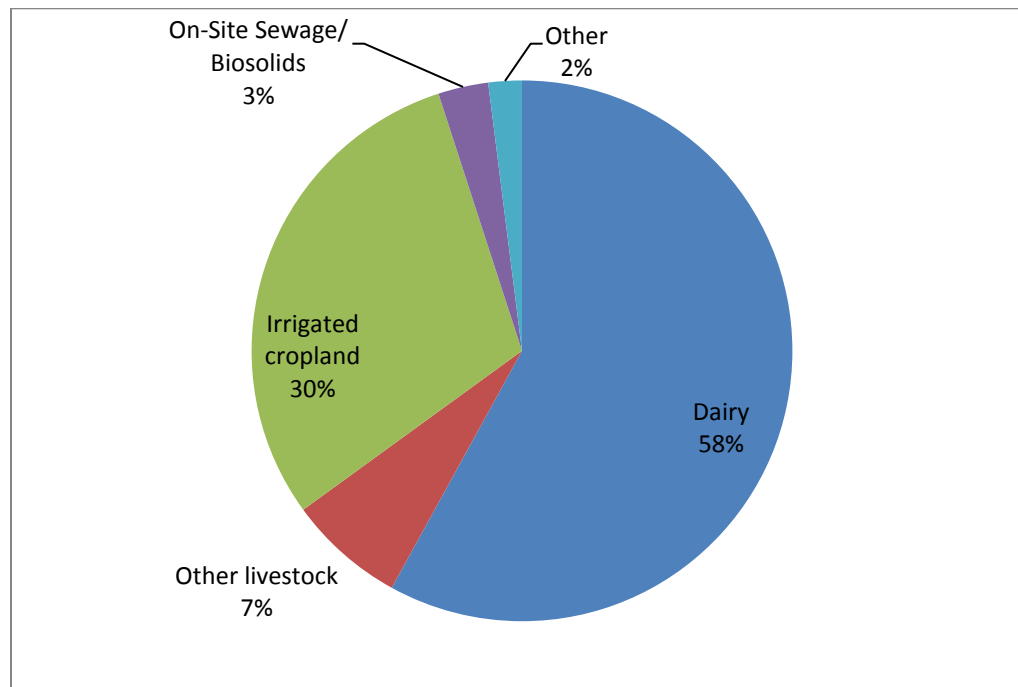


Figure 3. Estimates of nitrogen generated by sources in the Lower Yakima Valley (modified from EPA, 2013a).

Table 2. Estimates of nitrogen generated by sources in the Lower Yakima Valley.

Nitrogen Sources	%	lbs/year
Dairy	58	32,132,000
Other livestock	7	3,878,000
Irrigated cropland	30	16,620,000
On-site sewage/biosolids	3	1,662,000
Other	2	1,108,000

(Modified from EPA, 2013a.)

EPA (2012) estimated nitrogen generated based on area-specific assumptions for the identified sources in the Lower Yakima Valley. The on-site sewage estimates for the Lower Yakima Valley assume 6 lbs of nitrogen are generated per person per year, based on the total population in the area. Since the entire population is included in this calculation and many area residences do not use on-site sewage systems, this number overestimates the actual load from on-site sewage systems. This number more accurately reflects total nitrogen load from human wastewater, since it also includes loading from biosolids. The *other* category is composed of nitrogen loading from precipitation and from fertilizer used on lawns, parks, and golf courses.

The Sumas-Blaine aquifer (Whatcom County) nitrogen sources are illustrated in Figure 4 and Table 3 (Almasri and Kaluarachchi, 2004; Cox and Kahle, 1999). These percentages are based on the amount of nitrogen generated by the activity and the potential loading to the ground; they do not represent loadings to groundwater.

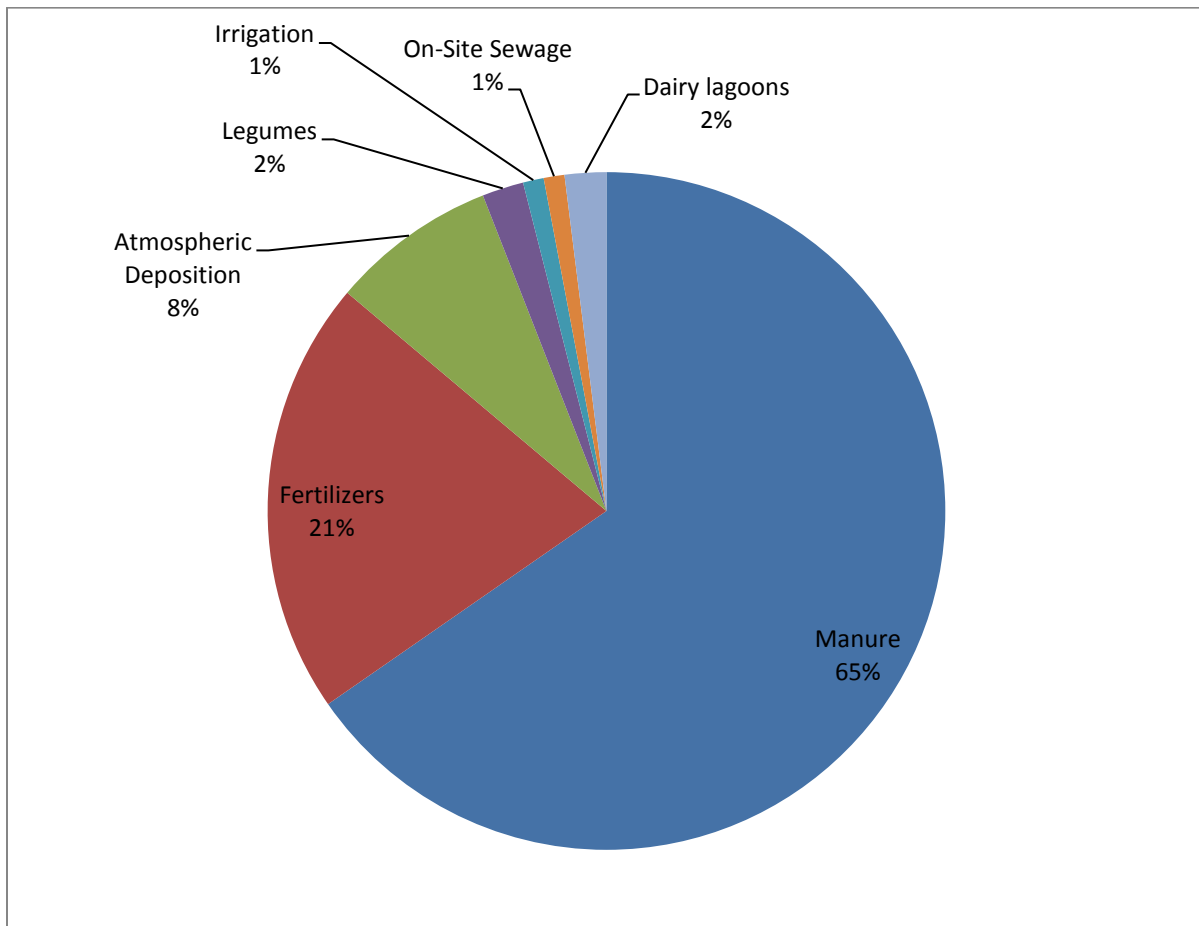


Figure 4. Nitrogen loading to the ground over the Sumas-Blaine aquifer (modified from Almasri and Kaluarachchi, 2004).

Table 3. Nitrogen loading to the ground over the Sumas-Blaine aquifer.

Nitrogen Sources	Percentage (%)	N Loading (lbs/acre/year)	Area (acre)	Total Nitrogen Load (lbs/year)
Manure	66	356	41,252	14,689,837
Fertilizers	21	73	63,196	4,625,947
Atmospheric deposition	8	7	236,196	1,724,231
Legumes	2	5	85,604	428,020
Irrigation	1	2	106,298	170,077
		(lbs/unit)	(units)	
On-site sewage	1	23	11,619	264,913
Dairy lagoons	2	1900	190	361,000

(Modified from Almasri and Kaluarachchi, 2004.)

Almasri and Kaluarachchi (2004) estimated nitrogen loading to the land, based on area assumptions for the Sumas-Blaine aquifer. Manure nitrogen contribution is calculated based on the number and the average nitrogen contribution for each type of cow category (Table 4). The contribution from dairy lagoons was calculated based on the following assumptions: (1) lagoons are filled from October through April, and (2) there is a leakage rate of 0.1 to 5 mm/day, (3) with an average ammonium concentration of 840 mg N/L, and (4) an average surface area of 30,000 square feet. (Almasri and Kaluarachchi, 2004; Cox and Kahle, 1999).

Atmospheric deposition includes dissolved nitrogen in precipitation (wet) and dry deposition. Dry deposition is the nitrogen that has volatilized into the atmosphere from manure and has been redeposited. Dry redeposition is calculated to contribute approximately 15 lbs/acre/year. (Almasri and Kaluarachchi, 2004; Cox and Kahle, 1999). These loading calculations consider the concentration of nitrate, the volume of precipitation, and the corresponding areas of land. The regional annual deposition rate for western Washington is estimated at 1 lb/acre/year, based on the assumption that precipitation contains an average of 0.26 mg N/L. (Cox and Kahle, 1999). This value is consistent with the average nitrate concentration of 0.24 mg N/L which Hem (1985), presents as the average of four studies where nitrogen was measured from rainfall.

Almasri and Kaluarachchi (2004) estimated the nitrogen loading from on-site sewage systems at 10 lbs per capita per year, and assumed 26,400 people were using on-site sewage systems. Legumes were assumed to contribute 5 lbs N/acre over the hay and pasture land class (clover is a common hay/pasture crop in the area). (Almasri and Kaluarachchi, 2004; Cox and Kahle, 1999).

Almasri and Kaluarachchi (2004) note that while the percentage contributions from on-site sewage systems (1%) and dairy lagoons (2%) may be small, these sources could cause localized impacts to groundwater.

Table 4. Nitrogen contribution based on cow type.

Dairy Cow Type	Number	Average N/head (lbs)
Milking	53,000	165 – 250
Dry	7,500	120 - 180
Heifer	12,800	60 – 90
Calves	7,600	74 - 112

(Almasri and Kaluarachchi, 2004)

Documented Manure Impacts on Groundwater

The long-term elevated nitrate groundwater concentrations in the Lower Yakima Valley (Ecology, 2010b) and the Sumas-Blaine aquifer (Redding, 2008) highlight a concern about nitrogen management and groundwater quality. Findings on impacts to groundwater quality from manure management practices are consistent within Washington State, in other areas of the United States, and the world.

Lower Yakima Valley

In the Lower Yakima Valley surface water is impacted by nutrient loading to the land surface, and groundwater is contaminated with elevated concentrations of nitrate. Numerous studies over the last 20 years indicate that elevated nitrate concentrations in groundwater are impacting public and private drinking water wells.

The EPA (2013a) recently conducted a groundwater investigation in the Lower Yakima Valley, which linked CAFO discharges to groundwater contamination. Over 20% of the private wells sampled during the study did not meet the nitrate drinking water standard of 10 mg N/L. The EPA determined that local CAFOs are a source of nitrate in the Lower Yakima Valley and have impacted groundwater quality. They estimated that dairies and other livestock operations contribute 65% of the nitrogen load to the land surface (Figure 3). Groundwater nitrate concentrations from monitoring wells directly downgradient of one CAFO were 190 mg N/L. The mean nitrate concentration in eight downgradient wells near the site was 45.1 mg N/L. This report also concluded that land application or fertilizer applications on irrigated crop fields are a likely contributing source of nitrate and that septic systems could also be a source of nitrate in groundwater, but that there was insufficient information to confirm this. (EPA, 2013a and 2013b)

Sumas-Blaine Aquifer

The Sumas-Blaine aquifer is the primary drinking water source for the majority of residents in rural northwestern Whatcom County. The aquifer has been impacted with elevated concentrations of nitrate. Numerous studies conducted over the last 30 years document that nitrate did not meet the drinking water standard of 10 mg N/L in 29% of sampled wells (Carey and Cummings, 2012). An increasing nitrate trend in groundwater was noted from March 2003 to March 2005 based on samples collected every other month from 35 wells. The mean nitrate

concentration for these wells was 11.5 mg N/L, with 26% of the wells consistently not meeting the standard for every sampling (Redding, 2008).

Elevated nitrate concentrations in groundwater are well documented in the Sumas-Blaine aquifer: Obert, 1973; Erickson and Norton, 1990; Erickson, 1991; Erickson, 1992; Garland and Erickson, 1994; Zebarth et al., 1998; Cox and Kahle, 1999; Hughes-Games and Zebarth, 1999; Carey, 2002; Mitchell et al., 2005; Chesnaux et al., 2007; Redding, 2008; Redding, 2011; Carey and Harrison, 2014.

The assessment conducted by Almasri and Kaluarachchi (2003) indicated that agriculture is the predominant source of nitrogen in the area (Figure 4), with manure contributing the largest portion of nitrogen to the land surface.

Studies Documenting Impacts to Groundwater Quality in Other Areas

The following sections summarize research studies which identify impacts to groundwater quality from manure management practices, including land application, lagoon storage, pens and corrals, and impacts to surface water.

Land Application

Spalding and Exner (1993) evaluated over 200,000 nitrate data points across the United States to determine the extent and spatial distribution of groundwater contamination. These researchers correlated elevated levels of groundwater nitrate with areas over well drained soils and irrigated cropland where there are excessive manure and fertilizer applications.

The USGS installed 30 monitor wells near irrigated agricultural fields in the Central High Plains States to evaluate the effects of irrigated agriculture on groundwater quality. These researchers sampled nutrients, pesticides, inorganic constituents, tritium, and nitrogen isotopes. The monitoring wells were designed to capture recently recharged water. They found 70% of sampled wells were affected by agricultural land use activities. Forty-three percent of the wells were impacted by commercial fertilizer, and 13% were determined to be impacted by an animal manure source. (Bruce et al., 2003)

Elrashidi et al. (2005) determined that loss of nutrients from agricultural lands through runoff to surface water and leaching to groundwater have contaminated waters in the United States.

Kirchmann et al. (2002) concluded that traditional farming practices and increased density of animals per unit of land base contributes to elevated nitrate to groundwater and surface water in Sweden.

In 1998 the USGS estimated that over 9% of the private domestic wells in Oregon's Willamette Valley were contaminated with nitrate from agricultural sources.

Sánchez-Pérez et al. (2003) noted that nitrate pollution of groundwater in northern Spain occurs in agricultural areas where there are excessive nitrogen applications.

Manure is a significant source of the nitrate in groundwater in the Lower Susquehanna River Basin in Pennsylvania and Maryland (Lindsey et al., 1998).

Fox et al. (2001) summarized water quality studies in Southeastern Pennsylvania, Chesapeake Bay, and the Gulf of Mexico, which document nitrate contamination of groundwater and surface water from agricultural areas.

Feaga and Selker (2004) correlated elevated soil nitrate concentrations with poultry manure applied at greater than agronomic rates.

Burkholder et al. (2007) conclude from a review of data in the literature that waste management practices from livestock operations are not effective to adequately protect water quality from nutrients, pathogens, and pharmaceuticals. These researchers state that this occurs when manure is over-applied to the crops but also note that impacts to groundwater quality can also occur when application is at recommended rates. These researchers note that in addition to application rates, the timing of application, residual nitrogen, crop, and climatic conditions also affect nitrate leaching to groundwater.

Sajil Kumar et al. (2013) documented impacts to groundwater quality in India as a result from manure and fertilizer applications.

Ju et al. (2006) studied manure application impacts to groundwater quality in North China. They documented nitrate leaching from all three cropping systems studied and exceedances of the drinking water standard in shallow groundwater where nitrogen application was in excess of the crop requirements. They also noted the presence of high residual soil nitrate.

Nolan and Stoner (2000) studied shallow groundwater quality across the United States from 20 study areas and 2,130 wells from 1992 to 1995 (including the central Columbia Plateau in Washington State). These researchers statistically evaluated data from different land uses. They found the highest median nitrate concentration of 3.4 mg N/L beneath agricultural areas, which include animal manure, inorganic fertilizer and atmospheric deposition. Additionally they detected nitrate in 71% of the samples, with 15% of the wells not meeting the drinking water standard.

Olson et al. (2009) found elevated nitrate concentrations in shallow groundwater in areas with irrigated manure, with maximum concentrations of 100 mg N/L in Alberta, Canada.

Rosenstock et al. (2014) concluded that agriculture has significantly impacted groundwater quality in California's Salinas Valley and Tulare Lake Basin. They estimate that the nitrate concentration in recharge water has doubled over the last 30 years, due to increased manure and fertilizer applications.

Dzurella et al. (2012) acknowledge that while on-site sewage systems are locally a significant source of nitrate in groundwater, in the agriculturally dominated region of California's Central Valley, animal manure practices are the predominant source of nitrate contamination in groundwater.

The Council of Canadian Academies (2013) expert panel noted that in Canada where residual soil nitrate is elevated, shallow groundwater nitrate concentrations are also elevated. These experts stated that groundwater systems tend to respond more slowly to changes in land management between nitrogen application and impacts to groundwater. Timing is important to evaluate success with groundwater quality. The legacy effects from over-application of nitrogen may take years to fully impact groundwater. Therefore, these experts state that it is important to make positive changes in areas where groundwater has been impacted.

The European Union introduced a *Nitrate Directives* in 1991. This directive identifies nitrate from manure and fertilizer applications as the main cause of pollution from diffuse sources affecting European communities. This cooperative directive concluded that high numbers of concentrated livestock results in manure production that is out of balance with available land and crop requirements, which in turn creates a surplus of nutrients ultimately lost to water and air. The European Union has imposed a number of management directives that are improving both groundwater and surface water quality. For example, nitrate-vulnerable zones were designated where nitrogen applications were restricted in the amount and the times of application. The results from this directive have been difficult to quantify; however, the European Union (2010) found that from 2004 to 2007 there were stable or decreasing nitrate trends in 70% of surface water samples and stable or decreasing trends in 66% of the groundwater samples. Sutton et al. (2011) released “The European Nitrogen Assessment” as a scientific approach to address the nitrogen issues raised by the European Directive.

Lagoons

Erickson (1994) investigated the effects of dairy lagoons on groundwater quality in Washington State. This researcher found leakage at three of the four lagoons investigated, and two had sufficient leakage to impact local groundwater quality. He also noted that the proximity of the water table to the lagoon liner was an important factor that affected the ammonium load to the subsurface. This study also found elevated levels of chloride, total dissolved solids (TDS), bacteria, total organic carbon, chemical oxygen demanding substances, and ammonium in groundwater near leaking lagoons. The predominance of ammonium over nitrate was attributed to the close proximity of the lagoons to the water table and the saturated soils beneath the lagoon, which created anaerobic conditions. The maximum concentration of ammonium was 180 mg N/L, with a typical range of 30 to 60 mg N/L. Some level of nitrification occurred with nitrate concentrations in groundwater noted at over 90 mg N/L at one site, over 80 mg N/L at another, and concentrations that exceeded the 10 mg N/L groundwater standard at the third site.

Garland and Erickson (1994) conducted a groundwater quality evaluation in the area in and around a dairy lagoon in Whatcom County, Washington. Sampling began before the lagoon was used and continued for three years after. These researchers concluded that leakage from the lagoon adversely impacted groundwater quality (causing exceedances of the drinking water standard) in the immediate vicinity of the lagoon.

Ham (2002) noted that discharge from manure lagoons occurs and these losses can affect groundwater quality.

McNab et al. (2006) sampled monitor wells at a manure lagoon in the San Joaquin Valley, California to evaluate nitrogen loading and nitrogen transformations in groundwater. These researchers measured dissolved gases in manure lagoon contents and in groundwater along with gases released to the atmosphere. This profile identified the unique geochemical composition that was used as a tracer to track manure leakage. They concluded that at this site, leakage from the lagoon definitively migrated to groundwater.

The Minnesota Pollution Control Agency (MPCA) compiled a summary report (2001) for four studies, evaluating the effects of liquid manure storage systems on groundwater quality. These researchers concluded that manure storage systems impact groundwater quality. Specifically, they determined that unlined manure storage units have a greater impact on groundwater quality than open feedlots or earthen (cohesive, soil-lined) or concrete-lined storage systems. They found evidence of impacts to shallow groundwater downgradient of manure storage areas at each site studied although these impacts varied widely. These researchers documented the plume length and concentrations of nutrients and other inorganic parameters.

Rudolf (2015) evaluated agricultural best management practices (BMPs) in Woodstock, Ontario, Canada, where public drinking water supply wells had been contaminated with elevated nitrate concentrations. This community decided to purchase an additional 275 acres of land with the goal to keep the agricultural land in production but to change BMPs to reduce water quality to below (meet) drinking water standards. The application rate was reduced from 100 lbs/acre to 54 lbs/acre. The result was a 60% reduction in soil nitrate concentrations, from 20 ppm to 8 ppm. Agricultural production increased slightly from 135 bu/acre to 140 bu/acre of corn grown at this site. This researcher concluded that incentives to producers are influential for improvements in water quality. (Rudolf, 2015; Rudolf et al., 2015)

Stephen et al. (1999) evaluated groundwater monitoring data from dairies in New Mexico. These researchers found the mean nitrate concentrations in wells located directly downgradient of clay-lined lagoons were significantly higher than wells located near synthetic-lined lagoons.

Miller et al. (1976) monitored soil beneath manure lagoons at four hog farms. Elevated concentrations of ammonium were present below all lagoons. Elevated ammonium concentrations were found at depths up to 20 – 30 cm (8 – 12 in) in fine-grained soils, but in medium- and coarse-grained soils, elevated ammonium concentrations were found down to the water table. These researchers concluded that earthen manure lagoons are not appropriate over medium- and coarse-grained soils.

Baram et al. (2012) investigated the fate of ammonium and nitrate below earthen waste lagoons at a dairy in Israel. Lysimeters were used to collect leachate over four years. These researchers concluded that there are two distinct infiltration processes that occur. The first is the slow constant infiltration from the base of the lagoon. And the second is the fast infiltration of wastewater and precipitation through desiccation cracks that form on the inside of the lagoon banks. This is an area with high clay content soils, which facilitated nitrification coupled with denitrification within the top 0.5 m. This coupled nitrification-denitrification phenomenon (CND) resulted in 90 to 100% reduction in total nitrogen in the vadose zone. Average wastewater ammonium concentrations were 2,012 mg/L (+/-482 mg/L) with average groundwater nitrate concentrations under the lagoon of 71 mg N/L (+/- 19). These nitrate

concentrations were 3.5 times higher than the mean concentration in regional groundwater. These researchers concluded that the presence of this nitrate indicates leachate from the waste lagoon have reached groundwater.

Baram et al. (2014) conducted a subsequent investigation in the Beer-Tuvia region, Israel which is a historical dairy farming area (1960 to 2010). These researchers developed a methodology to assess the past and future impacts of lagoons on regional groundwater quality using spatial analysis of chloride and total nitrogen. They found that while irrigated agriculture is the predominant contributor of nitrate and chlorides to the aquifer, that lagoon leakage is also a contributor. They determined that lagoons contributed 6% of the total mass of chloride and 12.6 % of the total mass of nitrate even though lagoons comprised less than 1% of the land use in the area. These researchers also noted that allowing the lagoons to go through a drying period resulted in a reduction of 11% of chloride and a 25% reduction in the nitrate contribution to the subsurface.

DeSutter et al. (2005) investigated the movement of manure leachate below four animal waste lagoons. Ammonium in the top 0.5 m (1.6 ft) ranged between 94 to 1139 mg/kg (ppm) with elevated concentrations as deep as 4 m (13 ft) below the base of the lagoon.

Ham and DeSutter (2000) determined that leachate from manure lagoons can affect groundwater quality, and they recommend synthetic liners in areas with vulnerable groundwater.

Koike et al. (2007) detected tetracycline in groundwater from manure seepage from unlined lagoons.

Ham (2002) conducted a study in Kansas investigating 20 anaerobic manure lagoons using a water balance method. These lagoons included 1 dairy, 5 cattle, and 14 swine facilities. Ham concluded that the swine manure had 3 to 5 times the nitrogen content as cattle or dairy manure. For dairy and cattle, the mean ammonium load discharged through the lagoon is 385 kg/ha/yr. This researcher also collected soil profiles beneath old lagoons. He found the highest concentrations of ammonium, organic nitrogen, and phosphorous and other cations, directly below the lagoon liner to a maximum depth of 3 m (10 ft), where concentrations declined to background conditions. Anions, such as chloride and nitrate, were not retained in the soils and migrated deeper. Ham determined that the ammonium retention was dependent upon the cation exchange capacity of the soils, the chemical composition of the manure (particularly other cations), and depth to groundwater.

Reddi et al. (2005) advocate using chloride as a conservative tracer of lagoon leakage rather than ammonium or nitrate. These researchers explained that most of the nitrogen is in the organic or ammonium form. In these forms, it is readily attenuated in the subsurface and can be dependent on the cation exchange capacity of the soils. Chloride is highly mobile, moves with water, and is not attenuated due to its negative charge. Chloride concentrations are elevated, typically greater than 1,000 mg/L.

The MPCA (2001) monitored the bottoms, perimeters, tiles, and sidewalls from different lagoon types. Chloride concentrations were used to illustrate the predominant location of leakage from

these lagoons. The highest concentrations were detected in the sidewalls, and bottom of the lagoon, with perimeter and center tiles showing elevated concentrations in some lagoons.

Nicholson et al. (2002) conducted a risk assessment of a variety of storage methods looking at water pollution, odor, ammonia emissions, and pathogen inactivation. These researchers determined that earthen liners pose the greatest risk to groundwater and that above-ground, lined structures with a cover were best, since they provided a high level of protection to groundwater and prevented atmospheric emissions and odors.

Pens and Corrals

Animal holding areas include pens and corrals. They typically have unlined compacted soil floors, but some have concrete floors. There is little research available on impacts or seepage from these areas. (Harter et al., 2014)

Contributions from CAFO pens and corrals have not been extensively investigated. Miller et al. (2008) investigated the seepage difference between coarse- and fine-textured soils. These researchers found that there was no difference between the different soil types. They determined that leaching below the earthen floor of the feedlots was restricted by three distinct layers: manure, a black interface layer, and underlying soil. These layers are created from a unique process which occurs from compaction by cattle, physical clogging of pores by manure, and the dispersion of clay from sodium and potassium. The result is a restrictive zone that does not promote infiltration or leaching. Because this self-sealing process occurs readily with coarse-textured soils, the researchers found that soil texture is not a factor. These researchers also noted that this restrictive black interface layer should be maintained and not disturbed when cleaning pens. (Miller et al., 2008)

Miller et al. (2008) investigated three feedlots using chloride as a conservative tracer of leakage. They found elevated chloride concentrations down 2.3 feet below land surface before they returned to background concentrations. These researchers also determined that the surface permeability was 4 to 93×10^{-7} cm/sec.

Vaillant et al. (2009) found elevated levels of ammonium, nitrate, and chloride at the land surface in the animal holding areas and noted that these concentrations decreased to background levels below 3 to 7 feet below land surface.

van der Schans et al. (2009) calculated that urine and manure from the animal holding areas contribute approximately 20 in/yr of liquid to the corral area. However, these researchers determined that this is mostly lost to evaporation.

Surface Water Impacts from Winter Manure Application

While the primary focus of this literature review is on groundwater, the documented continuity between surficial groundwater and surface water in Washington State lends itself to considering the potential risks to surface water (Ecology, 2010a; Winter et al., 1998).

The risk of fecal coliform bacteria runoff to surface waters increases when manure application occurs during high precipitation periods. Nunez-Delgado et al. (2002) noted that fecal coliform levels increase in soils after land application of manure. Fecal coliform bacteria survival is highly variable and is dependent upon a number of factors including salinity, temperature, moisture, sunlight, predation, organic content, and nutrient content.

In a 2003-2004 study, Nennich et al. (2005) observed that after dairy manure had been land-applied during the winter at a field in western Washington, fecal coliform counts in soils increased 100 to 10,000 times the original concentration approximately three weeks after application (Figure 5 early December application and Figure 6 late January application). Fecal coliform die-off to original levels took approximately 60 days during both application periods.

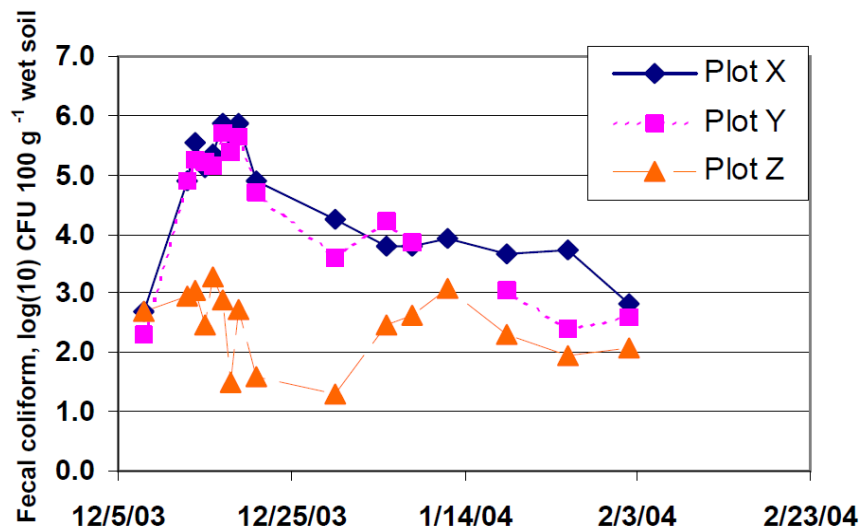


Figure 5. Fecal coliform bacteria survival in soil, 2003-2004 (Nennich et al., 2005).

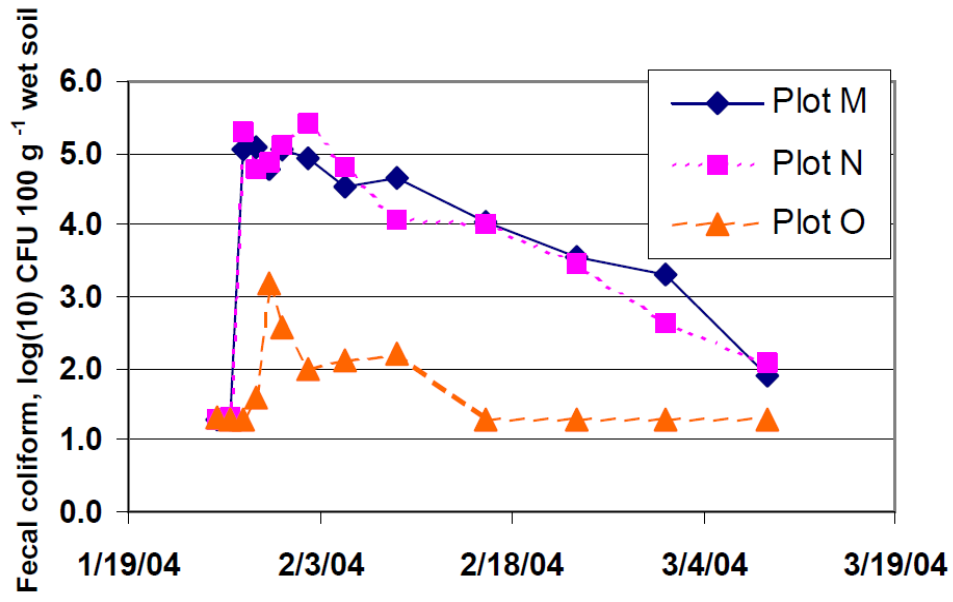


Figure 6. Fecal coliform bacteria survival in soil, 2004 (Nennich et al., 2005).
Plot O is located in the vegetative buffer.

For two monitored soil plots within the manure application area, Nennich et al. (2005) noted that it took 52 and 42 days for the fecal coliform levels in soils to return to background levels following a manure application. Other researchers have noted longer recovery times; Stoddard et al. (1998) observed 60 days for fecal coliform bacteria, and Avery et al. (2004) observed 162 days for *E. coli* to return to background levels.

Researchers noted that bacteria present in manure can survive in soils for long periods of time. (Hubbs, 2002; Nunez-Delgado et al., 2002). During winter, the increased frequency, intensity, and duration of precipitation events and increased volume of rainfall can combine with bacteria in soils, increasing susceptibility of impacts to ground and surface water. Both Nennich et al. (2005) and Nunez-Delgado et al. (2002) documented fecal coliform transport into vegetative buffer zones during precipitation events.

Nennich et al. (2005) studied the fate and transport of manure to surface water. These researchers found that fecal coliform bacteria can impact surface water through runoff from pasture land three or more days after manure application in the winter. They also noted that a 2.2-inch (5.6 cm) rainfall event three days after a light manure application (25% to 30% of normal applications) in January caused direct runoff of bacteria from the application area into the setback area (Figure 6). It took 17 days for bacteria levels in the setback area to return to background levels.

Panhorst (2002) noted that in soils where dairy manure was applied to cropland, complete fecal coliform destruction took 130 to 165 days from the date of application (Figure 7). This researcher also noted that in soils where dairy manure was applied to pastureland, complete destruction of fecal coliform bacteria took between 125 to 145 days from the date of application (Figure 8). Figures 7 and 8 illustrate the elevated bacteria levels that were present in manure, and the gradual, slow die-off that occurred in the environment.

Hubbs (2002) studied fecal coliform concentrations in runoff from manure applied to grass plots following multiple rainfall events. Fecal coliform concentrations remained high (1,000 – 10,000 CFU/100 mL) 30 days after the manure application (Figure 9).

Additionally, authors note that increased wintertime precipitation can result in a seasonally high water table. Under saturated soil conditions, the infiltration rates may be lower, resulting in higher potential runoff to surface water.

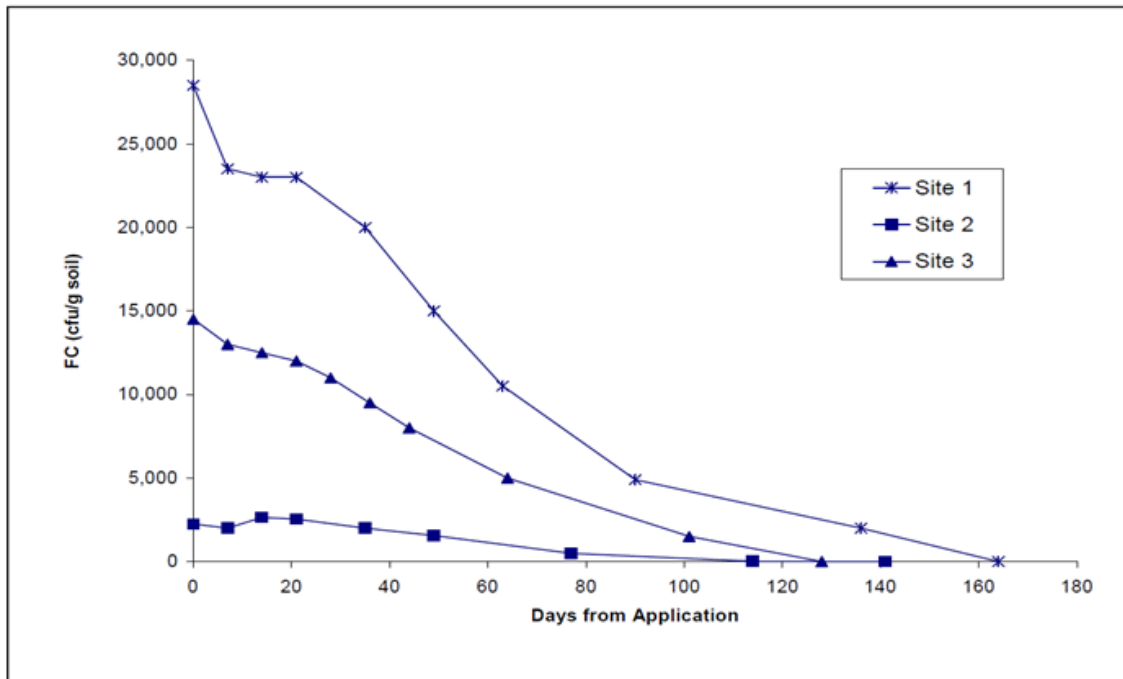


Figure 7. Fecal coliform concentrations in soil for dairy manure applied to cropland (Panhorst, 2002).

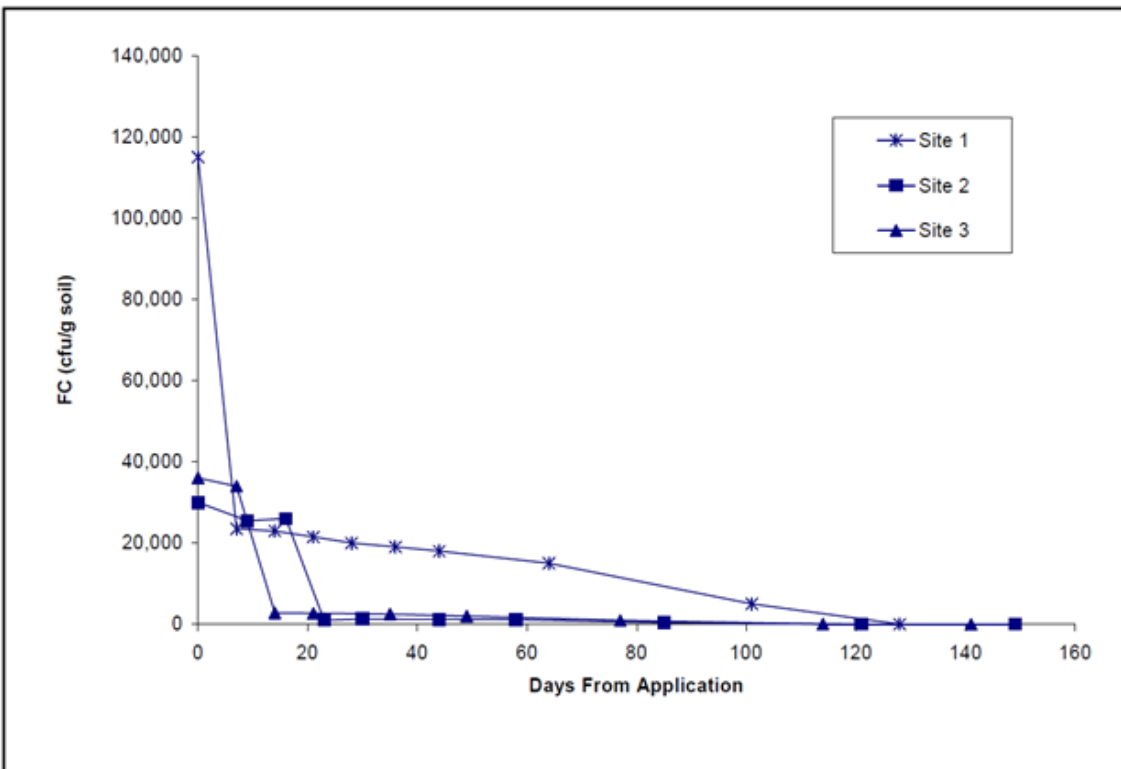


Figure 8. Fecal coliform concentrations in soil for dairy manure applied to pastureland (Panhorst, 2002).

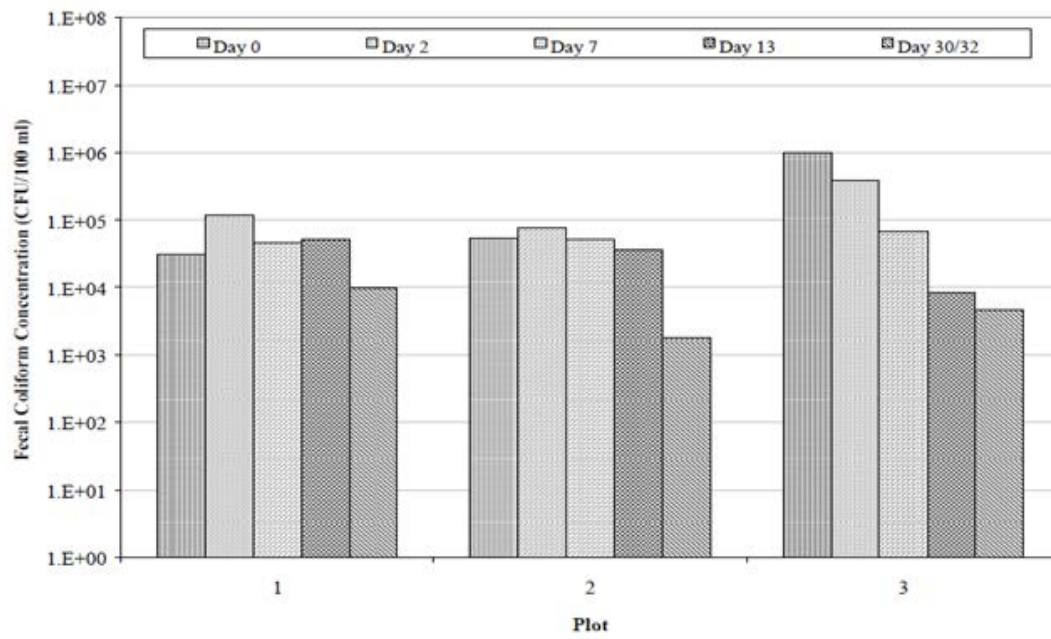


Figure 9. Die-off of fecal coliform bacteria from 0 to 32 days (Hubbs, 2002).

Manure Management and Treatment

Manure is generated during AFO operations as a waste by-product that must be properly managed to prevent water quality problems. Manure can be managed in several ways: collection, storage, treatment, utilization, and transfer (NRCS, 2009b). This section on manure management and treatment focuses on the effectiveness of commonly used manure management practices that utilize the nutrients (application rates, crop uptake, and timing), treatment of manure through attenuation (soil mechanics, vegetative buffers, and setbacks) and storage in constructed lagoons.

Manure is typically managed in three forms: solid, liquid, and slurry. In western Washington waste is generally managed as slurry and is land-applied to crops. In eastern Washington the solids are generally dry-stacked and then land-applied. Liquids are stored in lagoons and then later land-applied.

The most common manure treatment method is land application and crop uptake. Land application of manure is used to grow crops and enhance soil health (NRCS, 2005) and can be part of the treatment system for the waste produced by an AFO. If manure is managed properly, the nutrients contained within this by-product can be a beneficial resource when used as a fertilizer for crop production and can protect water quality (Hermanson et al., 2000).

Dzurella et al. (2012) recommend a suite of manure management practices that can help to protect and improve the quality of groundwater. These include: optimizing application rates, more precisely timing the application of irrigation water and manure to better match crop needs, evaluating crop rotation strategies, improving storage and handling of manure, and reducing fertilizer applications. These researchers state that effective manure storage and management are essential to protecting groundwater quality.

Application Rates

Summary

*A review of literature on manure management illuminated the inconsistent use of the term “**agronomic rate**.” Consistent definition and use of terms is important to implementing consistent manure management practices. This section describes various definitions of “agronomic rate,” discusses factors affecting manure application rates, then proposes the use of “**application rates**” to avoid confusion tied to historic definitions.*

Several factors identified in the literature affect application rates. These include accounting for all sources of nitrogen, the type of nutrients applied, when they are applied, the type of crop grown, the type of soils, and climate. Researchers agree that all sources of nitrogen be considered in the total load. Residual soil nitrate and continued mineralization of organic nitrogen are often overlooked sources.

Studies verify that there is a positive correlation between nitrogen application rates and crop uptake and that there is also a positive correlation between nitrogen application rates and residual soil nitrate. Soil nitrate accumulation is related to nitrate leaching. Researchers have observed that improvements to manure management practices can reduce the soil nitrate concentrations, as well as significantly improve groundwater quality.

Researchers note that at a higher than optimum nitrogen application point, the crop yield and economic return declines. Other researchers have observed that even under optimal conditions and appropriate application rates, nitrate leaching can occur.

Definitions

The term *agronomic rate* has many definitions depending upon the objective of the activity. Agricultural activities have different goals for irrigation and fertilizer use than a land treatment system where the goal is waste treatment. The primary goal of agricultural production is to maximize crop yields. The primary goal for a land treatment system, such as an AFO that generates manure, is waste management; it is not to maximize crop yields. (Ecology, 2004a)

The Implementation Guidance for the Groundwater Quality Standards defines *agronomic rate* for land treatment systems, as *the rate at which a viable crop can be maintained and there is minimal leaching of chemicals downwards below the root zone*. (Kimsey, 2005). This definition is appropriate for manure treatment.

The Natural Resources Conservation Service (NRCS, 2005) utilizes “*realistic yield goals*” as their criteria for nitrogen application. The NRCS (2009c) defines “*agronomic rate*” as the amount of crop nutrients required to achieve the expected yield after considering the contribution of the soil, plant, water, and atmospheric nutrient sources.

In a literature review on nitrogen use by crops (Hermanson et al., 2000) the term *agronomic rate* is defined as the recommended rate of nitrogen addition to the soil that is needed to produce an expected yield, while minimizing adverse environmental effects.

The EPA defines *agronomic rates* for two similar types of nutrient land application: (1) for biosolids, agronomic rate is defined as the rate at which plants require nitrogen during a defined growth period, and (2) for sewage sludge, agronomic rate is defined as the amount of nitrogen needed by a crop or vegetation grown on the land, while minimizing the amount of nitrogen that passes below the root zone of the crop or vegetation and enters groundwater (EPA, 1994).

Maximizing crop yield is not equivalent to maximizing crop uptake. Maximizing crop yield is not the goal of a land treatment system. (Ecology, 2004a)

Factors Affecting Application Rates

Several factors identified in the literature could affect application rates. These include accounting for all sources of nitrogen, the type of nutrients applied, the time they are applied, the type of crop grown, the type of soils, and climate. The following paragraphs summarize information that addresses cropping and leaching as affected by the factors listed above.

NRCS (2014) practice standard 590 identifies essential criteria to be considered when choosing application rates. The NRCS requires that a nutrient management plan be developed which includes a nutrient budget for nitrogen, phosphorus, and potassium from all potential sources, with the goal to manage the rate, source, placement, and timing of nutrients.

Fertilizer guides offered from the land grant universities provide extensive information on application rates based on climate and crop type. <http://www.extension.uidaho.edu/crops.aspx>

Hermanson et al. (2000) state that the estimation of agronomic rate for a crop must factor in all sources of nitrogen available to the crop during the growing season, including: mineralization, residual inorganic nitrogen in the root zone, and nitrogen in the irrigation water. These researchers also note that if the agronomic rate is properly calculated and utilized, this will help minimize the buildup of soil organic nitrogen.

Cogger (2000) concluded that in Washington State there is a history of manure applications that have resulted in increased rates of mineralized soil nitrogen. This buildup of soil nitrate reduces the need to supplement soils with manure or fertilizer.

Chesnaux et al. (2007) concluded that soil nitrate concentrations are higher in manured fields than fields using synthetic fertilizer when the same amount of nitrogen is applied. These researchers attribute this to the high organic nitrogen content in manure, and the total organic carbon contribution to the soils which acts to retain nitrogen for a longer period of time. Once this nitrogen mineralizes, it becomes available to the plant or is susceptible to leaching to groundwater. These researchers conclude that it is essential to consider the mineralized soil nitrogen as a source when calculating application rates.

Lindsey et al. (1998) investigated manure management in karst topography. They concluded that the application rate of manure is one of the most important factors in managing nitrate concentrations in streams.

Harter and Menke (2004) found that changes to nutrient management can result in significant improvements to water quality. These researchers specifically showed that there was a 70% improvement in groundwater quality beneath fields where manure is land-applied when the following changes were made: (1) eliminating the use of commercial fertilizer, (2) reducing manure applications, and (3) applying manure only during the crop growth stages. These researchers noted that these measures did not result in a considerable reduction in crop yield.

Carey and Harrison (2014) found that groundwater nitrate concentrations near the top of the water table were close to the groundwater standard of 10 mg N/L when manure application rates were similar to the crop removal rates and when manure was applied during the growing season.

Feaga and Selker (2004) found that once manure application rates were reduced, the soil nitrate concentrations also declined.

Carey (2002) found that estimated nitrogen application rates from manure on a grass field in western Washington that exceeded the typical grass crop uptake resulted in median groundwater nitrate concentrations at the top of the water table of 27 to 31 mg N/L with a maximum concentration of 74 mg N/L. This was in contrast to a grass field in the same area where estimated manure nitrogen application rates were similar to crop uptake rates. In this case the resulting groundwater nitrate concentrations were close to 10 mg N/L.

Accurate crop yield estimates are necessary for planning nitrogen application rates. Sawyer et al. (2006) noted the amount of nitrate leached to groundwater increases as the rate of applied nitrogen increases. These researchers demonstrated that increasing the applied nitrogen above the optimum crop uptake actually yielded a smaller crop and a smaller economic return. They also noted that the amount of nitrogen in underlying soil pore water and groundwater dramatically increased, as illustrated in Table 5 and Figure 10.

Buss et al. (2005) describes the relationship between crop yield and nitrogen application. This is dependent upon site-specific soil conditions, such as soil texture, pH, organic content, crop type, the method of planting, irrigation, and fertilizer application. These researchers developed a typical crop nitrogen response curve and noted that for each crop and set of conditions, there is a nitrogen application rate at which crop yield is maximized and leaching is minimized. These researchers also noted that even with optimal conditions and appropriate application rates, significant nitrate leaching tends to occur.

Cornell University (2012) developed nitrogen guidelines for corn, based on the soil nitrate concentration (Table 6). The pre-sidedress soil nitrate test (PSNT) is a springtime test to determine the amount of nitrogen present in the soils prior to the start of the growing season. This test provides an estimate of the amount of nitrogen which needs to be applied to support crop growth. These researchers also noted that above 25 ppm soil nitrate concentration, that there is no increased economic benefit anticipated by adding additional nitrogen. These findings are similar to those from Sawyer et al. (2006) presented in Figure 10.

Table 5. Diminishing economic returns and yields with increasing nitrogen application.

Annual N Rate (lbs/acre)	Grain Yield (bushels /acre)	Economic Return (dollars/acre)	Nitrate as N in Soil Pore Water at 7.5 ft (mg N/L)
0	82	--	2
75	141	95	4
150	168	130	17
225	164	103	32

(Sawyer et al., 2006.)

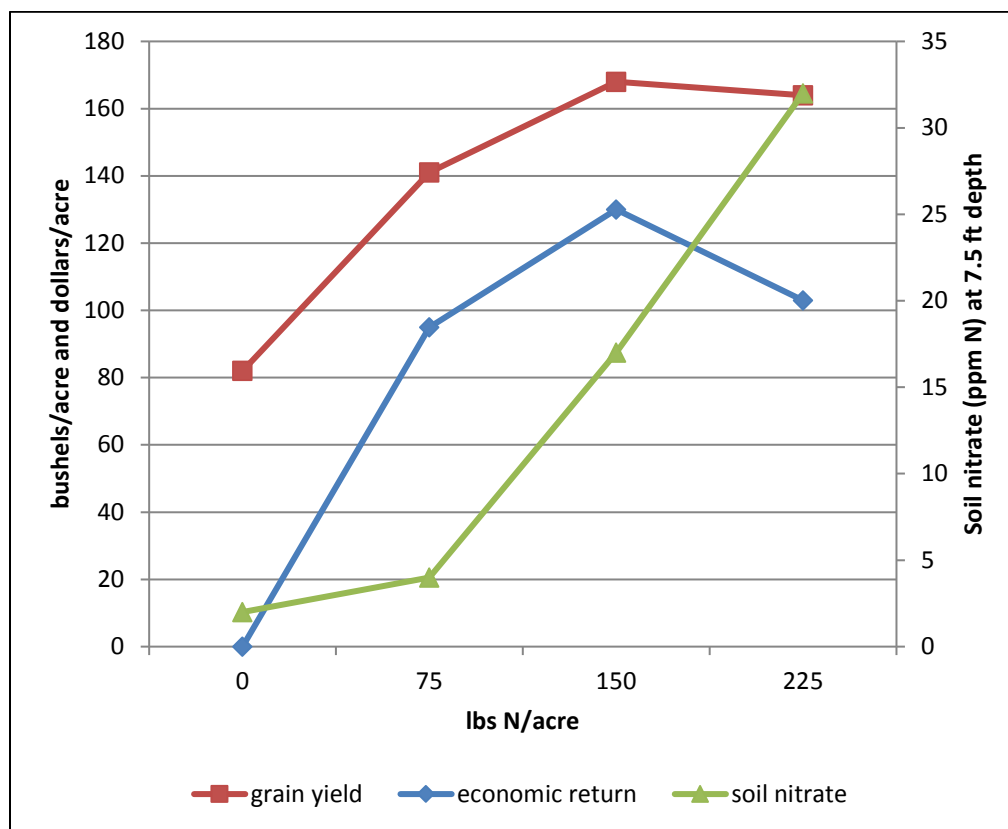


Figure 10. Application rates effect on crop yield, economic return and soil nitrate concentrations (modified from Sawyer et al., 2006).

Yin et al. (2007) concluded that soil nitrate accumulation in China is closely related to nitrate leaching, which can result in elevated nitrate concentrations in groundwater. These researchers concluded that there is a positive correlation between nitrogen application rates and the nitrogen uptake of corn. However, they also found a positive correlation between nitrogen application rates and residual soil nitrogen. Elevated soil nitrate creates a risk of increased nitrogen losses.

Table 6. Cornell University (2012) interpretation of the pre-sidedress nitrate test for corn.

PSNT ppm nitrate (as N)	Likelihood of an economic benefit to extra N	N guideline
≥ 25	Low	No additional N needed
21 - 24	About 10%	If you expect a yield response, consider sidedressing 25 to 50 lbs N/acre
< 21	High	Apply sidedress N according to the Cornell N guidelines for corn

Crop Uptake

Summary

Crop uptake is the primary nitrogen treatment mechanism and removal component for manure land treatment systems. The type of crop grown and the site-specific climatic conditions are the main factors influencing the treatment capacity of the system. Winter cover crops are useful for removing residual soil nitrate still present after the growing season.

Crop uptake is one of the components in a mass balance equation that determines the appropriate application rate (Harter et al., 2012). Land treatment systems rely upon several factors to treat nutrients and minimize potential leaching of excess nitrate to groundwater. Usually, no pre-treatment is provided to the manure before it is land applied, so the crop and the soils are the predominant treatment mechanisms. A land treatment system relies on crop uptake, nutrient removal during harvest, attenuation and degradation in the soils. Once nitrate migrates below the root zone, it will move with water and eventually reach groundwater (Hermanson et al., 2000). Crop uptake and removal is the most significant mechanism for nitrate removal within a land treatment system. (Hermanson et al., 2000)

The timing and amount of nitrogen applied is dependent upon the crop grown and climatic conditions. Winter cover crops planted in fields with annual crops also utilize residual nitrogen and can reduce nitrate leaching to groundwater. The crop type influences the amount of nitrogen that can be utilized from a field. Perennial crops, like grasses with root systems that function for several years between re-plantings, remove more nitrogen than row crops. (Kirchmann et al., 2002; Hermanson et al., 2000)

Literature-Recommended Soil Nitrate Concentrations

Summary

Soil nitrate values are a proven tool for helping to determine nitrogen crop requirements and the effectiveness of manure management.

The research summarized from the literature in this section indicates there are two time periods when soil nitrate samples are typically collected: in the spring before manure application begins and in the fall after the crop has been harvested. These two sampling times have different purposes and provide different information. The fall soil nitrate test provides information on the effectiveness of manure management practices from the previous season. The spring soil nitrate test provides information about the amount of plant-available nitrogen present in the soil at the start of the growing season. This test also provides information for assessing appropriate nitrogen application rates at the beginning of the growing season.

A summary of the recommended soil nitrate threshold limits cited in the reviewed literature is compiled in Table 7. Recommended targets for fall soil nitrate values range from 5 to 24 ppm,

depending upon the site-specific conditions. The recommended targets for spring soil nitrate values range from 16 to 30 ppm.

Soil nitrate only represents the amount of nitrate present in the soils. It does not account for nitrate that has already leached to groundwater, nor does it account for nitrogen mineralization over time.

Researchers agree that soil nitrate tests are not a surrogate for groundwater monitoring. However, the majority of researchers also agree that residual soil nitrate can indicate when excess nitrogen has been land-applied and when there is a risk that groundwater may have been (or could be) impacted from nitrate leaching.

Table 7. Summary of soil nitrate threshold limits determined from the literature.

Soil Threshold	Notes	Reference
Fall (post-harvest) Soil Nitrate Test (at the one-foot depth) -- Acceptable nitrogen		
15 to 20 ppm	Western WA and OR, based on crop type	Sullivan and Cogger (2003)
5 to 15 ppm	Based on a 1 ft and 2 ft sampling depth	Sullivan and Cogger (2002)
15 to 20 ppm	30 ppm is excessive	Bary et al. (2000)
5 to 15 ppm	Biosolids risk management	Ecology (2000)
< 8 or <11 ppm	Maryland, based on crop type	Kratochvil and Steinhilber (2013)
24 ppm	BC Ministry of Agriculture and Lands	Kowalenko et al. (2007)
7.5 ppm	Canadian National Agri-Environmental	Drury et al. (2005)
Spring (Pre-sidedress) Soil Nitrate Test (at the one-foot depth) -- No additional nitrogen application		
25 ppm	Western WA and OR	Staben et al. (2003).
25 ppm or 16	Iowa (use 16 ppm if heavy precipitation in the spring)	Iowa State University (1997)
21 ppm	Wisconsin	Laboski (2008).
25 ppm	New York	Cornell University (2012)
25 ppm	Indiana -- corn	Camberato et al. (2013)
25 - 30 ppm	vegetables	Heckman (2003)
25 ppm	Oregon -- corn	Hart et al. (2009)

Fall Soil Nitrate Test (Post-Harvest Soil Nitrate)

The fall soil nitrate test is intended to evaluate the effectiveness of manure management practices by estimating the residual nitrate that remains in the soil after crop uptake and harvest are complete. It provides the grower with a measure of how effective their manure management practices were at the end of the growing season. The fall soil nitrate test provides a snapshot of the residual nitrate that remains in the soil at the time and the location the sample was collected. This test specifically focuses on nitrate, the nitrogen species that is mobile and moves with

water. This test measures the soil nitrate concentration and does not account for the organic nitrogen present. After the growing season, any nitrate still present in the soils (including the root zone) is susceptible to leaching to groundwater. This measured residual soil nitrate after the last harvest indicates the amount of nitrate that could leach to groundwater. The risk of leaching is also a function of winter recharge and is therefore greater in areas of higher precipitation.

Sullivan and Cogger (2003) recommend the post-harvest (fall) soil nitrate testing as a tool for assessing nitrogen management where manure is land-applied to crops. The post-harvest soil nitrate test was not intended to project impacts to groundwater. This test only indicates the nitrate concentration in the sampled soil profile after harvest; it does not address nitrate concentrations in the lower soil depths or the mass of nitrate that has already leached to groundwater. The interpretations presented in Table 8 are crop-specific and are recommended for producers in western Washington and Oregon. It is designed to evaluate past manure management practices, taking into account the amount of nitrogen applied and the amount removed in the crop.

Table 8. Manure management based on soil nitrate for western Washington.

Post-Harvest Soil Nitrate (0 to 1 foot depth)	Corn	Grass for Hay or Silage
<20 ppm	Continue present N management.	
<15 ppm		Continue present N management.
20-45 ppm	Reduce manure application after August 1. Reduce manure N application by 10 – 25%.	
15-30 ppm		Reduce manure N application by 10 – 25% and apply earlier in the season.
>45 ppm	Eliminate manure application after August 1. Reduce manure N application by 25 – 40%.	
>30 ppm		Reduce manure application after August 1. Reduce manure N application by 25 – 40% and apply earlier in the season.

(Refer to Sullivan and Cogger, 2003, for additional recommendations.)

Sullivan and Cogger (2002) developed a generalized matrix that provides interpretation for soil nitrate values at the two-foot depth (Table 9).

The Columbia Basin Groundwater Management Area (GWMA) in eastern Washington utilizes deep soil sampling to assess nitrate in the soil profile from the land surface down to ten feet (GWMA, 2009). This nitrate profile is used to assess the effectiveness of nitrogen application management practices and determine the extent of accumulation in the subsurface. The Columbia Basin GWMA is a local non-regulatory consortium whose goal is to identify management practices that protect groundwater.

Bary et al. (2000) recommend the use of fall soil nitrate values to determine the effectiveness of manure application. If the fall soil nitrate in the top foot of soil is greater than 15-20 mg/kg (ppm), these researchers suggest that too much nitrogen was applied and they recommend that the amount be reduced in subsequent years. They further state that soil nitrate values greater than 30 mg/kg (ppm) are excessive.

Table 9. Manure management based on soil nitrate at the one and two foot depths.

Risk	Measured Soil Nitrate (ppm)	Estimated Mass of Soil Nitrate for 1-ft sample (lbs/N /acre)	Estimated Mass of Soil Nitrate for 2-ft sample (lbs/N /acre)	Interpretation
Low	< 5	0 - 15	0 - 25	Evaluate crop performance. If yield is low, insufficient N may have been applied. If yields are adequate and irrigation is not excessive, the low nitrate levels reflect excellent management.
Medium	5 - 15	15 - 55	25 - 75	Nitrogen application rate was adequate for the yield with little to moderate nitrate accumulation. Continue current nitrogen application.
High	15 - 30	55 - 105	75 - 150	Considerable N was not utilized by the crop at the end of the growing season. Evaluate N inputs and application. Decrease N application rates or improve management to increase N removal.
Very High	> 30	> 105	> 150	Major management changes may be needed. Check agronomic rate assumptions and calculations.

(Sullivan and Cogger, 2002.)

Biosolids are the treated solids component of human waste. While biosolids receive a higher level of treatment than manure, the established nitrogen classifications can provide a useful comparison to manure. Ecology (2000) establishes risk guidelines for application of biosolids based on soil nitrate levels in Table 10. Soil nitrate is a measure of the fraction of nitrogen that poses a risk to leaching to groundwater if the nitrate is not taken up by plants. This document provides risk classification for biosolids management. When soil nitrate exceeds 30 ppm, the guidelines state that the risk of groundwater contamination is very high and direct the following actions: (1) make major management changes (2) discontinue biosolids application, and (3) re-assess the agronomic rate assumptions and calculations. (Ecology, 2000)

Table 10. Biosolids groundwater contamination risk classification.

Soil Nitrate Level (mg/kg = ppm) in the top 1 foot	Risk Classification
< 5	Low
5 – 15	Medium
15 – 30	High
>30	Very high

(Ecology, 2000.)

Risk to groundwater is a function of soil nitrate concentration, precipitation and irrigation, manure application rates, timing of application and soil properties.

Gehl et al. (2006) determined that elevated post-harvest soil nitrate concentrations indicate that excessive fertilizer was applied. This residual represents the nitrate available to leach. These researchers also noted that there is considerable evidence that verifies nitrate leaching in the presence of elevated soil nitrate concentrations in the lower part of the soil profile. This occurs when residual nitrate that is not utilized by plants migrates downward with excess precipitation or irrigation. Once the residual nitrate is below the root zone it will eventually migrate to groundwater.

Hart et al. (2009) state that in Oregon post-harvest soil nitrate values below 20 ppm indicate that the silage corn utilized the majority of plant-available soil nitrogen. These researchers note that a low fall soil nitrate test does not indicate that too little nitrogen was applied, since continued mineralization of nitrogen is likely to occur and will increase the amount of plant-available nitrate in the soils.

A positive correlation between the spring soil nitrate (PSNT) results and the fall soil nitrate results was found by Hart et al. (2009) based on data collected in Whatcom County, Washington and western Oregon (Figure 11). This correlation indicates that the PSNT can be used as an indicator of whether nitrogen application is necessary; additionally, it can be used as an indicator of post-harvest fall soil nitrate.

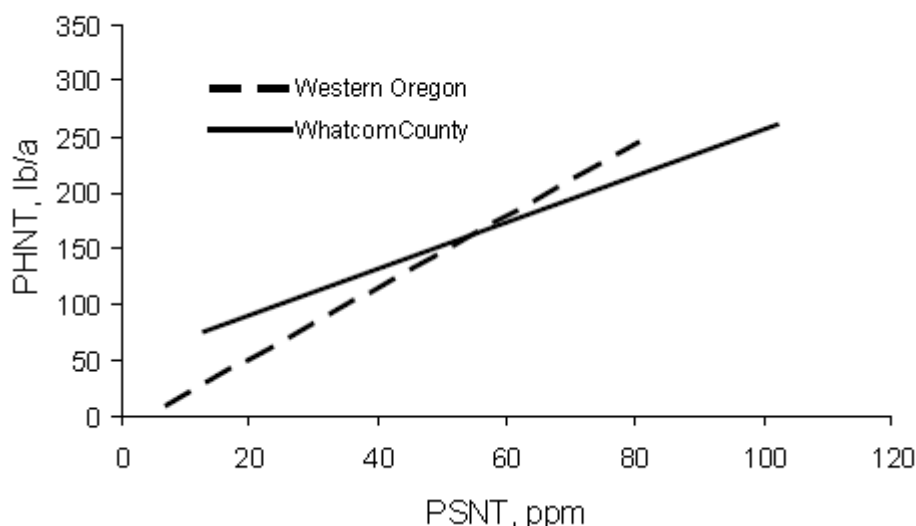


Figure 11. Positive correlation of spring and fall soil nitrate testing (Hart et al., 2009).

PHNT = post-harvest nitrate test; *PSNT* = pre-sidedress nitrate test

Hart et al. (2009) recommends that plant development should be used to time manure applications rather than the calendar.

The British Columbia (Canada) Ministry of Agriculture and Lands proposed soil nitrate concentrations based on risk of groundwater contamination (Table 11) in the Lower Frazier River Valley, which is an area of high winter precipitation. These researchers establish 89 lbs N/acre (24 ppm) in the top two feet as a high risk for groundwater contamination (Kowalenko et al., 2007).

Table 11. British Columbia Ministry of Agriculture and Lands proposed environmental risk classes for soil nutrient interpretation.

Risk Classes	Residual Soil Nitrate + Ammonium in top 2 feet (60 cm)		
	Kg N/ha	Lbs N/acre	ppm
Low	0 to 49	0 to 44	0 - 12
Medium	50 to 99	45 to 88	12 - 24
High	100 to 200	89 to 178	24 - 50
Very High	> 200	> 178	> 50

(Modified from Kowalenko et al., 2007)

Drury et al. (2005) propose an environmental risk matrix based on soil nitrate concentrations (Table 12). These researchers suggest 27 lb N/acre (7.5 ppm) as a high risk for groundwater contamination.

Table 12. Environmental risk matrix based on soil nitrate concentrations.

Risk Classes	National Agri-Environmental Indicator for Residual Soil Nitrate		
	Kg N/ha	Lbs N/acre	ppm
Low	0 to 20	0 to 18	0 - 5
Medium	20 to 30	18 to 27	5 – 7.5
High	30 to 40	27 to 37	7.5 - 10
Very High	> 40	> 37	> 10

(Modified Drury et al., 2005)

Spring Soil Nitrate Test (Pre-sidedress Soil Nitrate Test)

Spring soil sampling is used to assess conditions before the growing season and to calculate the application rates based on the crop needs. Northwest researchers advocate using the Pre-sidedress Soil Nitrate Test (PSNT) for corn in western Oregon and western Washington. They recommend that no additional nutrients be added if the soil nitrate value is greater than 25 ppm (Staben et al., 2003). This test measures nitrate concentrations only and does not account for organic nitrogen that is present in the soil.

Iowa State University (1997) researchers recommend that the nitrogen application rate be based on the springtime soil nitrate test in conjunction with springtime precipitation. Their work concludes that no manure be applied if the soil nitrate value is greater than 25 ppm, or greater than 16 ppm if there is excessive springtime precipitation (Table 13).

The University of Wisconsin recommends the use of the PSNT to assess the nitrogen in soils prior to application of additional nutrients. It was concluded that if the concentration of nitrogen in the one foot-depth soil sample is greater than 21 ppm, then no additional fertilizer is needed for the crop (Laboski, 2008).

Table 13. Manure nitrogen recommendations for soils based on the springtime soil nitrate test.

Soil Nitrate Test	Recommended Nitrogen Rate	
	Excessive rainfall (>5" in May)	Normal rainfall
(ppm N)	(lbs N/acre)	
0 – 10	90	90
11 – 15	60	60
16 – 25	0	30
>25	0	0

(Iowa State University, 1997)

Hart et al. (2009) note that in western Oregon early season nitrogen uptake by corn is minimal, and that rapid nitrogen uptake occurs between the 10-leaf stage and silk emergence, which is approximately mid-June through the end of July (Figure 12). During this time of rapid uptake, corn uses close to 3 lbs/acre/day. These researchers recommend that the PSNT should be conducted when the corn has 5 or 6 leaves, in order to measure soil nitrate during the growing season before the crop's greatest nitrogen demand.

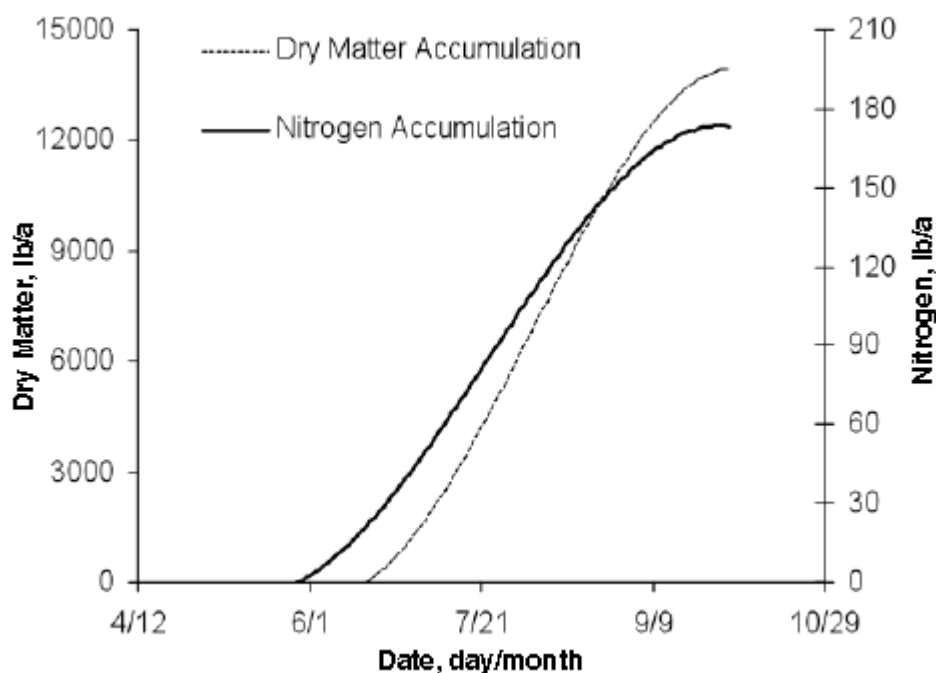


Figure 12. Nitrogen and dry matter accumulation and timing of uptake silage corn.
Based on data from California, Oregon and New Jersey, 1984-2009 (Hart et al., 2009).

Table 14 provides nitrogen application recommendations based on soil nitrate values. These researchers recommend that if the PSNT value is greater than 25 ppm, then the addition of nitrogen is not needed. Hart et al. (2009) also developed guidelines that interpret the soil nitrate results. These are presented in Table 15.

Table 14. Nitrogen application rate recommendations for western Oregon using the pre-sidedress nitrate test in the spring.

PSNT soil Nitrate value (mg N/L)	Application Rate (lbs/acre)
0 – 10	100 – 175
11 – 20	50 – 100
21 – 25	0 – 50
> 25	0

Hart et al. (2009).

Table 15. Soil nitrate test results interpretation.

Soil Nitrate (mg N/L)	Interpretation
0 – 20	Acceptable
21 – 45	High
> 45	Excessive

Hart et al. (2009).

Hart et al. (2009) also note that soil nitrate values can be as low as 5-10 ppm without compromising yields.

Cornell University (2012) recommends to their producers that if the PSNT results are greater than or equal to 25 ppm, then no additional nitrogen is needed for a corn crop.

Soil nitrate concentrations can be used to assist in planning nitrogen management on a field with optimum concentrations of 25 ppm for corn in Indiana (Camberato et al., 2013).

Heckman (2003) found that when the soil nitrate concentration was 25 to 30 ppm that there was sufficient available nitrogen to grow a wide variety of vegetables (sweet corn, celery, lettuce, cabbage, peppers, pumpkin, winter squash, and sugar beets). Heckman also provides a soil nitrate test interpretation in Table 16.

Table 16. Interpretation of soil nitrate tests for annual crops (Heckman, 2003).

Soil Nitrate (as N) Test (ppm)	Interpretation
< 20	Very likely N deficient, sidedress N is recommended.
20 to 24	May be sufficient for some crops. A low rate of sidedress N may be applied to ensure that N is sufficient.
25 to 30	Sufficient N is available for most crops. Sidedress N is usually not recommended.
> 30	Sidedress N is not recommended.
> 50	Excessive. Indicates excessive application of manure, compost, or other sources of N.

Timing of Nutrient Application

Summary

Most studies indicate that there are two primary manure management mechanisms that affect leaching of nitrate to groundwater: (1) application rates and (2) the timing of manure application. Timing is important since it is recommended to coincide with the crop growth, but it is also related to nitrogen transformations in the subsurface that create the opportunity for mineralization and the opportunity to leach.

Researchers agree that applying nutrients during the non-growing season (when plant uptake of nitrogen has ceased or has significantly slowed) poses an increased risk to surface water from runoff and an increased risk to groundwater from leaching of nitrate and other contaminants. Additionally, there is general consensus that there is a higher risk of leaching and runoff when precipitation and irrigation exceeds evapotranspiration and when crop growth is slow.

Researchers advise that nutrients only be available when crops can utilize them, and they generally characterize this timeframe as the growing season.

References are provided that use site-specific climate data to assist in determining the appropriate timeframes for land application to specific crops.

Hermanson et al. (2000) note that nitrogen application at the time of maximum crop demand will result in the maximum removal of nitrogen from the soils. Nitrogen applied without regard to the seasonally variable nitrogen demand will result in a buildup in the soils of inorganic nitrogen that is susceptible to leaching. Late season applications in areas of Washington with high winter precipitation are particularly risky from a leaching standpoint.

Figure 13 illustrates the typical nitrogen uptake during the growth cycle of an annual crop. During phase 1 (typically spring), early plant growth, nitrogen uptake is relatively slow. Phase 2 (typically late spring to early summer) is the period of rapid plant growth and rapid nitrogen uptake. During phase 3 (typically summer through early fall), vegetative growth has slowed or stopped. Nitrogen uptake during phase 3 is slow and the addition of nutrients during this time is not effective in increasing crop yields (Heckman, 2003).

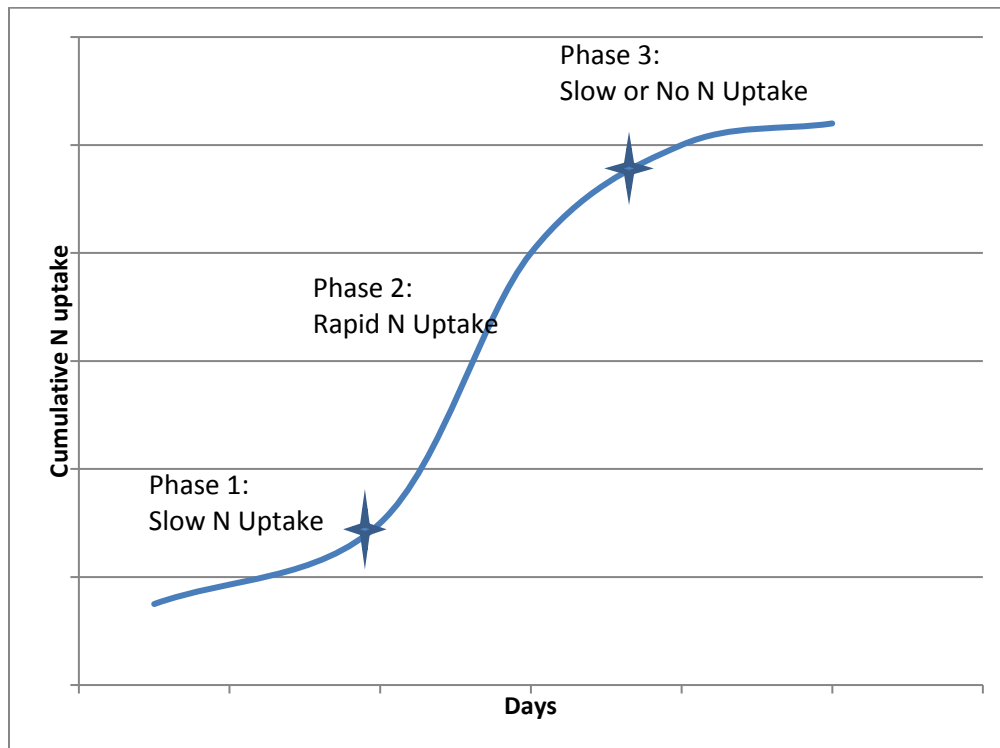


Figure 13. Nitrogen accumulation curve (modified from Heckman, 2003).

Chesnaux et al. (2007) determined that controlling the timing of application is important to minimizing leaching to groundwater.

Cogger (2000) found that there is no yield response to sidedress nitrogen applied in the spring to corn fields receiving manure in the Abbotsford-Sumas aquifer. The additional applied 135 kg N/ha resulted in an average increase of post-harvest soil nitrate of 130 kg N/ha in 1995 and 100 kg N/ha in 1996. This indicates that most of the nitrogen (96% in 1995 and 74% in 1996) applied in the spring was not used by the crop and was vulnerable to leaching in the fall.

Sawyer et al. (2006) determined that nitrogen is used more efficiently if applied during the growing season just prior to the time of maximum uptake. Most crops have the greatest capacity to take up soil nitrogen when it is applied in the spring or summer.

In Arkansas, Slaton et al. (2004) found that applying additional nitrogen in the fall is not necessary, even when a winter cover crop is grown. They concluded there was sufficient nitrogen in the soils to fully support crop growth. This conclusion was based on comparing crop yields to measured values of soil nitrate and ammonium concentrations. These combined soil nitrogen values ranged from 9.2 ppm to 42.8 ppm.

Sullivan (2008) concluded that in western Washington and Oregon, residual fall soil nitrate is usually sufficient to support winter cover crops. Sullivan also concluded that applying manure in the fall is generally not an efficient use of nitrogen. Jégo et al. (2008) recommended that in areas where groundwater was vulnerable to contamination and where agricultural practices are present, that nitrogen-based fertilizers not be applied in the fall.

Hermanson et al. (2000) state that winter cover crops temporarily store nitrogen from the root zone and return it to the surface, making it available for crop uptake the next season. Furthermore, these researchers conclude that the use of winter cover crops does not imply that nitrogen can be applied above agronomic rates without increasing the amount of nitrate leaching. If excess nitrogen is applied in one growing season, it must be offset by decreased nitrogen application in the following season to avoid residual nitrogen buildup and subsequent nitrate leaching (Hermanson et al., 2000).

Carey and Cummings (2012) characterize the vulnerable times for groundwater when nutrient application causes an increased risk. The timeframe from mid-September to mid-March were identified as the most vulnerable in Whatcom County, WA. The average precipitation is highest when the percent grass growth is lowest. Applying manure or fertilizer during this vulnerable period presents a high risk for leaching and runoff.

Vulnerable time periods also exist in eastern Washington, depending upon the crop growth, manure applications, irrigation, and precipitation events.

Carey and Harrison (2014) note that groundwater nitrate concentrations are sensitive to departures from agronomic nutrient application on the land surface. A manure application in early October 2006, just outside the normal growing season, resulted in an almost immediate increase in soil nitrate at the one-foot level, followed by an increase of up to 16 mg N/L in shallow monitoring wells beneath the field in Whatcom County.

The NRCS (2005) characterizes the winter period from October 1 to March 1. Application during this timeframe needs to meet specific risk criteria to minimize nutrients and associated contaminants from moving to surface water and groundwater. Some of these criteria include characterizing the fall soil nitrate content, assessing the potential for ponding, flooding, heavy rains, and frozen or saturated ground.

Nitrogen applied to land in the form of ammonium or organic nitrogen will convert to nitrate during the non-growing season and will leach out of the soils and migrate to the groundwater. Applying wastewater to the land during the non-growing season does not reliably protect the groundwater (Ecology, 2004a).

Hermanson et al. (2000) noted that organic nitrogen applied during the non-growing season will partially or totally convert to nitrate before the next growing season and that the amount converted will depend on the type of nitrogen applied, soil temperature, and moisture conditions. These researchers concluded that applying organic nitrogen during the non-growing season has an inherent risk of leaching nitrogen to groundwater. Additionally they advocate that steps be taken to minimize movement of nitrogen below the root zone during the growing and non-growing season. These researchers emphasize that there are enough uncertainties and uncontrollable variables associated with nitrogen dynamics in the subsurface to conclude that applying nitrogen to crops and soil systems during the non-growing season is not reliably protective of groundwater.

Application Risk Management

The Whatcom County Conservation District developed the Application Risk Management approach (ARM) to determine under what conditions, if any, manure could be safely land-applied during the non-growing season without an adverse impact to groundwater or surface water. This publication has recently been released, but the data from the multi-year ARM investigation were not included in this review for several reasons. It is not a peer-reviewed study and therefore does not fulfill the criteria of acceptable literature in this review. Also, the groundwater section of the study is being developed by USGS and is not yet complete. Finally, the raw data are not accessible beyond the District and EPA. Any conclusions about the degree of protection provided from this study cannot be included in this review until these issues are addressed. It is suggested that EPA, who funded the study and has access to the data, provide comment on the degree of protectiveness when the study, including the USGS groundwater portion, is complete.

Timing Considerations for Surface Water Quality

Fleming and Fraser (2000) conducted a literature review of the impacts of manure application on surface water quality. These researchers found that there were many similarities and a few conflicts in the literature, but they identified common themes:

- Nitrogen runoff to surface water from winter manure applications varies from negligible to over 20% of the applied nitrogen.
- The manure lost to surface water runoff is generally greatest during the winter months.

- Frozen soils are generally impervious, and snowmelt and rainfall are likely to cause manure runoff to surface waters.
- If the ground is not frozen, the water will infiltrate and leach to groundwater.
- The risk to surface waters is the same, whether the manure is applied to frozen bare ground or snow-covered ground.
- Manure application to a winter cover crop does not necessarily reduce the risk of runoff to surface waters.
- Some researchers concluded that applying solid manure can reduce the amount of runoff and associated soil erosion by acting as a mulch to slow the flow of runoff.
- Spreading solid manure, as a form of mulch, can impede runoff.
- One of the studies reviewed evaluated runoff rates for two sites with slopes of 10% and 20%. They found they had similar runoff rates. (Lower slopes were not evaluated in this one study).
- Pathogen die-off cannot be guaranteed during the winter months. Pathogens can remain viable in cold weather for extended periods of time.
- Volatilization rates are diminished during the winter, especially if the manure is covered by snow.

Fleming and Fraser (2000) concluded that weather is the biggest factor controlling surface water runoff.

Tools to Determine Timing of Manure Applications

Climate plays an important role in determining when manure should be applied. The following references provide site-specific information based on climate and crops to assist in determining the appropriate land application timeframes.

Washington Irrigation Guide

The Washington Irrigation Guide (NRCS, 1997) is a resource that reports the growing season for different crops based on the climate in different areas of the state. This guide provides both crop irrigation requirements as well as crop consumptive use estimates. It includes mean temperature and total precipitation by month.

http://www.nrcs.usda.gov/wps/portal/nrcs/detail/wa/technical/engineering/?cid=nrcs144p2_036314

Figure 14 is an example of the information contained in the Washington Irrigation Guide for Aberdeen, Washington. The crop consumptive use (CU) is the amount of water the specific crop needs. The crop irrigation requirement (CIR) is consumptive use minus the effective precipitation. The CIR is the amount of water that is needed to sustain the crop by meeting the consumptive use requirements in the different months. While this guide does not necessarily address nutrient requirements, the CU term provides information on the crop growing season, which is indicative of when crops would utilize nutrients.

APPENDIX B															
WASHINGTON STATE - WEST OF CASCADES															
CIR - CROP IRRIGATION REQUIREMENT and CU - CROP CONSUMPTIVE USE															
ABERDEEN	46.97 LATITUDE														
	JAN	FEB	MAR	APR	MAY	JUN	JUL	AUG	SEP	OCT	NOV	DEC	TOTAL		
MEAN TEMPERATURE (F)	40.0	42.5	44.5	48.4	53.0	57.2	60.2	60.8	58.9	52.7	45.4	41.5			
TOTAL PRECIPITATION (IN)	12.38	9.96	9.17	5.68	3.57	2.60	1.45	1.78	3.73	7.57	11.37	13.75	83.01		
REFERENCE CROP ET (IN)	0.11	0.56	1.37	2.19	3.22	3.51	3.91	3.10	2.47	1.46	0.36	0.04	22.30		
EFFECTIVE PRECIP (IN)	0.11	0.56	1.35	2.16	2.31	1.80	1.04	1.22	2.15	1.43	0.36	0.04	14.53		
PASTURE/TURF															
	BEG	END	JAN	FEB	MAR	APR	MAY	JUN	JUL	AUG	SEP	OCT	NOV	DEC	SEASON
CIR	1/ 2	12/31	0.00	0.00	0.00	0.00	0.75	1.54	2.67	1.72	0.19	0.00	0.00	0.00	6.88
CU			0.10	0.53	1.30	2.08	3.06	3.33	3.71	2.95	2.35	1.39	0.34	0.04	21.19
FIELD CORN															
	BEG	END	JAN	FEB	MAR	APR	MAY	JUN	JUL	AUG	SEP	OCT	NOV	DEC	SEASON
CIR	6/11	10/29	0.00	0.00	0.00	0.00	0.00	0.00	1.52	2.02	0.56	0.00	0.00	0.00	4.10
CU			0.00	0.00	0.00	0.00	0.00	1.11	2.54	3.26	2.72	1.20	0.00	0.00	10.82

Figure 14. Washington Irrigation Guide information example (NRCS, 1997).

Wetlands Climate Guide

The WETS tables are geographic area-based tables incorporating historic temperature and precipitation information for establishing the normal growing season range (NRCS, 1995).

http://www.wcc.nrcs.usda.gov/climate/wets_doc.html

WETS Station : DE SMET, SD2302								
Latitude: 4423			Longitude: 09733			Elevation: 01750		
State FIPS/County(FIPS): 46077			County Name: Kingsbury					
Start yr. - 1971			End yr. - 2000					
Temperature:			30 years available out of			30 requested in this analysis		
Precipitation:			30 years available out of			30 requested in this analysis		

Temperature (Degrees F.)				Precipitation (Inches)				
-----				-----				
Month				avg	30% chance will have		avg	avg
					# of	total		
					days			
	avg	avg	avg		less	more	w/.1	snow
	daily	daily			than	than	or	fall
	max	min					more	

COL #1	COL #2	COL #3	COL #4	COL #5	COL #6	COL #7	#8	#9
January	23.0	2.4	12.7	0.60	0.31	0.78	2	6.6
February	29.3	9.0	19.2	0.68	0.41	0.89	2	7.0
March	41.3	21.1	31.2	1.60	0.87	1.95	3	8.9
April	58.7	34.2	46.5	2.26	1.28	2.75	4	1.6
May	71.1	45.8	58.5	3.05	1.82	3.69	5	0.0
June	80.3	55.8	68.0	4.02	2.59	4.84	6	0.0
July	86.2	61.1	73.7	3.25	1.96	3.93	4	0.0
August	83.9	58.6	71.3	2.44	1.51	2.95	4	0.0
September	73.7	48.7	61.2	2.14	1.03	2.61	4	0.0
October	61.0	36.8	48.9	1.78	0.83	2.25	3	0.8
November	41.7	22.5	32.1	0.92	0.34	1.11	2	5.4
December	26.7	8.1	17.4	0.58	0.32	0.73	1	6.0

Annual					19.83	26.03		

Average	56.4	33.7	45.1	-----	-----	-----	--	----

Total	-----	-----	-----	23.30	-----	-----	40	36.3

GROWING SEASON DATES			
Requested years of data:	30	Available years of data:	30
Years with missing data	24 deg = 0, 28 deg = 0, 32 deg = 0		
Years with no occurrence	24 deg = 0, 28 deg = 0, 32 deg = 0		
Data years used	24 deg = 30, 28 deg = 30, 32 deg = 30		

Temperature			

Probability	24 F or higher	28 F or higher	32 F or higher

COL #10	COL #11	COL #12	COL #13
Beginning and Ending Dates			
Growing Season Length			
50 percent *	4/16 to 10/22 189 days	4/27 to 10/ 9 165 days	5/ 4 to 9/29 148 days
70 percent *	4/12 to 10/27 198 days	4/22 to 10/13 174 days	5/ 1 to 10/ 3 155 days

* Percent chance of the growing season occurring between the Beginning and Ending dates.

Figure 15. Wetlands Climate Guide information example (NRCS, 1995).

T-Sum 200

This is a method that was developed for western Oregon to calculate when to begin pasture fertilization. It is tied to the commencement of spring grass growth and involves summing daily heat units. A heat unit is defined as the average of each day's high and low temperatures. Starting on January 1, the average of the daily maximum and minimum air temperatures in °C (centigrade) is added for each day until the cumulative total reaches 200°C. At this point, the method assumes that at this time plants will begin growing and will require nutrients. Alternatively, this method can also use 360° Fahrenheit as the heat unit threshold. To calculate daily heat units in degrees Fahrenheit the following formula is used:

$$\text{Daily Heat Units} = [(\text{daily maximum temperature } ^\circ\text{F} + \text{daily minimum temperature } ^\circ\text{F}) \div 2] - 32$$

(Oregon State University Extension Service, 1996)

Soil Mechanics

Summary

The soil horizon and vadose zone are the locations where nitrogen treatment and transformations occur. These processes include crop uptake and removal, volatilization, mineralization, denitrification, and leaching. These processes are influenced by a variety of site-specific factors, and they also affect the fate and transport of chemicals in the environment.

Volatilization of nitrogen compounds after manure has been incorporated into the soils is minimal with documented rates of approximately 5%.

Mineralization is the transformation of organic nitrogen to ammonium and it occurs year-round, even during the winter months, although the rate varies seasonally. Studies have shown that mineralization and nitrification can occur at significant rates in frozen soils especially in the presence of organic matter. In Canada, investigations have demonstrated that soil organic nitrogen and immobilized nitrogen contributed one-third to one-half of the nitrogen lost during the non-growing season.

Several researchers noted that tilling or disturbance to the fields can stimulate mineralization.

Generally, the reviewed studies concluded that mineralization is a significant source of nitrogen in fields where manure has been applied. Additionally it was concluded that due to the continued mineralization during cold and freezing temperatures that fall-applied nitrogen poses a risk of leaching to groundwater, particularly in fine-textured soils.

Denitrification: Researchers conclude that denitrification may reduce nitrate loading to groundwater by 5% to 16% under some conditions, though they determined that it is of little importance in well-drained soils. It was observed that some degree of denitrification occurs at

all sites, but high denitrification rates reported in some literature are not representative of shallow sandy aquifers.

Nitrogen Storage in the Subsurface: *Researchers have cautioned that the practice of storing nutrients in the soils during the winter for use by crops in the spring poses a risk to groundwater. Mineralization continues during the winter months. Climatic conditions such as temperature and precipitation, and subsequently nitrogen transformations, are not elements that can be precisely controlled. These uncontrolled elements can promote nitrate leaching. Soil storage of nutrients is a practice that has not been proven to protect groundwater.*

The vadose zone includes the soil horizon and the unsaturated zone between the ground surface and the water table where applied nitrogen can be transformed, utilized, and mobilized (Figure 16). The primary forms of nitrogen in manure are organic nitrogen and ammonium. Organic nitrogen must be transformed into plant-available nitrogen (ammonium or nitrate) before it can be taken up by plant roots. These transformations are part of the nitrogen cycle and are explained in more detail in Appendix A.

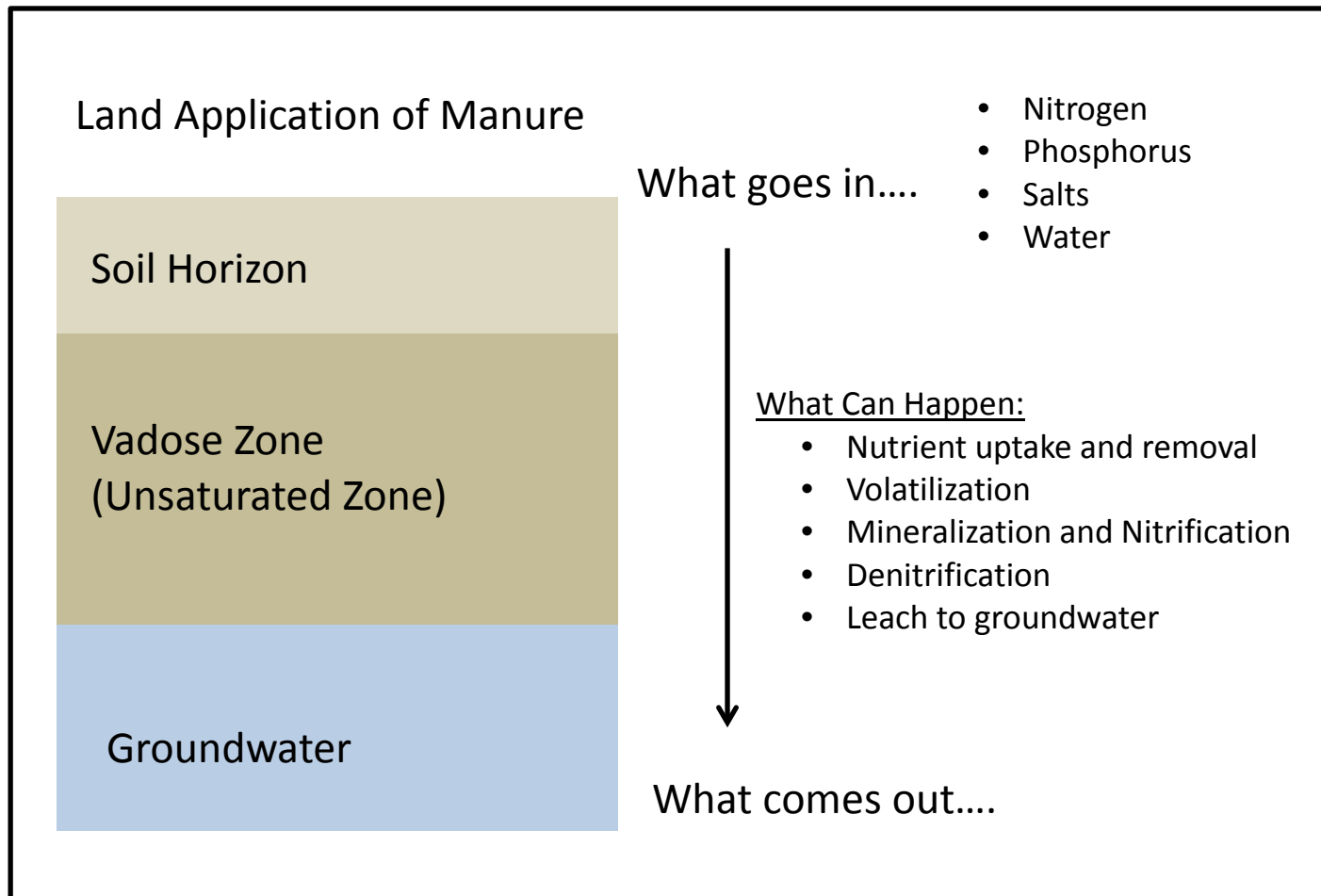


Figure 16. Soil treatment and transformation processes.

A University of California (2005) committee of experts recommends that a total nitrogen mass balance can be used to predict atmospheric losses prior to land application, but that it requires extensive data management and record keeping but it is also associated with significant estimation errors. These researchers advocate measuring or estimating losses based on site-specific conditions, and they caution against using one percentage for all situations.

Land treatment systems use the soil horizon and the vadose zone for attenuation, uptake, and degradation of nutrients and other constituents. Mineralization is the process that converts organic nitrogen into ammonium, while nitrification is the process that converts ammonium into nitrate. Volatilization and denitrification losses are minimal and occur under specific conditions (van der Schans et al., 2009; Sullivan et al. 2000; Dzurella et al., 2012; Hermanson et al., 2000). Nitrate and ammonium are the plant-available forms of nitrogen that are available for crop uptake in the root zone during the growing season. Once nitrate migrates below the root zone, it is no longer available to the crop and will eventually migrate to groundwater (Hermanson et al., 2000).

Green et al. (2008a) investigated the nitrogen transport processes that occur in the vadose zone at several sites across the United States including Washington State. Their goal was to verify previous findings that suggest a positive correlation exists between higher nitrate concentrations in groundwater and thicker unsaturated zones. These findings are inconsistent with vulnerability assessment methodology which assumes longer transport times in the vadose zone result in lower nitrate concentrations in groundwater due to denitrification. These researchers concluded that advective transport is the predominant process that influences nitrogen below the root zone. They found nitrogen fluxes to the water table ranged from 7 to 99 kg/ha/year. Values at the high end of the nitrogen range were measured at coarse-grained sites with high nitrogen application rates. They concluded that nitrogen application rates, water application, and evapotranspiration were the dominant factors that accounted for the differences between nitrogen concentrations at sites, not denitrification.

Volatilization

The percent of nitrogen in land-applied manure that is lost to volatilization in the soil was estimated by van der Schans et al. (2009) to be 5% or less.

Nitrogen losses were measured from different units within a farm, including animal exercise yards and feeding areas, liquid manure holding ponds, and land application areas. These researchers found that there was an average volatilization loss of 35% from the liquid manure ponds. Additionally these researchers concluded that volatilization and denitrification losses in the vadose zone are not significant. (van der Schans et al., 2009)

The University of California (2005) committee of experts estimates atmospheric losses from liquid manure range between 20 and 40% and that this figure does not include losses occurring during land application.

Mineralization

Nitrogen mineralization generally increases during warmer weather and slows during cooler weather, but the fraction of nitrogen that mineralizes and becomes available for crop uptake or leaching to groundwater is difficult to accurately estimate. Watts et al. (2007) state that nitrogen mineralization is most influenced by temperature and note that the greatest mineralization occurs at 77°F (25°C). However, recent studies show that significant mineralization occurs during the winter months, creating an additional soil nitrate load that is susceptible to leaching.

Lamb (2012) observed that the conversion of organic nitrogen to nitrate nitrogen typically occurs when the soil temperatures are greater than 50°F (10°C), but that this transformation continues at a decreased rate at lower temperatures until soil temperatures reach 43°F (6°C). Kowalenko et al. (2007) concluded that mineralization of soil nitrogen does not cease in the fall after the crop has been harvested but continues during the winter months in British Columbia.

Clark et al. (2009) investigated the fate of fall nitrogen application of manure (pig slurry) to loamy and clay soils. These researchers observed that nitrification and mineralization continued during the wintertime in frozen soils, but mineralization and nitrification were higher in clay soils and immobilization was higher in loamy soils. They also noted that at temperatures between -2°C and 2°C a significant portion of the ammonium in the slurry was nitrified, but little immobilization occurred even with soil amendments of organic matter. They found that nitrogen immobilization ceases at a higher temperature than nitrification, potentially resulting in an excess of nitrate. These researchers concluded that microbial activity occurs in frozen soils, with mineralization and nitrification occurring at significant rates, especially in the presence of organic matter. These researchers concluded that fall-applied nitrogen could pose a risk of leaching to groundwater, particularly in fine-textured soils.

Cookson et al. (2002) noted significant mineralization in temperate soils (2-15°C) amended with clover residues at temperatures as low as 2°C. These researchers concluded that nitrogen amendments applied in the winter pose a risk to groundwater leaching and recommend that nitrogen applications be limited until the spring.

Winter groundwater nitrate levels at the downgradient edge of a raspberry field at the top of the Abbotsford-Sumas aquifer in southern British Columbia indicated a continued source of newly mineralized nitrate throughout the winter (Kuipers et al., 2014).

Moberg et al. (2013) evaluated mineralization rates in manured agricultural soils during different seasons. Based on samples collected from October to February, the mean soil nitrate concentration was 24.4 mg/kg (ppm). These researchers reported an annual precipitation rate of 118 cm/yr (3.9 feet/yr). If this mineralized soil nitrogen were mixed with recharge, it would result in approximately 8 mg N/L of nitrate available to leach to groundwater.

Chantigny et al. (2014) state that the residual soil nitrate measured at harvest represents the risk of nitrate loss to groundwater during the non-growing season. These researchers utilized ¹⁵N isotopes as a tracer of nitrogen from applications of pig slurry. Their data indicate that 30% to 60% of nitrogen applied in the spring was still present in the soils at the fall harvest. Further,

these researchers found that in clay soils, 16% of the nitrate was lost to groundwater and 45% was lost to groundwater in sandy soils. This work provides evidence that soil organic nitrogen and immobilized nitrogen contributed one-third to one-half of the nitrogen lost during the non-growing season in Canada. Similar research by Jayasundara et al. (2010) discovered nitrogen losses of 16% to 29% from the fall application of pig slurry during the non-growing season. These researchers caution that measuring the fall soil nitrate concentration is inadequate to completely assess the risk of nitrate leaching to groundwater.

Uncertainty in the timing and rate of mineralization and nitrification makes it challenging to accurately estimate the amount of plant-available nitrate in the soil. These rates are dependent upon the amount of organic matter, climate, temperature, and biological activity. Dessureault-Rompré et al. (2010) assessed the variables that predict nitrogen mineralization rates in agricultural soils. These researchers found that soil bacteria in colder climates (mean annual temperature less than 2°C) adapt more readily to declining temperatures compared to bacteria in warmer climates (mean annual temperature greater than 6°C). This research verifies that mineralization occurs in the winter and that rates can be higher in colder climates. These researchers also noted a greater mineralization response to temperature in agricultural soils than in forested soils.

Denitrification

Sullivan et al. (2000) determined that denitrification rates in manured soils typically range from 5% to 15% with the highest rate of 16% noted in October and November after the soil was saturated following a dry summer. These researchers also noted that the remaining soil nitrate (85% to 95%) is lost to groundwater through leaching.

Denitrification requires low oxygen environments and the presence of electron donors, such as organic matter or reduced minerals. Green et al. (2008b) investigated natural attenuation of agricultural nitrate contamination in four areas within the United States, including the Yakima watershed in Washington State. This research utilized methods to analyze all nitrogen species simultaneously to determine nitrogen transformations. In Yakima they found that the zones of denitrification were not uniform across the watershed and were not consistent. They determined that this variability resulted from differences in land use and the intensive application of manure in some areas.

Green et al. (2008b) state that some degree of denitrification occurs at all sites but that the high denitrification rates reported in the literature are not representative of shallow sandy aquifers. These researchers concluded that many of the denitrification values reported in the literature were far higher than what they observed. They caution that assuming high denitrification rates based on reported literature values may be skewed due to method limitations and biased site selection, since in some denitrification studies the sites are often chosen for promoting higher denitrification rates. These researchers concluded that the electron donor concentrations from recharge were insufficient to promote high denitrification rates. They estimate that it would require decades or longer to denitrify the existing groundwater contamination to background conditions.

Singleton et al. (2007) investigated denitrification in shallow groundwater under dairy operations. These researchers anticipated that high rates of denitrification would occur under lagoons due to the saturated conditions created by the continuous seepage. However, they concluded that the prevalence of this phenomenon is unknown, due to the uncertainties in assessing the spatial extent of anaerobic conditions, the transport of organic carbon in different environments, and differing nutrient management practices.

Dzurella et al. (2012) estimate that 10% of the nitrogen in the applied manure in the California Central Valley is lost to denitrification.

Hermanson et al. (2000) noted that denitrification may reduce nitrate loading to groundwater under some conditions, though it is of little importance in well-drained soils.

Storage of Nitrogen in the Subsurface

Nitrogen storage in the subsurface during the winter months poses a high risk to groundwater, particularly where winter recharge is high. Manure contains primarily organic nitrogen and ammonium, which do not readily move with downward flowing water and are adsorbed to the soil matrix. Mineralization increases as temperatures increase, converting the organic nitrogen to ammonium and then to nitrate (Moberg et al., 2013). Mineralization has been documented to continue through the winter and in freezing conditions. The end of winter and early spring when temperatures rise are also typically times of high precipitation and low crop uptake. If nitrogen is converted to nitrate and is not utilized by a crop, recharge can cause nitrate to migrate below the root zone and leach into groundwater. Since temperature, microbial activity, precipitation, and recharge cannot be controlled at a field scale, and the amount of nitrogen cannot be precisely gauged to the crop's limited needs at this time, researchers have stated that the application of nitrogen outside of the growing season is a risk to groundwater (Hermanson et al., 2000; Qui et al., 2005).

Erickson and Matthews (2002) conducted an extensive five-year groundwater monitoring study to evaluate the effectiveness of storing manure over the winter in a newly constructed lagoon in Thurston County. Monitoring was conducted before and after lagoon construction. This allowed a comparison of impacts to groundwater when manure is applied year-round vs. manure applied only during the growing season. These researchers noted improvements in groundwater nitrate concentrations, although the mean concentrations remained elevated above the drinking water standard at the end of the study. They concluded that the improvements to groundwater quality were attributed to the lagoon which facilitated the following: (1) increased volatilization of ammonium during storage, (2) dilution of nutrients with the addition of precipitation to the lagoon, and (3) uniform application of manure to fields during the growing season (April through October).

Hermanson et al. (2000) conclude that it is not protective of groundwater quality to store nitrogen in the soils during the winter months when crops are dormant and to assume the nitrogen will remain available in the root zone when crops will utilize it.

Vegetative Buffers and Setbacks to Surface Water

Summary

Vegetative buffers and setbacks to surface waters are management tools that can protect surface water quality. A setback is defined as the distance from an area where an activity is occurring, such as manure application to fields and to waters of the state. A vegetative buffer zone is an undeveloped area composed of different types of vegetation directly adjacent to the body of water (Mathieu, 2012). Vegetative buffers and setbacks to surface water are transition areas between land uses that provide protection to surface water from manure runoff.

The primary purpose of a vegetative buffer zone is to reduce runoff to surface water by increasing infiltration into soil where nutrients, pathogens, and other contaminants can be utilized by vegetation or attenuated in the subsurface. Buffer zones also help to stabilize soils, reduce erosion, enhance wildlife habitat, and improve water quality (Mathieu, 2012). The NRCS recommends the use of vegetative buffers and setbacks if no other BMPs that reduce discharges to surface water are installed. They also note that when vegetative buffers and setbacks are used in combination they are more effective.

Researchers evaluate pathogen viability and transport distances based on different climatic conditions, vegetation type, and soils. Pathogens, such as enteric bacteria have the ability to survive in manure for as long as 3 to 4 months, depending upon the temperature and water content. (Wang et al., 2004; Sinton et al., 2007). Sinton et al. (2007) noted that in the first 1 to 3 weeks after manure application, populations of most bacteria increased in most conditions, with temperature being the primary influence on growth. They noted that bacteria levels remain elevated if moisture content was above 80%. Reductions of 90% of original bacteria populations took as long as 56 days, with the primary influence being desiccation caused by water content below 70% to 75%. Wang et al. (2004) concluded that manure management practices that store manure at temperatures greater than 41°C will decrease populations of E. coli and fecal coliform bacteria but not fecal streptococci.

Fecal coliform bacteria can survive in soil at high concentrations for a period of weeks to months after land application (Nennich et al., 2005). This poses a risk to water quality, since manure can still impact surface water at a later date. Bacteria can be remobilized with successive precipitation events, eventually migrating to surface waters. This means that the first rainfall after a manure application might not result in impacts to adjacent surface waters, but later events may (Hubbs, 2002; Nunez-Delgado et al., 2002; Sinton et al., 2007).

Increased buffer zone width allows a greater distance for contaminants to be removed and greater vegetation growth, which enhances the removal of nitrate. The Minnesota Pollution Control Agency (2000) states that vegetative buffer strips are an effective means of protecting surface water quality. They advocate that vegetative buffers be at least 60 feet wide and contain a variety of trees, shrubs, plants, and grasses.

Infiltration is greater in permanently vegetated areas rather than in agricultural areas, and the additional infiltration also acts to dilute chemical concentrations. Entry et al. (2000) determined that the effectiveness of vegetative buffer strips can be greatly reduced in the winter months when vegetation is dormant.

Storage Lagoons

Summary

Storage lagoons are an important part of manure management. Researchers advocate the use of storage facilities during the non-growing season.

Numerous studies have documented leakage from manure lagoons and some have documented impacts to groundwater from nitrate, ammonium, veterinarian pharmaceuticals, chloride, TDS, and bacteria.

Adequate storage lagoon design includes consideration of the following elements: soils, location, liner permeability, liner material, environmental conditions such as minimum vertical separation and seasonal high water table.

Storage lagoons are an important management component for liquid and slurry wastes generated by CAFOs. Lagoons provide storage during the non-growing season and during times when land-applying wastes is not protective of water quality. During times when a crop is not actively growing, or when the growth rate is very slow due to low air and soil temperatures, continued application will most likely exceed crop needs. Since manure is typically generated year-round, a storage lagoon is often used to temporarily store manure during those times when crops do not require supplemental nutrients. Manure contains elevated concentrations of total dissolved solids, BOD, total nitrogen, phosphorus, chloride, and microbiological pathogens.

Hermanson et al. (2000) state that the use of storage facilities to minimize nitrogen applications during the non-growing season is a safe alternative to protect groundwater instead of year-round application.

The literature indicates that the important considerations during AFO lagoon design and site selection for manure storage and reducing the potential groundwater impacts include:

- *Lagoon design and construction*
- *Suitable soils*
- *Liner permeability*
- *Seasonal high water table*
- *Minimum vertical separation*

These elements are described in greater detail below.

Lagoon Design

There is general agreement in the literature that lagoon design and construction is an important component to minimize the potential for adverse impacts to groundwater from lagoon seepage. The NRCS Agricultural Waste Management Field Handbook (NRCS, 2009b) is the industry standard and describes the protocols and specifications for storage lagoons. Table 17 classifies the soils based on their composition and permeability. Table 18 describes the soils that are suitable for constructing lagoons.

Table 17. Soil classification based on composition and permeability ranges.

USDA Group Definitions	Percent fines	Plasticity Index (PI)	Estimated Range of Permeability (cm/sec)	
			low	high
I	<20	< 5	3×10^{-3}	2
II	>= 20	<= 15	5×10^{-6}	5×10^{-4}
	<20	>= 5		
III	>= 20	16 to 30	5×10^{-8}	1×10^{-6}
IV	>= 20	> 30	1×10^{-9}	1×10^{-7}

(NRCS, 2009b; USDA, 1993)

Table 18. Soil permeability, classification, and groups suitable for lagoon construction.

Unified Soil Classification System				Permeability (K) (cm/sec) ²	USDA Group ¹
Coarse-Grained Soils	Clean Gravels	GW	Well graded gravels, gravel-sand mixtures, or sand-gravel-cobble mixtures	$> 10^{-2}$	I
		GP	Poorly graded gravels, gravel-sand mixtures, or sand-gravel-cobble mixtures	$> 10^{-2}$	I
	Gravels with Fines	GM	Silty gravels, gravel-sand-silt mixtures	10^{-3} to 10^{-4}	II
		GC	Clayey gravels, gravel-sand-clay mixtures	10^{-6} to 10^{-8}	III
	Clean Sands	SW	Well graded sands, gravelly sands	$> 10^{-3}$	I
		SP	Poorly graded sands, gravelly sands	$> 10^{-3}$	I
	Sands with Fines	SM	Silty sands, sand-silt mixtures	10^{-3} to 10^{-6}	II
		SC	Clayey sands, sand-clay mixtures	10^{-6} to 10^{-8}	III
Fine-Grained Soils	Silts	ML	Inorganic silts, clayey silts of low to medium plasticity	10^{-3} to 10^{-6}	II
		MH	Inorganic silts, micaceous or diatomaceous silty soils, elastic silts	10^{-4} to 10^{-6}	III
	Clays	CL	Inorganic clays of low to medium plasticity, gravelly, sandy, and silty clays	10^{-6} to 10^{-8}	III
		CH	Inorganic clays of high plasticity, fat clays, sandy clays of high plasticity	10^{-6} to 10^{-8}	IV
	Organic Silts and Clays	OL	Organic silts and clays of low to medium plasticity, sandy organic silts and clays	10^{-4} to 10^{-6}	N/A
		OH	Organic silts and clays of high plasticity, sandy organic silts and clays	10^{-6} to 10^{-8}	N/A
Organic Soils		PT	Peat	N/A	N/A

Shading – acceptable to reduce lagoon seepage.

N/A – not appropriate for use as foundation or base material.

¹ NRCS, 2009b; NRCS, 2009a; USDA, 1993.

² Freeze and Cherry, 1979.

The NRCS Handbook (2009b) describes conditions where an additional measure of safety from lagoon seepage is necessary and when a constructed liner is warranted. (Constructed liners include compacted clay, concrete, and synthetic materials.) These circumstances include:

- Locations near a well, spring, or other vulnerable water source.
- Areas where there are less than two feet of soil between the land surface and the seasonal high groundwater or bedrock.
- Group I soils.
- Some Group II soils. These include Group III flocculated clays, and Group IV highly plastic clays with blocky structure.
- Other soil properties that affect permeability include: dry density, structure, chemical composition, high calcium and magnesium content, alluvial soils, and other deposition forces that create anisotropic conditions.

Other measures advocated by the NRCS (2009b) to minimize lagoon liner seepage, include:

- Clay liner of a maximum permeability of 1×10^{-7} cm/sec.
- Flexible membrane liner.
- Geosynthetic clay liner (GCL).

Nicholson et al. (2002) examined environmental effects and developed a rating system for different storage methods for slurry and solid manure. Based on the rating system these researchers developed, they determined that for slurry storage, the most protective to least protective systems are: (1) cylindrical above-ground tanks, (2) weeping wall storage, (3) below-ground concrete tanks, (4) lined lagoons, and (5) unlined lagoons. For solid manure storage, the most protective to the least protective are: (1) roofed storage with a concrete base, (2) a concrete pad with a tank, (3) a concrete pad without a tank, (4) field heap with a different site each year, and (5) field heap with the same site each year.

Suitable Soils for Lagoons

Natural soils in permeability groups III and IV (Tables 17 and 18) are usually acceptable to reduce lagoon seepage. These soils need at least 15% clay content. Clean sands and gravels always require a liner. (NRCS, 2009b; USDA, 1993)

Organic soils (OL, OH, PT) have a high organic content, which makes them unsuitable for use as a base foundation. The organic matter decays over time, which alters the soil structure; this ultimately can cause instability. As decay occurs, the pore space increases, making the soil susceptible to subsidence associated with hydrocompaction. (NRCS, 2009b; USDA, 1993)

Liner Permeability

The NRCS (2009b) recommends an allowable seepage quantity of 1×10^{-7} cm/sec (based on the historical permeability for clay liners). Ecology (Kimsey, 2002 [appendix C]) specify that agricultural wastewater lagoons have a final maximum liner permeability of 1×10^{-7} cm/sec or less. This permeability can be achieved through the use of adequate soils, liner design, and

sealing from manure. Soils with a permeability of 1×10^{-5} cm/sec are suitable for lagoon construction since this provides a base for constructing a lagoon, which will ultimately achieve the maximum recommended liner permeability through compaction or amendments. The optimal lagoon design has a maximum ratio of 8:1 lagoon depth to liner thickness, with a minimum liner thickness of one foot. This flexibility allows for utilizing site-specific conditions to achieve the best design. (NRCS, 2009b) This approach provides an optimal combination of liner thickness and permeability to achieve an effective and economical liner design.

Sealing of the soils is the result of physical, chemical, and biological processes. Suspended fines settle and clog soil pores, anaerobic bacteria produce byproducts that accumulate at the soil/water interface and reinforce the seal. Salts can cause dispersion of some of the aggregates that act to clog the soil voids and enhance the sealing mechanism. Additionally, the soil structure can be enhanced by the metabolism of organic material. (NRCS, 2009b)

The degree of sealing depends upon many site-specific variables. NRCS (2009b) cautions that even though manure sealing is well documented, it may not be adequate in all instances. Conservatively, the addition of manure to a lagoon can reliably contribute one order of magnitude reduction in seepage. (NRCS, 2009b)

Ham (2002a) noted that there are similarities between all the animal manure lagoons. The physical, chemical, and biological processes of the manure act similarly to create an additional restrictive layer that decreases the hydraulic conductivity of the lagoon to a mean of 1.8×10^{-7} cm/sec. The variations in discharge (seepage) rates were found to be relatively small, despite the large differences in soil types, manure composition, and depth to water found at the 20 sites studied.

Ecology modeled the impacts of various lagoon designs, estimated the impacts to groundwater quality, and determined the appropriate design, based on an acceptable level of degradation. The recommended maximum liner permeability was determined to be 1×10^{-6} cm/sec, with the assumption that manure sealing will provide approximately an order of magnitude of additional protection resulting ultimately in a permeability of 1×10^{-7} cm/sec. (Kimsey, 2002). This paper is found in Appendix C.

MPCA (2001) monitored an earthen manure storage unit that was upgraded with a geosynthetic liner with a bentonite clay liner. The liner was then covered with a foot of native soil. Additionally, a filter strip was installed downgradient of the pens and corrals. The results indicate that within three years of these improvements, total nitrogen concentrations in groundwater decreased by 55%; phosphorus and organic carbon decreased as well.

Leakage of contaminants below the lagoon liner is a function of many factors including the seepage rate, constituent concentrations, and the soil mineralogy underlying the lagoon. Researchers advocate using caution when interpreting soil results to compare one site to another. A thorough understanding of individual lagoon design, construction, concentrations, and management strategies are important when evaluating monitoring results.

Seepage Rates

Glanville et al. (2001) used a whole basin water balance approach to calculate leakage rates from 28 earthen manure storage structures and lagoons in Iowa. They determined that 53% of the storage structures had leakage rates close to the 1.6 mm/d regulatory rate in Iowa, 4% had significantly greater leakage, and 43% had significantly less leakage.

Ham and DeSutter (1999) investigated seepage losses from three animal waste lagoons in Kansas using a water balance method. These researchers found that seepage can be decreased to less than 1.6 mm/day by increasing the thickness of the soil liners from 0.3 to 0.46 m (assuming adequate soils and construction methods are used).

Reddi et al. (2005) noted that there is a relationship between seepage rates and liner permeability. They recommend that permeability should be between 1×10^{-6} and 1×10^{-7} cm/sec in order to achieve the Kansas seepage rate goal of 0.25 in/day (0.64 cm/day). These researchers also noted that liner thickness is an important consideration in retaining liquids in the lagoon, and they recommended a minimum liner thickness of 0.5 m (1.6 ft). They found liner thickness was more important than liquid depth. Based on modeling, they found that when liner thickness was increased from 0.15 m to 0.9 m, chloride breakthrough from the lagoon reaching groundwater occurred up to 60 years later.

Studies investigating lagoon design and the seepage of contaminants are summarized in Table 19.

Table 19. Summary of lagoon design and discharge characteristics.

Lagoon Seepage (Leakage) (mm/d)			Hydraulic Conductivity (cm/s)			Nitrogen Loading	Mean Liquid Depth in Lagoon	Liner Thickness	Reference
min	max	mean	min	max	mean	lbs N/ac/yr			
0.2	2.4	1.1			1.8×10^{-7}	385 kg/ha/yr		0.3 m (assumed)	Ham, 2002b
0.3	1.6	Range of 75% lagoons studied							Ham, 2002b
		1.3					3.5 m		DeSutter and Pierzynski, 2000
		1.2					1.8 m		Glanville et al., 2001
		2.7 ft/yr				270 lbs/ac/yr			Van der Shans et al., 2009
0.8	1.1	1.0	1.5×10^{-7}	7.8×10^{-8}		2187 - 2726 kg/ha/yr	5.6 m	0.3 - 0.46 m	Ham and DeSutter, 1999
0.6	1.0						1.66 m		Ham, 2002a
						Recommended		0.5 m	Reddi et al., 2005

Glanville et al. (2001) investigated 15 slurry pits and 12 lagoons. These researchers determined that there was no statistically significant difference in leakage rates between these two types of storage facilities.

Ham (2002a) tested and validated his theory that a precise water balance can be used to estimate seepage rates (± 0.25 mm/day) from animal waste lagoons and earthen structures. This method can be completed with five consecutive days of good weather by collecting detailed measurements of depth changes, and cumulative evaporation. This researcher advocates this method to quantify whole lagoon seepage since it has been determined that design permeability data do not adequately predict actual seepage rates.

Sidewall Seepage

Ham (2002) studied the differences in discharge rates from the floor of the lagoon and the side-wall embankments. He found that the permeability of the lagoon floor decreased from the effects of manure sealing. He also noted that the side-walls were susceptible to impacts from climatic effects such as freezing and thawing, wetting and drying, erosion, and from the formation of macropores (from earthworms and weeds).

MPCA (2001) investigated seepage losses from the bottom of the lagoon, the sidewall, and the center and perimeter tile. Chloride was used as a conservative tracer of manure migration. These researchers concluded that there is greater seepage through the sidewalls than the bottoms of the basins.

Glanville et al. (2001) studied 27 lagoons and slurry pits in Iowa. They determined that significantly greater leakage occurs through the sidewalls than the floor of the lagoon. They attributed this to manure sealing of the floor and greater compaction during construction.

Seasonal High Water Table Considerations in Lagoon Design

Groundwater levels naturally fluctuate seasonally. Saturated conditions can affect the treatment capacity of the soils and this can create reducing conditions. The fate and transport of contaminants is dependent upon the ability of the soils to treat and attenuate contaminants, which is affected by the soil conditions. (EPA, 2002) Saturated soils can be present in both western and eastern Washington, depending on the local hydrologic conditions.

The seasonal high water table is traditionally determined through visual inspection by digging a hole during the wet season to observe the level of standing water. (NRCS, 2009b) Seasonal groundwater can also be determined from static water levels taken from wells.

Fletcher and Veneman (2012) advocate that during the drier months, the highest groundwater levels can also be estimated from examining the soil morphology, mottling, and discoloration. Soils without excess water during the year usually are aerated and exhibit a yellow-brown color. Soils with high water tables will exhibit gray coloration during some part of the year at the depth of the high water mark and below. Gray colors are typically associated with saturated and chemically reducing soil environments, while yellow-brown colors are typically related to aerobic and chemically oxidizing conditions. The more distinct the wet period is, the grayer the soil. Many soils exhibit both gray and yellowish-brown colors, reflecting the presence of an elevated water table in spring and a drier, more aerated condition during late spring and summer when the water table subsides. The Munsell color identification system is a standard tool for delineating soil colors. (Fletcher and Veneman, 2012)

Soil saturation occurs when all the soil pores are filled with water. During this process of saturation, there is a chemical reduction of iron in soils. Soils are primarily made up of gray silicate minerals with small amounts of iron oxides that are brown or red. When anaerobic conditions are created (as with the inundation of groundwater), the iron is reduced and becomes dissolved. This allows the underlying gray silicate materials to become more pronounced. A mixture of the gray and red coloration is called mottling and is indicative of alternating oxidizing and reducing conditions. This is an indicator of a seasonal high water table. (NRCS, 2009b)

Soil saturation and reducing conditions are often correlated; however, it is not a universal occurrence. It was determined that it takes approximately 29 days of saturation to create reducing conditions, or 8% of the year (Franzmeier and Jenkinson, 2004).

Thurston County, WA requires a winter water study to determine the seasonal high water level. This is defined as the highest water level for seven consecutive days, or the highest mottling in soil profile (Thurston County, 1999).

Minimum Vertical Separation

Vertical separation is defined as the distance between the bottom of the lagoon liner and the top of the water table, at its highest level during the season. Maintaining an aerobic unsaturated environment beneath the lagoon is important to the attenuation and destruction (inactivation) of bacteria and viruses. (EPA, 2002). The minimum vertical separation provides this treatment space. Soils generally function as attenuation zones by filtering the larger bacteria and adsorbing the smaller viruses onto the negatively charged particles in the soil where they are inactivated by soil microbes. (EPA, 2002) The literature indicates that manure lagoons have a potential to impact groundwater quality if the microbiological pathogens are not destroyed in the vadose zone. (EPA, 2002)

Vertical separation is also important in maintaining the integrity of the liner. Lagoon liners which are inundated by water from below are more prone to failure. (NRCS, 2009b)

The minimal acceptable vertical separation is dependent upon site-specific characteristics, including soil texture. In fine-grained soils, a minimum of 2 feet may be adequate; but in coarser-grained soils, 10 to 12 feet may be necessary to remove all pathogens. Groundwater mounding from excessive infiltration can become a concern when the mound reduces the vertical separation by artificially raising the water table, such that pathogen attenuation is no longer effective (EPA, 2002; Hall, 1990). Researchers agree that if there is insufficient vertical separation, the lagoon should be constructed above ground.

Hall (1990) conducted a literature review on vertical separation as a treatment component for on-site sewage systems. This summary recommended that 2 to 4 feet of vertical separation is needed between the bottom of the pipes and the top of the water table to adequately attenuate bacteria and other pathogens. The literature also advised that additional separation may be necessary to account for groundwater mounding.

Monitoring

Monitoring a land treatment system provides an understanding of how the system is operating and if management practices need to be adjusted to meet performance goals.

This section describes the monitoring elements found in the literature that may be used to determine nitrogen availability for crops, and fate and transport of nitrate in the environment. Published researchers agree that a well-planned and comprehensive monitoring program provides information on the effectiveness of operations and impacts to the environment. This information can be incorporated into a site-specific plan that can be a vehicle for a producer to make informed decisions about manure and farm management. An accurate assessment allows producers to optimize nutrient application while protecting groundwater quality.

Harter et al. (2014) present various monitoring approaches to assess impacts of animal feeding operations (AFOs) on the environment. The Netherlands have established an extensive monitoring program focusing on soils, shallow groundwater, and deep groundwater. The Dutch have a national monitoring network in a randomized fashion by soil type, aquifer type, and farm type. In New Mexico all dairy farms must establish groundwater monitoring networks. California originally required groundwater monitoring networks but is now putting more emphasis on source management. The California Water Resources Control Board has adopted a combination of source management monitoring in conjunction with regional groundwater monitoring for dairies in the Central Valley.

Researchers recommend that a comprehensive monitoring plan include:

- Mass balance calculations
- Crop monitoring
- Manure monitoring
- Soil monitoring
- Irrigation water monitoring
- Groundwater monitoring

A more detailed description of standard sampling methodology is included in Appendix D for manure, soil, and groundwater.

Mass Balance

Mass balance accounting is an application of conservation of mass by accounting for all the nitrogen entering and leaving a treatment system. The nitrogen transformations or unmeasurable components can be accounted for with this technique.

A mass balance is an accounting of inputs and outputs (Equation 1). The difference between inputs and outputs is an indicator of the relative environmental risk. Nutrient and hydraulic mass balances calculated for each field where manure is land-applied provides a characterization of nutrient loadings and an assessment of environmental impacts. This exercise assists in determining accurate application rates that supply the correct amount of nutrients to the crop at

the time when the crop needs them and in a way that manages the waste in an environmentally responsible way

Equation 1. Mass Balance Equation

$$\text{Net Nitrogen (Gains or Losses)} = \text{Nitrogen Inputs} - \text{Nitrogen Outputs}$$

Summary

Researchers advocate the use of mass balance calculations to plan and refine manure management practices depending upon changing conditions. An accurate mass balance takes all nitrogen inputs into account, including; irrigation water, commercial fertilizer, manure, wastewater, crop residue, precipitation, and any other nitrogen additions. Some researchers have noted problems if the mass balance is not calculated for the entire farm, with monthly balances calculated for each field.

Several researchers advise caution against using the mass balance to determine if there are impacts to groundwater quality. Impacts to groundwater have been documented in circumstances where the mass balance calculations indicate a balanced treatment system. One study observed a correlation with changes in the mass balance corresponding with shallow groundwater nitrate concentrations increasing and decreasing.

Researchers in California observed that nutrient imbalances were typically the result of increasing herd size without the proportional increase in land base. They concur with other studies that if the amount of nutrients generated on the farm exceeds the ability of the crops to utilize the nutrients, the nutrients will accumulate in the soil and result in an increased risk of nitrate loss to groundwater.

A mass balance may be used to characterize nutrient loading, hydraulic loading, and salt loading. Harter and Menke (2004) recommend that a whole farm nutrient mass balance be computed for each field with monthly inputs and outputs. An annual summary indicates excess nutrient availability and the risk of adverse impacts to groundwater quality. Averaging applications over the year is not recommended, since timing is a critical component to achieving appropriate application rates.

Jégo et al. (2008) recommend that all nitrogen inputs be taken into account, including irrigation water, commercial fertilizer, manure, wastewater, crop residue, precipitation, and any other nitrogen additions. Harter et al. (2012) advocate that the elements in Table 20 be considered in calculating the nitrogen additions and losses as part of a mass balance.

Table 20. Nitrogen inputs and outputs.

Nitrogen Inputs	Nitrogen Outputs
Manure Application	Crop Removal
Inorganic Fertilizer	Volatilization
Mineralization of Soil Organic Matter	Denitrification
Previous Season Legume Crops	Leaching to Groundwater
Irrigation Water	
Precipitation	

Researchers found that mass balance estimates for nitrogen based on data from dairy farms in California were consistent with nitrogen leaching rates from a field-calibrated groundwater flow transport model. This suggests that nutrient mass balances can be a good tool to indicate potential impacts to groundwater (Harter, 2002; van der Schans et al., 2009).

Carey and Harrison (2014) found that mass balance increases or decreases corresponded with increases and decreases in shallow groundwater nitrate concentrations at an intensively monitored field in Whatcom County. However, actual concentrations of nitrate in groundwater were significantly higher than indicated by the mass balance estimates. These researchers speculate that this problem may be affected by the nitrogen that accumulates in the soils over time and then gradually mineralizes.

Calculating mass balances considers inputs and outputs and is most accurate when computed on a monthly basis for each field and compiled annually. Calculating mass balances by averaging the loading rates over the entire site or over the entire year can result in localized impacts to groundwater quality. Good recordkeeping is essential to developing an accurate assessment of manure management (Cogger, 2000).

Harter and Menke (2004) state that mass balance calculations are the primary management tool for determining how effectively the land treatment system is utilizing nutrients and when excess nitrogen is being applied. Cogger (2000) concluded that there is no single, simple, economical way to correct nutrient imbalances. Nutrient imbalances are typically the result of an increasing herd size without a proportional increase in land base for manure application. If the amount of nutrients generated on the farm exceeds the ability of the crops to utilize the nutrients, the nutrients will accumulate in the soil or leach to groundwater.

Two large scale indicators were developed for Canadian agricultural lands to assess risk to the environment. Drury et al. (2007) created the concept of residual soil nitrogen (RSN), which is the amount of inorganic nitrogen that remains in the soil at the end of the growing season after the crops have been harvested. This residual is calculated as the difference between nitrogen inputs and outputs. The RSN was calculated on a provincial and national scale for five years to assess trends. DeJong et al. (2007) created the second large-scale assessment called the “indicator risk of water contamination by nitrate” (IROWC-N). This indicator links the RSN to climate and soil conditions to determine the risk of nitrate losses. These results were mapped to calculate spatial and temporal changes in nitrate losses.

Cogger (2000) recommends the following practices as part of the nutrient management plan (NMP), which will assist in developing an accurate mass balance:

- Testing nutrients in manure and soil.
- Determining crop nutrient yields.
- Measuring nutrient application rates for each field and crop.
- Determining appropriate timing of applications.
- Calibrating application equipment.
- Measuring applications to determine if nutrient goals were met.
- Using manure storage during periods when manure cannot be safely applied.
- Isolating stormwater runoff from manure.
- Using buffers to protect surface water.
- Keeping accurate records.

Harter and Menke (2004) caution against presuming that there is a “negligible threat to groundwater” even when the mass balance calculations indicate a balanced treatment system.

Hermanson et al. (2000) recommend that if the mass balance indicates an excess of nutrients have been applied to the field, or if the soil nitrate, or groundwater concentrations are elevated, the nutrient management plan needs to be revised to address how excess manure will be managed and how surface water and groundwater quality will be protected.

Crop Monitoring

Crops are an important part of the land treatment process. Removing crops during harvest is part of the manure treatment process and is an important component in the mass balance calculation. Field measurements of crop nitrogen removal provide a verification of the amount of nitrogen removed from each field as part of the mass balance calculation. Equation 2 calculates the nitrogen use efficiency (NUE) which is based on measurements of crop removal and the amount of nitrogen applied in manure (EPA, 2011). Equation 2 is an estimate since the accuracy depends on the validity of the measurements for all inputs and outputs.

$$\text{Nitrogen Use Efficiency (NUE)} = \frac{\text{Amount of nitrogen applied}}{\text{Amount of nitrogen removed in crop}}$$

Equation 3 calculates the amount of nitrogen potentially available for leaching to groundwater, which is the difference between the amount of nitrogen applied to a field and the amount removed in the crop (EPA, 2011). Equation 3 is also an estimate since it does not account for all inputs and outputs.

Equation 3. Nitrogen at Risk to Leach to Groundwater Equation

$$\text{Amount of nitrogen potentially available for leaching} = (\text{Amount of nitrogen applied}) - (\text{Amount of nitrogen removed in crop})$$

The higher the NUE, the less residual nitrogen that is left in the soils and the lower the risk for leaching to groundwater. Dzurella et al. (2012) estimate that it is unrealistic to achieve a NUE of greater than 80% due to uncontrolled and unpredictable nature of precipitation events, recharge, mineralization rates of soil organic nitrogen, the spatial variability of soils, and the need to leach salts from the root zone.

Actual field measurements verify how much of the applied nitrogen was removed from fields with the harvested crop. Yield estimates need to be current and accurate to enable producers to be successful in maximizing their NUE.

Table 21 contains the nutrient uptake values for common forage crops in Washington State where manure is commonly applied (NRCS, 2009c). These values are based on the percent of dry matter for crops and realistic yield goals for planning purposes when developing NMPs (Equation 4). The “general” category applies when the growth stage at harvest is unknown. Additional crops are contained in the USDA-NRCS PLANTS database (<http://plants.usda.gov/java/>). Actual removal from fields varies, making field measurements important to determine how much of the applied nitrogen is removed with the crop.

Equation 4. Nutrient Uptake Values (NRCS, 2009c)

$$\text{Dry Matter Yield (lbs/acre)} = \text{Harvest Yield (lbs/acre)} * \% \text{ Dry Matter}$$
$$\text{Nutrient Uptake} \left(\frac{\text{lbs}}{\text{acre}} \right) = \text{Dry Matter Yield} \left(\frac{\text{lbs}}{\text{acre}} \right) * \text{Nutrient \% of Dry Matter}$$

Default dry matter % when estimates are unknown: silage = 25%; hay = 88% (NRCS, 2009c).

Table 21. Nutrient uptake values for common Washington forage crops (NRCS, 2009c).

Grass Forage	Nutrient % of Dry matter		
(mixed species)	N	P	K
General	1.93	0.28	2.61
vegetative	2.68	0.39	2.84
early bloom	2.12	0.32	2.34
late bloom	1.54	0.3	2.19

Grass / Clover Forage	Nutrient % of Dry matter		
(mixed species)	N	P	K
General	2.17	0.27	2.3
vegetative	2.84	0.34	2.78
early bloom	2.56	0.31	2.52
late bloom	2.22	0.3	2.29

Grass / Alfalfa Forage	Nutrient % of Dry matter		
	N	P	K
General	2.27	0.28	2.44
vegetative	2.95	0.36	2.88
early bloom	2.61	0.31	2.54
late bloom	2.07	0.3	2.29

Grain Crops Forage	Nutrient % of Dry matter		
	N	P	K
Corn silage	1.23	0.26	1.16
Cereal grains	2.13	0.43	1.84

Alfalfa Forage	Nutrient % of Dry matter		
	N	P	K
General	2.61	0.27	2.26
vegetative	3.21	0.33	2.93
early bloom	3.11	0.3	2.74
late bloom	2.6	0.3	2.4

Clover Forage	Nutrient % of Dry matter		
(mixed species)	N	P	K
General	2.41	0.26	1.99
vegetative	3	0.3	2.71
early bloom	3	0.3	2.71
late bloom	2.9	0.3	2.4

Timothy, Forage	Nutrient % of Dry matter		
	N	P	K
General	1.63	0.28	1.86
vegetative	2.1	0.36	2.17
early bloom	1.59	0.27	1.77
late bloom	1.29	0.26	1.72

N = nitrogen
P = phosphorous
K = potassium

Manure Monitoring

The nutrient content of manure is highly variable and is dependent on many factors including: type of bedding used, storage conditions, manure age, manure handling, and animal diet (Bary et al., 2000). An accurate assessment of the manure nutrient content is critical to calculate an accurate mass balance.

Sullivan (2008) determined that manure nutrients are highly variable. Manure analysis prior to land application assists in establishing appropriate application rates.

Appropriate application rates are based on crop needs. This requires an accurate assessment of the existing plant-available nitrogen present in the soils. Manure nutrient samples provide the

nitrogen content which allows accurate application rates to be calculated. Since the nitrogen content in manure is variable, these researchers recommend sampling manure during application to each field, to quantify the actual total nitrogen application. (Bary et al., 2000)

If a facility processes other waste streams on site, such as offsite feedstock for an anaerobic digester, more frequent pre-application sampling of the digester effluent may be necessary to adequately characterize nutrients that are available prior to land application.

A more detailed description of recommended manure monitoring protocols is contained in Appendix D.

Soil Monitoring

Summary

The literature recommends that soil samples be collected two times a year:

(1) after the fall harvest to assess the effectiveness of previous manure management practices and (2) in the spring before the first nutrient application to determine how much nitrogen is needed. Sampling the top foot of soil is the standard practice for assessing manure management practices.

Numerous studies have documented the variability of soil nitrate with depth and time, indicating that soil nitrate values are only indicative of the conditions at the time and location of sampling. Researchers clarify the limitations with soil nitrate data, stating that it cannot be used to extrapolate soil nitrate conditions at other locations or depths, or to estimate nitrate concentrations in groundwater.

It is generally accepted in the literature that excess nitrate in soils poses a risk of leaching to groundwater. Due to the mobility of nitrate and the uncontrolled nature of precipitation, soil nitrate can be mobilized and migrate to groundwater. Soil nitrate sampling provides a snapshot of what is present in the soils at the time the soil sample was collected. It cannot provide information on what has already moved through the soils to groundwater, what has moved below the sampling depth, or how much organic nitrogen will be converted to nitrate throughout the year and will leach to groundwater. Soil nitrate can indicate when excessive nitrate is present in soil and therefore poses a risk to leach to groundwater, but it cannot provide assurance that groundwater has been protected.

Variability in Soil Nitrate Concentrations with Time

Nitrogen availability from soil can change rapidly. Lamb (2012) notes that soil nitrate concentrations taken from the same locations between early August and late October varied dramatically (by as much as 75 lbs/acre) with no observed consistent trends over time. This researcher concluded that the results from soil sampling are indicative only of the nitrate available at the time and location the sample was collected.

Sullivan and Cogger (2003) suggest that soil sampling and analysis be conducted twice annually: in the spring before nutrient application and in the fall soon after the last harvest but before winter precipitation. These researchers advocate monitoring at the top 1 foot of soil to assess manure management. Kowalenko et al. (2007) recommend that fall soil nitrate be measured concurrently with the crop harvest in the fall.

Intensive soil nitrate sampling at a manured grass field over 5 years in Whatcom County, WA, illustrates the substantial fluctuations in soil nitrate concentrations that can occur at the 1-foot depth. The difference between the minimum and maximum weekly soil nitrate samples collected between September and October each year fluctuated by 14 to 24 ppm (43 to 74 lbs/acre). These researchers also observed nitrate concentrations in the top 1 foot of soil varied 45 mg/kg (ppm) within 1 week in November following a heavy rainfall. (Carey and Harrison, 2014)

Environmental factors that influence the conversion of organic nitrogen and ammonium to nitrate are soil pH, precipitation, and temperature (Heckman, 2003). In general, soil nitrate accumulates in the spring as temperatures increase and organic nitrogen mineralizes. In the summer the soil nitrate decreases as crops utilize nutrients. Any residual soil nitrate remaining after the crop has been harvested is susceptible to leaching with rainfall and irrigation.

Staben et al., (2003) noted a high temporal and spatial variability with soil nitrate samples as nitrogen transforms and migrates downward in the soil profile. In order to characterize this variability, they recommend sampling for nitrogen (nitrate and ammonium) at least two times during both fall and spring, with roughly one week between sampling. These researchers also recommend that all plant nutrients can be sampled, including nitrogen, phosphorus, and potassium.

Variability in Soil Nitrate Concentrations with Depth

Soil nitrate concentrations also vary with depth. Gehl et al. (2006) estimate that in sandy soils a large amount of nitrogen accumulated at depths between 60 and 210 cm (2 to 7 ft), with the greatest accumulation between 120 and 150 cm (4 to 5 ft). These researchers state that elevated post-harvest soil nitrate concentrations are an indicator that excessive nitrogen was applied. These researchers conclude that nitrate leaching to groundwater occurs when recharge mobilizes soil nitrate present in the lower part of the soil profile, below the root zone.

Camberato et al. (2013) indicate that normal background soil concentrations in sandy mineral soils in Indiana within the upper 1 foot of soil range from 5 to 10 ppm nitrate and 4 to 8 ppm ammonium. These researchers also noted that nitrogen concentrations were slightly lower in the 1 to 2 foot depth of soil. When the goal is to assess nitrate leaching losses, these researchers recommend sampling at the 1 foot depth and the 2 foot depth for both nitrate and ammonium. Sampling soil nitrate deeper within the soil profile is recommended in areas of high precipitation where the depth to water is greater than 2 feet. These researchers state that the residual soil nitrogen will eventually become nitrate, regardless of the form of nitrogen that was applied to the soils. Due to the complex biological transformations of nitrogen in the soils and the variable influences, it is difficult to accurately estimate the amount of nitrate loss that may have occurred at any point in time. Soil sampling is a way to determine soil nitrate concentrations at a particular point in time at the depth and location where sampling occurred.

Sullivan and Cogger (2002) indicate that sampling the upper 1 foot of the soil zone to characterize nitrate is appropriate in areas where wintertime precipitation is low, such as eastern Washington (east of the Cascade Mountains). However, they recommend deeper soil sampling, to 2 feet, in western Washington where heavier precipitation or recharge moves nitrate deeper into the soil profile.

Sánchez-Pérez et al. (2003) measured soil nitrate concentrations for 1 year at a cultivated field at 5 depths. Figure 17 graphs the soil nitrate concentrations over the year and at various depths. This illustrates the temporal and spatial fluctuations with the vertical movement in the soil column during the growing season and the non-growing season. These researchers also measured cumulative soil nitrate at 5 depths (Figure 18). It is interesting to note the downward migration of the nitrate peak during the year. Based on these findings it is clear that nitrate is not preferentially located in the top foot of soil. Together, Figures 16 and 17 illustrate the variability of soil nitrate concentrations over time and with depth.

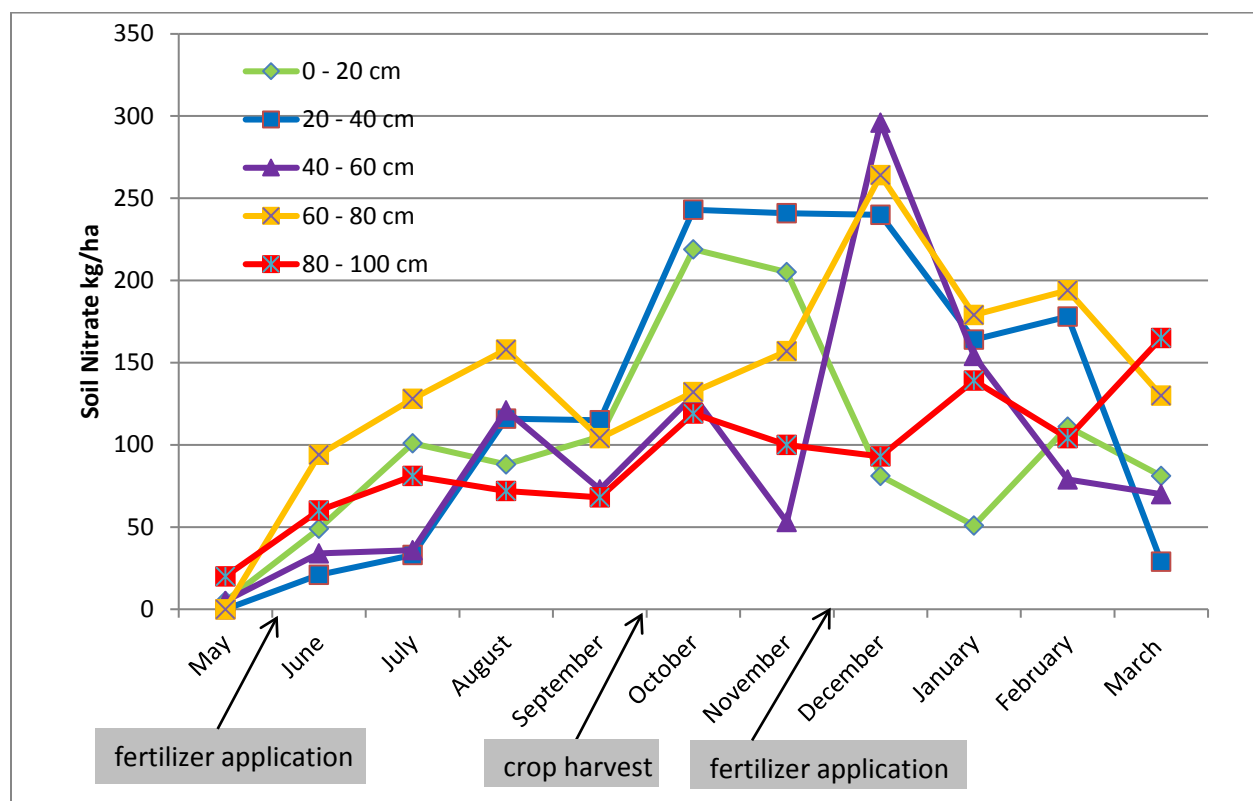


Figure 17. Soil nitrate concentration versus depth measured on a cultivated field over one year (modified from Sánchez-Pérez et al., 2003).

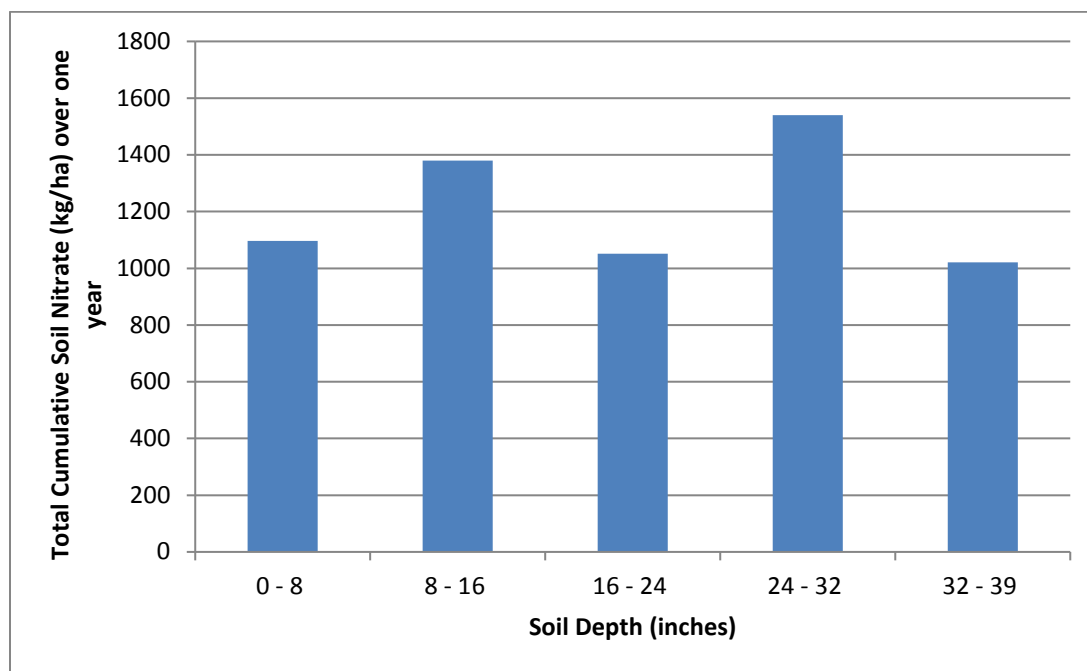


Figure 18. Total cumulative soil nitrate over one year by depth (modified from Sánchez-Pérez et al., 2003).

Recommended Soil Sampling Strategies

Soil nitrate samples provide a snapshot of the nitrate concentrations at the particular time and at the particular depth that the sample was collected. Soil nitrate concentrations vary as organic nitrogen is mineralized, as crops uptake ammonium and nitrate, and as excess recharge (precipitation and irrigation) mobilizes nitrate downward out of the sampling horizon.

Authors recommend that soil nitrate samples be collected during two times of the year: (1) in the spring before manure application begins and (2) in the fall after the crop has been harvested. These two sampling events have different goals and provide different information. The spring sampling is an indication of the current nitrate available to crops and is often used to generate nutrient budgets and application rates. Producers can use this information to adjust the amount of applied manure to more accurately meet the crop needs during the growing season. The fall soil nitrate sampling indicates the residual nitrate that the crop did not utilize and that is available for leaching during the winter non-growing months. The residual soil nitrate measured within the upper 1 foot is only part of the total nitrate pool that is subject to leaching over the year, because it does not take into account the residual nitrate in the lower soil profile or the fraction of nitrate which has already leached to groundwater. Accurate estimates of residual nitrate load is also a way for producers to adjust the future applications so that leaching to groundwater is minimized. (Kowalenko and Bittman, 2002).

Sampling the soil profile at 1 foot depth increments down to the water table provides the best estimate of the total residual soil nitrate, as well as the estimated nitrate load that poses a risk of leaching to groundwater.

Hart et al. (2009) further specify the optimal time for sampling fall soil nitrate based on soil texture and rainfall. To sample after the harvest and before the heavy rainfall begins, these researchers recommend that in medium- to fine-textured soils, the samples should be taken before 5 inches of cumulative rainfall occurs after September 1. In coarse-textured soils, samples should be taken before 3 inches of cumulative rainfall occurs after September 1.

Sullivan and Cogger (2003) found that it can take 3 to 5 years following improvements in field management for elevated soil nitrate concentrations to equilibrate with the new management practices. This highlights the importance of long-term management strategies for improving site conditions.

Soil monitoring protocols found in the literature are described in more detail in Appendix D.

Soil Monitoring Limitations

Soil samples can be useful in evaluating nitrogen crop use efficiency, residual soil nitrate in the fall, and the available nitrate for plant uptake in the spring. Standard soil tests are not accurate tools for quantifying the risk of nitrate leaching, especially when only the top one foot of soil is sampled. Few studies have tested the link between soil nitrate concentrations, leachate, and groundwater nitrate concentrations. Harter et al. (2012) found that the loading estimates calculated from soil nitrate concentrations underestimate the potential loading to groundwater estimated from an evaluation of nitrate leachate samples.

Soil nitrate values are useful in determining the amount of nitrate available to the crop at the time of sampling within the root zone. But because of the heterogeneity and complex biochemical reactions in the vadose zone and within individual fields, soil nitrate samples at the 1 foot depth were not found to be an accurate indicator of groundwater nitrate concentrations. (Carey and Harrison, 2014)

Low soil nitrate values may not identify all of the available nitrogen, especially if large amounts of organic nitrogen are present and mineralization occurs after sampling. The soil nitrate test provides only a limited prediction of the soil's nitrate-leaching potential (Flaten, 2001).

Gehl et al. (2006) investigated the post-harvest soil nitrate distribution in sandy soils under an irrigated corn field in Kansas. These researchers concluded that relatively low nitrate concentrations in the post-harvest soil samples do not necessarily indicate the lack of leaching to groundwater. Rather, the low soil nitrate concentrations could indicate that soil nitrate has already leached to groundwater before the growing season ended. In other words, a low soil nitrate concentration may or may not indicate impacts to groundwater quality. However, a high fall soil nitrate concentration indicates that leaching to groundwater is more likely to occur and there is an increased risk to groundwater.

Groundwater Monitoring

Summary

The majority of researchers agree that groundwater monitoring is the only way to definitively determine impacts to groundwater quality from residual soil nitrate. Monitoring other media, such as soils, can indicate whether manure management practices need to be adjusted, but it cannot conclusively determine the extent of the impacts to groundwater quality.

Researchers agree that soil nitrate will leach to groundwater during the winter with recharge (irrigation and precipitation). The extent of leaching is dependent upon the climate, soil type, mass of nitrogen present, and the hydraulic loading. Studies document nitrate leaching to groundwater under varying conditions.

Researchers have also been successful using other contaminants as indicators of impacts from manure applications. These include veterinarian pharmaceuticals, antibiotics, steroid hormones, calcium, chloride, magnesium, sodium, boron, bromide, and argon.

Groundwater monitoring provides a direct assessment of impacts to groundwater quality from land uses and is an important tool for determining how effective manure management practices are being implemented and thus minimizing impacts to groundwater. Groundwater monitoring is also an effective verification tool used to help evaluate the fate and transport of nitrate in the subsurface.

Leaching of Nitrate to Groundwater

Irrigation, manure application, and salt leaching practices can mobilize soil nitrate. Jégo et al. (2008) demonstrated that excessive irrigation could dramatically increase nitrate leaching to groundwater. Sawyer et al. (2006) noted the amount of nitrate leached to groundwater increases as the rate of applied nitrogen increases. Yin et al. (2007) concluded that groundwater in agricultural areas is especially vulnerable where salt leaching is practiced to maintain soil health.

Hermanson et al. (2000) conducted a literature review on nitrogen dynamics in the soil. The purpose of this review was to determine the fate and transport of nitrogen in subsurface soils for land treatment systems. The following is a list of the relevant general principles identified in this review:

- All nitrogen applied to the soil, and not utilized by crops, volatilized, or denitrified, will eventually convert to nitrate.
- Soil nitrate that moves below the root zone will eventually leach to groundwater.
- Nitrogen applied substantially before or after maximum crop demand may result in nitrate leaching.
- Organic nitrogen applied during the non-growing season will partially or totally convert to nitrate before the next growing season.

Research by Kowalenko and Bittman (2002) indicates that essentially complete nitrate loss occurs during the fall and winter, due to the distribution of heavy rainfall and fluctuating temperatures. They recommend minimizing residual soil nitrate at the end of the growing season to minimize losses to groundwater.

Studies conducted in British Columbia, Canada found that most (80%) of the nitrate and ammonium in the soils were leached to groundwater during the winter months (Zebarth et al., 1995; Kowolenko, 1987).

Chesnaux et al. (2007) applied a simulation model for the northern portion of the Abbotsford-Sumas aquifer with nitrate fertilizer applied at a rate of 80 lbs/acre. This model showed that nitrate from commercial fertilizer migrates through the vadose zone at a faster rate than manure. It also projected that nitrate applied at the surface of the model in mid-April would reach groundwater in August and be completely leached to groundwater by mid-November, with a travel time of seven months. These researchers also noted that depth to water table had little effect on the groundwater nitrate concentration, and the additional time to reach groundwater was not linear. Shallow groundwater concentrations were similar to the estimated concentrations modeled using simulations based on soil nitrate. They concluded that with the amount of recharge at this location, a loading rate of 41 lbs/acre of fertilizer would result in an exceedance of the groundwater standard of 10 mg N/L, assuming maximum crop uptake.

Carey and Harrison (2014) did not find a correlation between post-harvest soil nitrate concentrations and groundwater concentrations. However, they did note a correlation between fall manure applications and increases in shallow groundwater nitrate concentrations.

Groundwater monitoring protocols described in the literature are described in Appendix D.

Indicators as Evidence of Manure Impacts

EPA (2012 and 2013) recently conducted an investigation in the Lower Yakima Valley, to assess the source of nitrate contamination in groundwater. Multiple indicator parameters were used to assist in identifying sources. Analyzed parameters included nitrate, bacteria, nutrients, metals, ions, pesticides, trace organics, pharmaceuticals and personal care products (PPCPs), veterinary pharmaceuticals, hormones, and isotopes. This study found veterinarian pharmaceuticals and several inorganic constituents (calcium, chloride, magnesium, sodium, and boron) to be good indicators of manure sources.

Hudak (2002) recommends using chloride/bromide ratios to determine sources of nitrate in groundwater. This researcher compiled typical ratios for different contaminant sources. He concluded that there was no statistically significant difference between animal sources when using chloride/bromide ratios, but there was a statistically significant difference between animal waste, human waste, and synthetic fertilizer.

Batt et al. (2006) concluded that in Washington County, Idaho, CAFOs are the source of antibiotics in groundwater. This study found elevated concentrations of nitrate and ammonium, and detected two veterinarian pharmaceuticals, sulfamethazine and sulfadimethoxine, in six

groundwater wells. Since these pharmaceuticals were only approved for veterinarian uses, these researchers concluded the impacts originated from animal wastes.

Watanabe et al. (2008) investigated various waste streams from two dairy farms in California. These researchers found monensin in the flush lane water, in the lagoon and in groundwater wells adjacent to (and downgradient from) the lagoon. Groundwater concentrations were an order of magnitude lower than the lagoon concentrations. Monensin was not detected in groundwater at the land application area, but was in the flush lanes and lagoon, indicating that biodegradation and sorption are likely occurring in the soils. Since monensin is a veterinarian antibiotic, the researchers concluded that monensin transport into the shallow alluvial aquifer was linked to leakage from the dairy lagoon.

Reddi et al. (2005) advocate using chloride as a conservative tracer of lagoon leakage rather than ammonium or nitrate. These researchers explained that most of the nitrogen is in the organic or ammonium form. In these forms, it is readily attenuated in the subsurface and can be dependent on the cation exchange capacity of the soils. Chloride is highly mobile, moves with water, and is not attenuated, due to its negative charge. Chloride concentrations are elevated, typically greater than 1,000 mg/L.

McNab et al. (2007) found that argon can serve as a unique tracer for lagoon seepage distinguishing this discharge from fertilizer application. Other isotope tracers they evaluated were not effective as indicators for animal waste.

Kolodziej et al. (2004) investigated the presence of androgens, estrogens, and progestins at a dairy farm in California. These steroid hormones were present in the dairy waste lagoon, and were detected in surface water samples. There were no detections in groundwater. The absence of steroid hormones in groundwater suggests that biodegradation and sorption may be effective treatment mechanisms.

Tools Presented in the Literature to Evaluate Nutrient Impacts

Summary

This section presents examples found in the literature of tools that can be used to evaluate the fate and transport of nitrogen in the subsurface. The combination of calculating the mass load of nitrate in soil with using the groundwater Nitrate Loading Mass Balance Model can indicate potential impacts to groundwater. This can be used as a screening tool to indicate when groundwater monitoring is important. These tools are limited in that they cannot assure that groundwater is protected; they can only indicate when groundwater is at risk of contamination. These tools provide an estimate of impacts to groundwater quality.

The literature summarized in previous sections of this report indicates that it is difficult to accurately predict impacts to groundwater quality based solely on soil nitrate samples. There is a general consensus among groundwater scientists that the most reliable way to determine impacts to groundwater quality is through groundwater monitoring.

In the absence of acceptable groundwater monitoring data, soil nitrate values can be used as a tool to estimate potential impacts and to determine if groundwater monitoring is necessary. This section provides examples of broad scale tools for assessing nutrient impacts, such as estimating the potential nitrate leaching concentration from soil nitrate samples and using a nitrate loading mass balance model.

Soil nitrate is routinely monitored at manure land treatment sites. These data, combined with recharge and soil bulk density, can provide an estimate of the nitrate leaching potential. Tables 22 and 23 contain ranges of soil nitrate values under various conditions and project the concentration of nitrate in the soil water. These values were calculated using Equations 5, 6, and 7 (Kimsey, 1997). These are predictive nitrate concentrations. They are intended to be used as a screening tool for farm management but are not intended to be a precise predictor of groundwater impacts.

These equations require measured soil nitrate concentrations, soil bulk density, and local recharge rates. Soil nitrate values are converted into a mass loading rate by factoring in the weight of the soil based on the bulk density. The following equations estimate the mass in million pounds (Equation 5), the mass of soil nitrate (Equation 6), and the concentration of nitrate leaching to groundwater based on the soil nitrate concentration and recharge (Equation 7). This method yields a general estimate that only accounts for nitrate in the zone where the sample was collected. This limitation results in an underestimate of the total nitrate in the soil profile since nitrate in the lower depths is not included. These equations assume the entire soil nitrate mass migrates to groundwater. Additionally, these equations do not account for the nitrate fraction that may have leached to groundwater prior to sample collection. A more accurate estimate of residual nitrate would involve soil nitrate sampling at multiple depths to the water table and calculating the cumulative mass using Equation 5 for each depth.

Similar approaches to calculating impacts are used by many researchers: Harter and Menke, 2005; Meisinger et al., 2008; Barik, 2012; Feaga and Selker, 2004; Camberato et al., 2011; Elrashidi et al., 2005; Zebarth et al., 1995; van der Shans et al., 2009.

Equation 5. Soil Mass.

$$\text{Soil Mass (million lbs/acft)} = \left[\frac{\left(\text{Bulk Density} \frac{\text{gm}}{\text{cm}^3} \right) * \left(0.00220462 \frac{\text{lbs}}{\text{gm}} \right)}{\left(8.107 \times 10^{-10} \frac{\text{acft}}{\text{cm}^3} \right)} \right] * (1 \text{ E} + 6)$$

Equation 6. Mass of Soil Nitrate.

$$\text{Mass of Soil Nitrate} \left(\frac{\text{lbs}}{\text{acre}} \right) = (\text{soil nitrate (ppm)}) * (\text{soil mass (million lbs)})$$

Equation 7. Nitrate Available to Leach to Groundwater.

$$\begin{aligned} & \text{Nitrate Concentration (mg N/L) available to leach to groundwater} \\ & = \text{Mass of nitrate} \left(\frac{\text{lbs}}{\text{acre}} \right) \div \text{Recharge} \left(\frac{\text{acft}}{\text{year}} \right) \times (0.3677) a \times [0.9 b] \end{aligned}$$

- a) 0.3677 is a conversion factor that adjusts measurements of lbs/acre/year and acre/feet/year into mg N/L.
- b) 0.9 is a standard assumption that 10% of the nitrate is lost to denitrification. This factor can be altered if site-specific data indicates a different percentage is appropriate.

Equations 5, 6, and 7 are the basis for the values in Tables 22 and 23. These tables are designed to be a general guide to project impacts to groundwater and to establish protective targets for soil nitrate values.

The following conditions and assumptions are used in Equations 5, 6, and 7:

- This method is intended to be used as a screening tool and is not intended to be a precise predictor of impacts to groundwater quality.
- Soil nitrate samples are typically taken from the 1 foot depth. A more accurate estimate would account for the nitrate present lower in the soil profile.
- Sampling at additional depths provides more information on the extent of nitrate distribution throughout the soil horizon.
- The average weight of one acre-foot of soil in eastern Washington based on bulk density, is assumed to be 4.0 million pounds. The average weight of one acre-foot of soil in western Washington is assumed to be 3.2 million pounds. The average statewide value is 3.6 million pounds. (Sullivan and Cogger, 2002)
- Typically the percent of nitrogen lost to volatilization is 5% or less, and this occurs under specific conditions (van der Schans et al., 2009). Therefore, volatilization of ammonia is not considered in the calculation.
- Equation 7 assumes that there is negligible nitrate in the recharge water. If the nitrate concentration of the recharge water is significant, this can be factored into the total nitrate available to leach to groundwater.
- Recharge water includes the fraction of irrigation water plus precipitation which infiltrates into the ground.
- These equations assume that the soil nitrate mass is completely mixed with the volume of recharge water for the period of interest. Tables 22 and 23 provide nitrate leachate estimates based on assumed recharge rates of 0.5, 1, 2, and 3 feet. A more precise measurement of recharge could be substituted in Equation 6 to more accurately assess the potential leachate concentration.
- The conversion rate of 0.3677 in Equation 7 adjusts the measurements of lbs/acre/year, and acre/feet/year into mg N/L.
- Equation 7 assumes 10% loss of nitrogen from denitrification.
- These equations assume annual applications of nitrogen and water.

Table 22. Soil nitrate concentrations and projected concentrations available to leach to groundwater in eastern Washington.

Eastern Washington (with 10% denitrification)					
Soil Nitrate (ppm)	Soil Nitrate (lbs/acre)	Nitrate Leachate (mg N/L)			
		0.5 ft annual recharge	1 ft annual recharge	2 ft annual recharge	3 ft annual recharge
4	16	11	5	3	2
8	32	21	11	5	4
11	44	29	15	7	5
15	60	40	20	10	7
23	92	61	30	15	10
30	120	79	40	20	13
38	152	101	50	25	17
45	180	119	60	30	20
60	240	159	79	40	26

(Harter and Menke, 2005; Meisinger et al., 2008; Barik, 2012; Feaga and Selker, 2004; Camberato et al., 2011; Elrashidi et al., 2005).

Table 23. Soil nitrate concentrations and projected concentrations available to leach to groundwater in western Washington.

Western Washington (with 10% denitrification)					
Soil Nitrate (ppm)	Soil Nitrate (lbs/acre)	Nitrate Leachate (mg N/L)			
		0.5 ft annual recharge	1 ft annual recharge	2 ft annual recharge	3 ft annual recharge
4	13	8	4	2	1
8	26	17	8	4	3
11	35	23	12	6	4
15	48	32	16	8	5
23	74	49	24	12	8
30	96	64	32	16	11
38	122	80	40	20	13
45	144	95	48	24	16
60	192	127	64	32	21

(Harter and Menke, 2005; Meisinger et al., 2008; Barik, 2012; Feaga and Selker, 2004; Camberato et al., 2011; Elrashidi et al., 2005).

These tabled values (Tables 22 and 23) provide possible screening values to identify combinations of soil nitrate values and recharge rates that may migrate to groundwater. The concept is that the mass of soil nitrate is mixed with the amount of recharge to provide the resulting concentration. These tables also include a 10% loss to account for denitrification. For example, if a soil nitrate value collected in eastern Washington (Table 22) is 4 ppm in an area

with one foot of annual recharge from a combination of precipitation and irrigation, then the estimated nitrate concentration in the leachate migrating to groundwater would be 5 mg N/L.

The Washington State groundwater standard for nitrate is 10 mg N/L (Chapter 173-200 WAC). This is the same standard for the groundwater quality standards, the Washington State drinking water standards, and the federal maximum contaminant levels (MCLs). The groundwater quality standards apply to all activities that can impact groundwater quality and protect all waters in the saturated zone.

Recharge

Recharge water includes the annual fraction of irrigation water plus precipitation that is not used by crops and infiltrates into the ground. The method of using recharge and soil nitrate concentrations as a screening tool for possible impacts to groundwater quality is supported by researchers (Harter and Menke, 2004; Chesnaux et al., 2007). De Jong et al. (2005) advocate using recharge to estimate the nitrate concentration that will leach to groundwater as a result of the soil nitrate concentration. Zebarth et al. (1995) suggest that 1000 mm (39.37 in) of recharge water will dilute 100 kg N/ha (89.21 lbs N/acre) residual soil nitrate to 10 mg N/L in the leachate that migrates to groundwater. This is consistent with the values in tables 19 and 20.

Sánchez-Pérez et al. (2003) found a correlation between elevated soil nitrate concentrations and elevated groundwater concentrations. These researchers noted that this correlation was dependent upon the amount of recharge. The scale of nitrate leaching depends on the amount of recharge, nitrogen applied to the soils, and the residual nitrogen that exists in the soils. They state that in temperate and humid zones, precipitation mobilizes the soil nitrate and is the vehicle which carries the nitrate to groundwater.

Nitrate Loading Mass Balance Model

Groundwater nitrate concentrations can be predicted using a *Nitrate Loading Mass Balance Model*. (Kimsey, 1997; Frimpter et al., 1990). This model is currently used by Ecology's Water Quality Program (Ecology, 2008; Ecology, 2011). This model includes: (1) an assessment component that describes the potential impact to groundwater quality and (2) a mitigation component that establishes a soil nitrate concentration threshold limit, which helps in minimizing impacts to groundwater quality and with achieving compliance with established groundwater goals. Model variables are described in Table 24.

The combination of calculating the mass load of nitrate in soil (Equations 5 and 6) with the use of the groundwater Nitrate Loading Mass Balance Model, described below, can estimate potential impacts to groundwater. This can be used as a screening tool to indicate when groundwater monitoring is important. These tools are limited in that they cannot assure that groundwater is protected; they can only indicate when groundwater is at risk of contamination.

The *Nitrate Loading Mass Balance Model* (Kimsey, 1997) focuses on the following four factors: (1) the residual soil nitrate, (2) the amount of recharge, (3) the aquifer's ability to assimilate the contaminants, and (4) target soil nitrate values to help achieve compliance with water quality standards.

1. Equations 5, 6, and 7 (Tables 22 and 23) are used to estimate the mass of nitrate and the nitrate concentration that could migrate to groundwater through the soils, based on soil nitrate measurements and recharge rates. This value is N_{Leachate} .
2. The *Assessment Component* of the model calculates the impacts from the residual soil nitrate on groundwater quality at the downgradient property boundary. The assessment component is composed of three calculations: (1) the volume of recharge that is contributed over the property, (2) the groundwater discharge, and (3) the resulting estimated groundwater nitrate concentration (Equations 8, 9, and 10). Equation variables are defined in Table 23.

Equation 8. Volumetric rate of recharge (gpd) which falls over the manured land application area.

$$V_R = A_{\text{LAR}} (0.0017)$$

Equation 9. Volumetric rate of groundwater flowing into the upgradient end of the property (gpd).

$$Q = KibW_A (7.48)$$

Equation 10. Nitrate concentration in mg N/L in groundwater at the downgradient property boundary after the recharge mobilizes the soil nitrate and disperses with upgradient groundwater.

$$N_{\text{GW}} = [QN_B + V_R N_{\text{Leachate}}] / Q + V_R$$

The N_{Leachate} value is the groundwater leachate value calculated from Equations 5 and 6 (which can also be found in Tables 22 and 23).

3. The *Mitigation Component* of the model calculates the target soil nitrate concentration which would suggest that the AFO is in compliance with the Groundwater Quality Standards (Chapter 173-200 WAC). Since this is only a screening tool, this method cannot be used to assure compliance with the groundwater quality standards; it can only indicate a potential, estimated risk. The nitrate concentration (mg N/L) derived from the mitigation component is determined using Equations 11 and 12.

The information from these tools can assist in determining where groundwater monitoring is likely to be important.

Equation 11 projects the nitrate leachate concentration from soils that would theoretically be protective of groundwater quality under the assumptions described here. N_{GW} is established based on the groundwater quality enforcement limit or target goal.

$$N_{Leachate} = [N_{GW}(Q + V_R) - N_BQ] / V_R$$

This calculation determines the concentration of leachate from the residual soil nitrate. The value $N_{Leachate}$ equals the nitrate concentration (mg N/L) available to leach to groundwater. This value can be converted into soil nitrate by using Equation 12. This soil nitrate value is an approximate value that would be expected to be protective of groundwater given the site-specific conditions.

Equation 12. Soil nitrate concentration which is protective of groundwater quality.

$$\frac{Nitrate\ Leachate\left(\frac{mg}{L}\right)}{Recharge\ (acre - ft)(0.9\ denitrification\ rate)(1.342)} = Soil\ Nitrate\ ppm\ \left(\frac{mg}{kg}\right)$$

The soil nitrate concentration calculated in Equation 12 is the soil nitrate threshold value that is estimated to be protective of groundwater quality under the described assumptions.

The tools described in this section are offered as a way to estimate the relative impacts to groundwater quality and the risk when soil nitrate monitoring is the only assessment tool. These models can be used as a preliminary tool to assess whether it is important to conduct groundwater monitoring at a specific site. These tools are estimates and cannot provide the same level of assurance that groundwater monitoring can.

Table 24. Variables used in calculating nitrate impacts to groundwater.

Variable Description	Variable	Units
Volumetric rate of recharge over land treatment area (V_R)	V_R	gpd
Area where manure is land-applied (A_{LA})	A_{LA}	ft ²
Recharge rate (R) (sum of infiltrated precipitation and irrigation)	R	in/yr
Nitrate concentration in the leachate from land treatment area. This value can be found in Tables 19 and 20 ($N_{Leachate}$)	$N_{Leachate}$	mg N/L
Aquifer discharge (Q)	Q	gpd
Horizontal hydraulic conductivity of aquifer (K)	K	ft/day
Horizontal hydraulic gradient (i)	i	ft/ft
Thickness of vertical dispersion in aquifer (b)	b	ft
Width of property boundary (W_A) perpendicular to flow direction	W_A	ft
Downgradient groundwater nitrate concentration (N_{GW})	N_{GW}	mg N/L
Nitrate concentration of upgradient gw (N_B)	N_B	mg N/L

gpd = gallons per day

mg N/L = milligrams nitrate nitrogen per liter

ft = feet

in/yr = inches per year

Other Groundwater Models

Many groundwater models are available to assist in projecting impacts to groundwater quality. Some require intensive site-specific data, which typically generate more accurate results. NLEAP is a model developed by Shaffer et al. (1991) that stands for Nitrate Leaching and Economic Analysis Package. It is designed for use by farmers and the NRCS (formerly SCS) and other agricultural assistance agencies to estimate nitrate leaching potential under areas with agricultural crops. This model has a three-phased approach. An annual screening provides an initial estimate of nitrate leachings and more detailed monthly and event-based assessments.

GLEAMS (Groundwater Loading Effects of Agricultural Management Systems) is a model that was developed to evaluate the movement of agricultural chemicals, specifically pesticides, through the root zone (Leonard et al., 1987). This model augments CREAMS (Chemicals, Runoff, and Erosion from Agricultural Management Systems) which was the field scale model developed by Knisel (1980).

The U.S. Geological Survey (USGS) developed MODFLOW in 1984 as a standard three-dimensional, finite difference groundwater model which is suited to larger watershed assessments. It was originally developed in 1984, but since then has been modified from being solely a groundwater flow model to now include contaminant transport, unsaturated zone transport, and water use by vegetation, as well as other capabilities. MODFLOW is a data-intensive model that requires an accurate knowledge of environmental conditions.

Success Stories

This section contains a compilation of success stories that were discovered during this literature review. These are investigations that, in some cases, employed innovative solutions resulting in improvements to water quality. In several instances, there were also reports of improved crop yield.

Harter and Menke (2004) found that changes to nutrient management can result in significant improvements to water quality. These researchers specifically showed that there was a 70% improvement in groundwater quality beneath fields where manure is land-applied when the following changes were made: (1) eliminating the use of commercial fertilizer, (2) reducing manure applications, and (3) applying manure only during the crop growth stages. These researchers noted that these measures did not result in a considerable reduction in crop yield.

Rudolf (2015) evaluated agricultural best management practices (BMPs) in Woodstock, Ontario, Canada, where public drinking water supply wells had been contaminated with elevated nitrate concentrations. This community decided to purchase an additional 275 acres of land with the goal to keep the agricultural land in production but to change BMPs to reduce water quality to below (meet) drinking water standards. The application rate was reduced from 100 to 54 lbs/acre. The result was a 60% reduction in soil nitrate concentrations, from 20 to 8 ppm. Agricultural production increased slightly from 135 to 140 bu/acre of corn grown at this site. This researcher concluded that incentives to producers are influential for improvements in water quality. (Rudolf, 2015; Rudolf et al., 2015)

The European Union introduced a *Nitrate Directives* in 1991. This directive identifies nitrate from manure and fertilizer applications as the main cause of pollution from diffuse sources affecting European communities. This cooperative directive concluded that high numbers of concentrated livestock result in manure production that is out of balance with available land and crop requirements, which in turn creates a surplus of nutrients ultimately lost to water and air. The European Union imposed a number of management directives that are improving both groundwater and surface water quality. For example, nitrate vulnerable zones were designated where nitrogen applications were restricted in the amount and the times of application. The results from this directive have been difficult to quantify; however, the European Union (2010) found that from 2004 to 2007 there were stable or decreasing nitrate trends in 70% of surface water samples and stable or decreasing trends in 66% of the groundwater samples. Sutton et al. (2011) released “The European Nitrogen Assessment” as a scientific approach to address the nitrogen issues raised by the European Directive.

MPCA (2001) monitored an earthen manure storage unit that was upgraded with a geosynthetic liner with a bentonite clay liner. The liner was then covered with a foot of native soil. Additionally, a filter strip was installed downgradient of the pens and corrals. The results indicate that within three years of these improvements, total nitrogen concentrations in groundwater decreased by 55%; phosphorus and organic carbon decreased as well.

Carey and Harrison (2014) found that groundwater nitrate concentrations near the top of the water table were close to the groundwater standard of 10 mg N/L when manure application rates were similar to the crop removal rates and when manure was applied during the growing season. Feaga and Selker (2004) found that once manure application rates were reduced, soil nitrate concentrations also declined.

Sullivan and Cogger (2003) found that it can take 3 to 5 years following improvements in field management for elevated soil nitrate concentrations to equilibrate with the new management practices. This highlights the importance of long-term management strategies for improving site conditions.

The Council of Canadian Academies (2013) expert panel noted that in Canada where residual soil nitrate is elevated, shallow groundwater nitrate concentrations are also elevated. These experts stated that groundwater systems tend to respond more slowly to changes in land management between nitrogen application and impacts to groundwater. Timing is important to evaluate success with groundwater quality. The legacy effects from over-application of nitrogen may take years to fully impact groundwater. Therefore, these experts state that it is important to make positive changes in areas where groundwater has been impacted.

Summary of Conclusions Drawn from the Literature

This report summarizes the relevant literature on manure management practices and the potential impacts on water quality. There are some predominant themes and innovative approaches to treating, managing, monitoring, and assessing land application of manure.

Potential Impacts to Groundwater

There are documented examples of manure impacts on groundwater quality in Washington State. In the Lower Yakima Valley, an EPA (2012 and 2013) investigation concluded that dairy manure contributed to groundwater contamination of the local unconfined aquifer. The Sumas-Blaine aquifer in Whatcom County, Washington has documented long-term groundwater contamination. While there continues to be on-going research into the source of this contamination, a groundwater study at a dairy farm correlated manure applications with increases in groundwater nitrate concentrations. Research in both the Lower Yakima Valley and the Sumas-Blaine aquifer identify manure as the predominant source of nitrogen loading in these areas.

Additionally, numerous studies from other areas in the United States and elsewhere document manure impacts to groundwater quality.

Hermanson et al. (2000) emphasizes the uncertainties associated with nitrogen dynamics in the subsurface and concludes that applying wastewater to crops and soil systems during the non-growing season is not reliably protective of groundwater. These researchers conclude that applying organic wastes during the non-growing season has an inherent risk and requires close soil monitoring to avoid nitrate leaching. The use of winter cover crops helps to mitigate the problem but does not guarantee a solution. These researchers advocate, as a safe alternative, the use of storage facilities to minimize waste applications during the non-growing season.

Hermanson et al. (2000) provides general principles and recommendations based on their comprehensive review of literature on nitrogen land treatment systems. This review is summarized in the “Background” section of this report.

It is apparent from this literature review that land application of manure has a potential to adversely impact groundwater quality. These studies concluded that land application of manure creates a risk to groundwater when:

- Manure is applied at application rates greater than what is necessary to maintain a viable crop.
- Crops are not present to use the nitrogen present in the soils.
- Manure is applied outside of the growing season.
- Manure is stored in a lagoon not constructed to a recognized standard.
- Manure is applied on irrigated cropland where irrigation water is applied in excess of what crops can use.

Treatment

Land treatment of manure involves applying manure in the right amounts at the right time.

Application Rates

Several factors that affect application rates are identified in the literature. These include accounting for all sources of nitrogen, the type (form) of nutrients applied, time they are applied, the type of crop grown, the type of soils, and climate. Researchers agree that all sources of nitrogen need to be considered in the total load. Residual soil nitrate and continued mineralization are often overlooked sources.

Studies verify that there is a positive correlation between nitrogen application rates and crop uptake, and that there is also a positive correlation between nitrogen application rates and residual soil nitrate. Maximizing crop yield is not equivalent to maximizing crop uptake. Maximizing crop yield is not the goal of a land treatment system. Hermanson et al. (2000) cautions that maximizing crop yield will generally increase the risk of nitrate accumulation in the soil.

Nitrogen applied at the time and in the amounts needed by the crop will minimize the buildup of soil nitrogen (Hermanson et al., 2000). Soil nitrate accumulation poses a risk to nitrate leaching. Researchers have observed that changes to manure management and application rates can result in a decline in the soil nitrate concentrations, as well as result in significant improvements to groundwater quality.

Winter cover crops generally do not require the addition of nutrients to the soils.

If excess nitrogen is applied in one growing season, it must be offset by decreased nitrogen application the following season to avoid residual nitrogen buildup and subsequent nitrogen leaching (Hermanson et al., 2000).

There is general agreement in the literature that applying manure at rates greater than what is consumed by the crop has been demonstrated to cause elevated nitrate levels in groundwater. These researchers recommend that nitrogen applications be governed by the crop nitrogen requirements with the goal to treat the wastewater, minimize leaching, and minimize the impact to groundwater quality.

Soil Nitrogen

Soil nitrate values are a proven tool to determine plant-available nitrogen present in the soils as well as providing the effectiveness of manure management. Research summarized from the literature note that there are two times when soil nitrate samples typically are collected: in the spring before manure application begins, and in the fall after the crop has been harvested. These two sampling events have different purposes and provide different information. The fall soil nitrate test provides information on the effectiveness of manure management practices from the previous season. The spring soil nitrate test provides information about the amount of plant-

available nitrogen at the start of the growing season and provides information on appropriate application rates.

Soil nitrate concentrations are subject to a high level of temporal and spatial variability. Soil samples estimate nitrate concentration at the location and the time the sample was collected. During the non-growing season, this nitrate concentration represents the amount available to leach to groundwater. However, soil samples do not measure nitrate that has previously migrated to groundwater. Nitrate that has travelled below the soil sample location has either reached groundwater, or is in the soil profile below the sample. Therefore, soil samples alone cannot provide assurance that groundwater is not, or will not, be impacted.

Researchers agree that soil nitrate tests are not a surrogate for groundwater monitoring. However, the majority also agree that residual soil nitrate can indicate when excessive nitrogen has been land-applied and when groundwater may have been impacted from leaching.

The soil nitrate threshold limits recommended in the literature are summarized in Table 7. These values are limits that researchers from 14 publications have advocated: that there is either enough nitrogen available to support a crop or that no additional nitrogen should be applied. Recommended targets for fall soil nitrate values range from 5 to 24 ppm depending on the site-specific conditions. Recommended targets for spring soil nitrate values range from 16 to 30 ppm depending on the site-specific conditions.

The soil nitrate values recommended in the literature correlate well with the site-specific assessment tool developed in *Tools Presented in the Literature to Evaluate Nutrient Impacts* section.

Timing of Manure Application

In addition to application rates, the literature states that protecting groundwater is also tied to the timing of manure application, the opportunity for mineralization, and the opportunity for leaching.

Researchers agree that a high-risk time to apply nutrients is during the non-growing season (when plant uptake of nitrogen has ceased or has significantly slowed) or during periods when precipitation and irrigation exceed evapotranspiration. This time period poses an increased risk to surface water from runoff and an increased risk to groundwater from leaching of nitrate and other contaminants. The growing season is dependent upon a number of factors including the crop grown, temperature, and precipitation.

Researchers noted a correlation between excessive fall manure applications and increases in groundwater nitrate concentrations. Researchers advocate that nutrients be available when crops can use them, and they generally characterize this timeframe as the growing season.

Manure applied substantially before or after maximum crop demand may result in the buildup of inorganic soil nitrogen that will subsequently be susceptible to nitrate leaching. Applying organic wastes during the non-growing season has an inherent risk in terms of leaching nitrate to groundwater. (Hermanson et al., 2000)

Winter manure application has not been demonstrated in the literature to be protective of groundwater quality. Scientific literature is not evident to support the theory that nutrients can be stored in the soils during the winter or that manure land application during the non-growing season is protective of groundwater quality.

This report identifies three sources that use site-specific climate data that can be used to identify the growing season for specific crops and the appropriate timeframe to land-apply manure:

- Washington Irrigation Guide
- Wetlands Climate Guide
- T-Sum 200

Soil Mechanics

The soil horizon and the vadose zone are the places where nitrogen treatment and transformations occur. These processes include crop uptake and removal, volatilization, mineralization, denitrification, and leaching to groundwater. These processes are influenced by a variety of site-specific factors, and they also affect the fate and transport of chemicals in the environment.

Volatilization of nitrogen compounds once the manure has been incorporated into the soils is minimal with documented rates of approximately 5%.

Organic nitrogen from applied manure accumulates in soil and gradually mineralizes to ammonium. Researchers have recently concluded that mineralization continues during the winter months, although at a slower rate. Studies have shown that mineralization and nitrification can occur at significant rates in frozen soils especially in the presence of organic matter. In Canada, investigations have demonstrated that soil organic nitrogen and immobilized nitrogen contributed one-third to one-half of the nitrogen lost to leaching during the non-growing season.

Several researchers noted that tilling or disturbance to fields stimulates mineralization.

Generally, the reviewed studies concluded that mineralization is a significant source of nitrogen in fields where manure has been applied. Additionally it was concluded that, due to the continued mineralization during cold and freezing temperatures, applying nitrogen in the fall poses a risk of over-loading the soils with nitrogen.

Denitrification losses may reduce nitrate loading to groundwater. Denitrification rates were found by researchers to be dependent upon site-specific conditions, but were generally found to be low. Furthermore, some studies stated that it is of little importance in well-drained soils. In the reviewed literature, denitrification rates ranged from 5% to 16%. Some researchers observed that some degree of denitrification occurs at all sites, but that the high denitrification rates reported in the literature are not representative of shallow sandy aquifers. In the absence of site-specific data, denitrification may be assumed to be approximately 10%.

Soil Storage of Nitrogen

Nitrogen storage in the subsurface is the practice of applying manure during the non-growing season with the assumption that the nitrogen will remain in the root zone until the crop needs it in the spring. This practice has not been proven to protect groundwater. Mineralization continues during the winter months. Climatic conditions such as temperature and precipitation, and subsequently nitrogen transformations, are not elements that can be precisely controlled. These uncontrolled elements can promote nitrate leaching. Researchers have cautioned that attempting to store nutrients in the soils during the winter for use by crops in the spring poses a risk to groundwater.

Storage Lagoons

Storage lagoons are an important part of manure management. Researchers advocate the use of storage facilities to minimize nitrogen applications during the non-growing season as a safe alternative to protect groundwater instead of year-round application.

Studies have documented leakage from manure lagoons and some have documented impacts to groundwater from nitrate, ammonium, veterinarian pharmaceuticals, chloride, TDS, and bacteria. Adequate storage lagoon design includes consideration of the following elements; soils, location, liner permeability, liner material, and environmental conditions such as minimum vertical separation and seasonal high water table.

Storage lagoons are part of the manure management system and provide a place for manure generated during the non-growing season to be stored until the spring when the manure can be land-applied for crop use. Literature indicates that winter storage lagoons constructed to a permeability of less than 1×10^{-7} cm/sec (1×10^{-6} cm/sec before manure sealing) are consistent with current guidance and will reduce the likelihood of groundwater impacts.

Researchers agree that lagoon liners are not an impermeable barrier to the downward movement of contaminants. In general, they note that contaminant concentrations are greatest near the floor of the lagoon and decrease with depth.

Chloride is commonly used as a tracer to evaluate leakage from storage lagoons. Chloride moves readily with water and is not attenuated in the subsurface, making it a useful indicator of migration of contaminants from lagoons.

The use of storage facilities is a safe alternative to minimize waste applications during the non-growing season (Hermanson et al., 2000).

Monitoring

Monitoring provides an assessment of manure management practices.

Mass Balance

Conducting a mass balance is critical to good manure management. This involves quantifying all inputs and outputs every month for the entire farm and for each field. All nitrogen inputs are taken into account, including irrigation water, commercial fertilizer, manure, wastewater, mineralized organic nitrogen, crop residue, precipitation, and any other nitrogen additions. Researchers advocate the use of mass balance calculations to determine if excess nutrients are being generated that cannot be utilized by crops.

Several researchers caution against using the mass balance to determine if there are impacts to groundwater quality. Impacts have been documented in circumstances where the mass balance calculations indicate a balanced treatment system. One study observed that patterns with the mass balance increases or decreases corresponded with shallow groundwater nitrate concentrations.

Researchers in California observed that nutrient imbalances were typically the result of increasing herd size without a proportional increase in land base. They concur with other studies that if the amount of nutrients generated on the farm exceeds the ability of the crops to utilize the nutrients, the nutrients will accumulate in the soil and result in an increased risk of nitrate loss to groundwater.

Soil Monitoring

Studies have documented the variability of soil nitrate with depth and with time, indicating that soil nitrate values are only indicative of the conditions at that time and location. Researchers clarify the limitations of soil nitrate data, stating that soil nitrate results cannot be used to extrapolate conditions in other locations, at other depths, or in groundwater. Soil nitrate can indicate when excessive nitrate is present in the soils and poses a risk to leach to groundwater, but it cannot provide assurance that groundwater has been protected.

It is generally accepted in the literature that excess nitrate in soils poses a risk of leaching to groundwater. Due to the mobility of nitrate and the uncontrolled addition of precipitation, soil nitrate can be mobilized and migrate to groundwater. Soil nitrate sampling only provides a snapshot of what is present in the soils at the time the soil sample was collected. It cannot provide information on what has already moved through the soils to groundwater, what has moved below the sampling depth, or how much organic nitrogen will be converted to nitrate throughout the year and leach to groundwater. Soil samples cannot provide assurance that groundwater quality has been protected.

Groundwater Monitoring

The majority of researchers agree that groundwater monitoring is the only way to conclusively assess impacts of nutrient management practices on groundwater quality. Monitoring other media, such as soils, can indicate whether manure management practices need to be adjusted, but it cannot conclusively determine the extent of impacts to groundwater quality.

Researchers agree that soil nitrate will leach to groundwater during the winter with recharge (irrigation and precipitation). The extent of leaching is dependent upon the climate, soil type, the amount of nitrogen present, and the hydraulic loading. Studies document nitrate leaching to groundwater under varying conditions.

Researchers have also been successful in analyzing other contaminants as indicators of impacts from manure applications. These include veterinarian pharmaceuticals, antibiotics, steroid hormones, calcium, chloride, magnesium, sodium, boron, bromide, and argon.

Assessment Tools

The literature summarized in this report indicates that it is difficult to accurately predict impacts to groundwater quality based on soil nitrate samples. There is a general consensus among groundwater scientists that the best way to determine impacts to groundwater quality is to collect and analyze groundwater samples.

In the absence of acceptable monitoring wells, soil nitrate values can be used as a rough tool to estimate the potential for groundwater impacts and determine if groundwater monitoring is necessary to determine actual impacts. This can be achieved by using a combination of tools presented in the literature. Soil nitrate concentrations combined with area recharge rates can be used to estimate the amount of nitrate potentially available to leach to groundwater. The soil nitrate values listed in Tables 22 and 23 provide a reference for estimating when excess nitrate may be present in the soils.

The Nitrate Loading Mass Balance Model is a useful tool that can be used to assess potential impacts to groundwater quality and to estimate soil nitrate threshold limits which would likely be protective of groundwater quality.

Recommendations

A review of the scientific literature highlighted several manure treatment components and assessment strategies that promote the goal of water quality protection:

- Application rates
- Timing of manure applications
- Mass balance calculations
- Soil nitrate concentrations
- Groundwater monitoring Manure management

Application Rate Definition

A review of literature on manure management illuminated the inconsistent use of the term “*agronomic rate*”. There are numerous definitions for agronomic rate established for different purposes. Consistent definition and use of terms is a critical component to implementing consistent manure management practices. A standardized definition of “*application rate*” is needed to reflect the wastewater treatment goal and the need to protect groundwater quality. The definition for *manure application rates* is recommended: *the rate at which a viable crop can be maintained with minimal leaching of contaminants downwards below the root zone*. Land application of manure at rates that do not meet this definition pose a risk to groundwater quality.

Timing of Manure Application

A standardized tool is needed to establish growing season dates based on the crop grown and site-specific climatic conditions. This report describes three references that can be used to develop such a tool: the Washington Irrigation Guide, the WETS climate data, and T-Sum 200.

Manure should not be applied to land during the non-growing season, due to the high risk of groundwater contamination.

Plant development should be the key factor used to time manure applications rather than the calendar.

Monitoring and Assessment

Mass Balance

A mass balance calculation accounts for all nitrogen inputs and outputs in a land treatment system. Soil samples provide a way to measure the accuracy of the mass balance approach. Researchers advocate calculating mass balances on a monthly basis for each individual field and for the entire farm. They caution that averaging values over the entire site or over the entire year is not recommended and will not accurately reflect impacts to groundwater quality.

Soil Nitrate Concentrations

Soil nitrate concentrations can be used as an initial assessment of the effectiveness of manure management. If soil nitrate values indicate that groundwater quality is at risk, then the nutrient management plan should be revised to reduce nutrient levels in subsequent years.

Assessment tools should be adopted to determine the mass load of soil nitrate that is potentially available to leach to ground (Equations 5 and 6, Tables 22 and 23). The use of the Groundwater Nitrate Loading Mass Balance Model can provide site-specific soil threshold limits based on potential impacts to groundwater quality. These values can be used as threshold limits to identify when excessive nitrate is likely to be present in the subsurface and when groundwater quality is at risk.

Groundwater Monitoring

Animal feeding operations (AFOs) that apply manure to crops as part of their treatment system can adversely impact groundwater. Groundwater monitoring is the most reliable and direct means of measuring impacts to groundwater from manure applications. Soil samples provide limited information to evaluate nutrient management practices. They can indicate over-application of manure, current available nitrate concentrations in the soil, and effectiveness of management practices. However, soil nitrate is not a direct or reliable indicator of impacts to groundwater quality.

Models are tools that can assist in projecting future conditions that cannot be measured. They can simulate possible impacts from different management strategies to help provide direction for planning.

Manure Management

Effective manure management beneficially utilizes the available nutrients in a manner that protects water quality. The research summarized in this report documents that manure management that is protective of water quality has the following characteristics:

- Manure application will not degrade groundwater quality.
- Migration of contaminants below the root zone will be minimized.
- A whole farm nutrient mass balance is conducted as a part of manure management.
- Manure application rates consider the following:
 - Manner of application
 - Timing of application
 - Irrigation management
 - Crop uptake
 - Residual soil nitrogen
 - Appropriate land base for application

- Vegetative buffers and setbacks to surface waters are used as protective measures to mitigate pathogen impacts to surface water quality.
- Lagoons are used to store manure during the non-growing season as another treatment component. Storage lagoons are properly engineered, sufficiently sized, constructed, and maintained to contain manure during the non-growing season, or until it is possible to apply manure to crops during the growing season.

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Appendices

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Appendix A: The Nitrogen Cycle

Nitrogen is a dynamic element. It exists in many forms and undergoes many complex transformations in the environment. The aggregate of these transformations is known as the nitrogen cycle (Figure A-1). The nitrogen cycle is a series of biological processes which are influenced by climatic conditions, the physical and chemical properties of a soil, and management of the land.

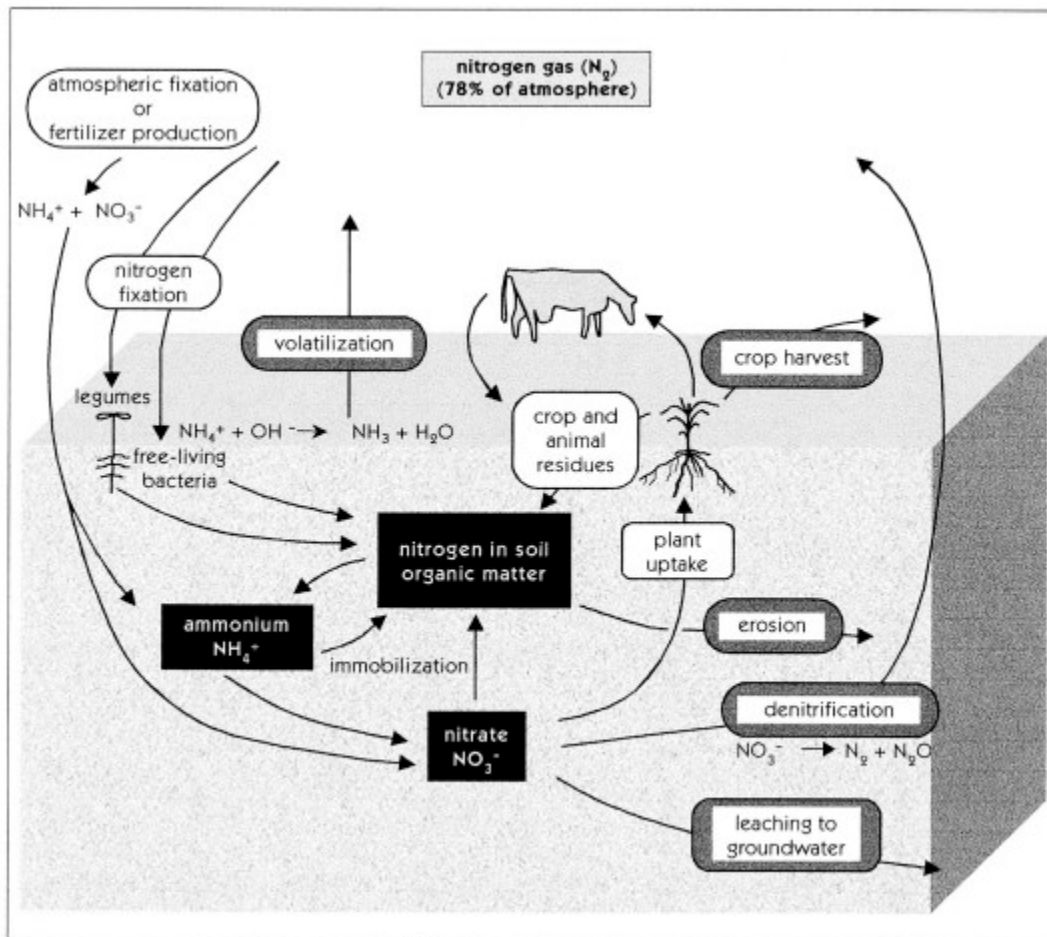


Figure A-1. Nitrogen Cycle (University of Western Australia, 2013).

Plants require nitrogen to grow. Animal manures and other organic wastes are sources of nitrogen for plant growth. The amount of nitrogen supplied by manure varies with the type of livestock, handling, rate applied, and method of application. The primary forms of nitrogen in manure are organic nitrogen and ammonium. Organic nitrogen must first be converted to an inorganic form (either ammonium or nitrate) before it can be taken up through roots and used by plants. When plants die, the organic matter becomes part of the soil. Then it is converted by bacteria, used by plants, and reverts back to organic matter. Table A-1 describes the different forms of nitrogen. Table A-2 describes the transformations that convert nitrogen into its different forms. (Killpack and Buchholz, 1993)

Table A-1. Nitrogen Forms.

Nitrogen Form	Chemical Formula	Description
Nitrogen gas Nitrous Oxide	N ₂ N ₂ O	The atmosphere contains 78% nitrogen gas. Nitrogen gas must be transformed into usable forms before it is available for plant uptake.
Organic Nitrogen	Various forms	Organic nitrogen is the dominant form of nitrogen in manure. Organic nitrogen originates in living material; it is present in animal and human wastes and decomposing plant material. Organic nitrogen is not usable by plants directly; it must first be converted to an inorganic form (ammonium, nitrate).
Ammonia	NH ₃	Ammonia can be present in either a liquid or gas state. Ammonia can escape from the surface of the soil under certain conditions. Anhydrous ammonia is the basic nitrogen form found in commercial fertilizers.
Ammonium	NH ₄ ⁺	Ammonium is an inorganic form of nitrogen and is available for plant uptake. Attenuation in soils occurs through cation exchange complexes.
Nitrite	NO ₂ ⁻	Nitrite is an intermediate product in the conversion of ammonium to nitrate (nitrification). It is usually present in low quantities but is toxic to plants.
Nitrate	NO ₃ ⁻	Nitrate is an inorganic form of nitrogen and is available for plant uptake. Nitrate is very soluble in water and highly mobile.

(Killpack and Buchholz, 1993)

Table A-2. Nitrogen Transformation Processes.

Nitrogen Process	Forms	Description
Nitrogen Fixation	$N_2 = > NH_4^+$	Nitrogen fixation is the process that allows plants to convert nitrogen gas from the atmosphere into a form usable for growth. Industrial fixation is the manmade process of creating fertilizers.
Mineralization (Ammonification)	Organic nitrogen = $> NH_4^+$	Mineralization is the conversion of organic nitrogen to ammonium. Bacteria are necessary in this process. Mineralization increases as microbial activity increases, which is directly related to soil temperature and water content.
Immobilization		Immobilization occurs when nitrate or ammonium present in the soil is used by bacteria to build proteins. These actively growing bacteria immobilize some soil N and break down soil organic matter to release N during the growing season. There is often a net gain of N during the growing season, because the additional N in the residue will be the net gain after immobilization-mineralization processes.
Nitrification	$NH_4^+ = > NO_2^- NO_2^- \Rightarrow NO_3^-$	Nitrification is the conversion of ammonium to nitrite, and nitrite to nitrate. Nitrification is a biological process which increases rapidly in warm, wet aerobic conditions. Nitrification slows when soil temperatures decrease below 50°F.
Denitrification	$NO_3^- = > N \text{ gas}$	Denitrification is the conversion of nitrate to atmospheric forms of nitrogen. Denitrification is a bacterial process and occurs in anaerobic zones typically created by saturated soils and the presence of organic matter. Denitrifying bacteria use nitrate instead of oxygen in their metabolic process.
Volatilization	$NH_3 = > N \text{ gas.}$	Volatilization is the loss of gaseous ammonia to the atmosphere. Volatilization can occur from manure and fertilizer products containing urea. Ammonia is an intermediate form of nitrogen during the process in which urea is transformed to ammonium.

(O'Leary et al., 2002; University of Western Australia, 2013)

Processes that Affect Nitrogen Fate and Transport

Table A-3 summarizes the physical, chemical, and biological processes that result in gains and losses of nitrogen, which occur as part of the nitrogen cycle. These processes directly affect the fate and transport of nitrogen in the environment.

When nitrogen inputs to the soil system exceed crop needs, there is a possibility that excessive amounts of nitrate may leach to groundwater or runoff to surface water. Minimizing impacts to groundwater quality can be achieved through sound management practices. Understanding the characteristics of nitrogen in the environment can help in efficiently managing nitrogen in land treatment systems.

Table A-3. Processes and conditions that affect nitrogen fate and transport

Nitrogen Process	Result	Description
Attenuation		The effect of all processes that reduce contaminant concentrations. Ammonium is a positively charged ion which allows it to be immobilized by binding to negatively charged soil and soil organic matter. Ammonium does not move downward in soils unless all the cation exchange sites are saturated.
Leaching	Loss to groundwater	Leaching is a physical process in which nitrate moves with soil water. Nitrate is a negatively charged ion and is not attenuated by negatively charged soil particles. Nitrate is water soluble and, once it migrates below the root zone, may leach to groundwater.
Run-off	Loss to surface water	Runoff to surface water occurs when fields are frozen or saturated and nitrogen cannot infiltrate into the soil pores. Water ponds and moves downhill towards drains, ditches, or surface water.
Consumption	Loss	Consumption of nitrogen by plants and other organisms occurs while nitrogen is retained in the root zone.
Decomposition	Loss	Any portion of a plant that is left after harvest, including roots and nodules, supplies N to the soil system. When the plant material decomposes, N is released.
Precipitation	Gain	Small amounts of N are added to the soil from precipitation.
Addition of Fertilizers or Manure	Gain of N to soil	Direct additions of manure, wastewater, or commercial fertilizers to crops.
Crop Removal	Loss	Crop removal during harvest accounts for the majority of the N that leaves the soil system.
Soil Organic Matter	Gain of nitrate. Loss of organic nitrogen	Decomposition of organic matter proceeds at a slow rate and releases approximately 20 lb N/acre/year for each percent of organic matter.

(O'Leary et al., 2002; University of Western Australia, 2013)

Appendix B: Regulatory Authority

Animal Feeding Operations (AFOs)

Ecology combines the National Pollutant Discharge Elimination System (NPDES) and State Waste Discharge (SWD) requirements in permits it issues for CAFOs. A permit issued to a CAFO that has a discharge may be an individual or general permit. Individual permits are a type of permit that is written for one specific facility. The permit requirements are unique and tailored to the activities taking place at that facility. General permits are a type of permit that covers a category of discharger. A category of discharger is usually a group of facilities that have similar discharge characteristics and utilize similar treatment mechanisms. Once a general permit is issued, facilities in that category can apply for coverage under the general permit. All facilities covered under a general permit have the same requirements. (Ecology, 2006)

There are legal classifications for defining AFOs and CAFOs, which are defined in 40 CFR 122.23 and are based on the type of animals, the number of animals, and how the animals are managed.

Large and medium size CAFOs are required to apply for a permit if they have (or had) a discharge to surface or groundwater of the state. For small CAFOs, and CAFOs with an animal type not listed in the federal CAFO rule, part of the process of designating the facility may be requiring the facility to apply for a permit. (Ecology, 2006)

The federal CAFO rule (current revision 2012) is the authority upon which Ecology bases its permits. It requires that CAFO permits be no-discharge permits. Discharges from CAFOs under a permit are prohibited, except in a narrow range of circumstances. CAFO permit requirements are designed to protect water quality through requiring the use of facility and manure management practices and record keeping requirements.

There is no requirement to monitor groundwater in the expired general permit, even when there are indications of groundwater impacts. Therefore, determining a discharge to groundwater (which is a water of the state) has been problematic. If a CAFO is not managing its manure properly, if manure is over-applied, if manure is applied at the wrong time, or if manure is stored in a lagoon not constructed to a recognized standard, then groundwater quality is likely impacted. Up to this point, CAFOs generally have not had to demonstrate compliance with groundwater quality standards. (Ecology, 2006)

In 1993, the Dairy Nutrient Management Act (Chapter 90.64 RCW) was passed by the state legislature. This legislation created a state Livestock Nutrient Management Program at Ecology, focused on regulating discharges from dairy operations. Part of the program was issuing permits to dairies that discharge. In 2003, the legislature moved this program from Ecology to the Washington State Department of Agriculture (WSDA). However, Ecology retained the authority to issue permits for facilities with discharges. This essentially split the program between Ecology and WSDA. The two state agencies coordinate resources: Ecology is responsible for administering, developing, and processing CAFO permits; WSDA inspects unpermitted dairies and permitted CAFOs and provides technical support to the operators of CAFOs. (WSDA, 2013; Ecology, 2013)

As a result of the Dairy Nutrient Management Program, the number of dairies and their locations is known. Because dairies are a known quantity and easily identifiable, they generally become the focal point of AFO related discussions. AFOs include CAFOs and the entire animal agricultural industry as a whole (e.g., beef, poultry, dairy, swine, horses), not just dairy farms. For example, based on USDA agricultural Census for 2007, there were approximately 243,000 milking cows in Washington State. At the same time, there were approximately 274,000 beef cows, 571,000 other cattle, 5.7 million laying hens, and 4.6 million broilers.

Washington State currently has 12 permitted CAFOs. One of these facilities is under an individual permit as a very large beef finishing operation. The remaining 11 facilities are 4 beef operations, 1 laying hens, and 6 dairies. These 11 facilities meet the requirements to continue coverage under the expired general permit and will most likely be rolled over into the next CAFO general permit that Ecology issues. (Jennings, 2013)

All dairies are required under Chapter 90.64 RCW to have a nutrient management plan. However, there is no legal requirement to implement the NMP on site or to collect or report annual soil test results. There are also no requirements for any environmental monitoring other than soil monitoring. These data are reviewed when WSDA inspects the dairy, which occurs approximately once every 2 years. Although a facility may be adhering to a site-specific NMP, it does not guarantee that a facility will be in compliance with water quality standards. Since there are no monitoring requirements for groundwater or surface water quality, there are no definitive means to verify that a dairy is complying with water quality laws and regulations. (Jennings, 2013)

Water Quality Regulatory Authority

Groundwater quality in Washington State is protected by the Water Pollution Control Act (Chapter 90.48 RCW), and the Groundwater Quality Standards (Chapter 173-200 WAC). These regulatory mandates are intended to protect all waters in the saturated zone, and apply to any activity which has a potential to pollute groundwater quality, including discharges from AFOs, waste storage facilities, and other agricultural activities.

The goal of the Groundwater Quality Standards is to maintain high quality groundwater and to protect existing and future beneficial uses through the reduction or elimination of contaminants discharged to the subsurface. This goal is achieved through treatment standards (AKART), the antidegradation policy, and the numeric criteria. The relevant criteria are listed below in Table B-1.

Table B-1. Groundwater criteria.

Groundwater Parameter	Groundwater Criteria (Standard)
Total Coliform Bacteria	1 colony/100 ml
Nitrate + Nitrite (as N)	10 mg N/L
Total Dissolved Solids (TDS)	500 mg/L

Ecology Guidance on Land Treatment Systems

Ecology regulates other land treatment systems by issuing permits, setting permit requirements, and monitoring water quality. Ecology developed a guidance document (*Guidance on Land Treatment of Nutrients in Wastewater, with Emphasis on Nitrogen*) to assist in managing these facilities. A land treatment system is intended to maximize nutrient uptake by the crop and minimize contaminant leaching below the root zone to protect the beneficial uses of the groundwater. Maximizing crop yield is not equivalent to maximizing crop uptake. Maximizing crop yield is not the goal of a land treatment system. Land treatment systems that have been approved and permitted by Ecology require that water and nutrients must not be applied in excess of the agronomic rate of the site's cover crop. (Ecology, 2004a)

Dairies

Dairies are one type of AFO that in some cases may be defined as a CAFO. Dairies produce milk and other dairy-related products. Milk is Washington State's second most valuable agricultural commodity, second only to apples. Washington State was ranked 10th nationally, producing 690 million gallons of milk in 2010. (WSDA, 2011)

In 2010, Washington housed 443 commercial dairy farms with approximately 250,000 mature cows. WSDA (2011) notes that the number of dairy farms has dropped over the years, yet the number of dairy cows have remained relatively constant.

Dairy farms are located in 28 of the 39 Washington counties. Table B-2 describes the dairy size classifications and Figure 1 illustrates the distribution of classes within the state. Whatcom County is home to the most dairies, with 125 farms housing a total 46,588 mature cows. Yakima County has the highest number of cows with 93,606 mature cows housed on 67 farms. Seventy percent of the large dairies are located in Yakima County and the Columbia Basin. The majority of small and medium dairies are located in western Washington. (WSDA, 2011)

Table B-2. Dairy size and distribution.

Dairy Size	Number of Cows	Percent of Total Dairies in Washington State
Small	1 – 199	40%
Medium	200 – 699	37%
Large	700 +	23%
	>2,500	4% (16 of the large dairies)

(WSDA, 2011)

Appendix C: Construction of Dairy Lagoons Below the Seasonal High Groundwater Table

Issue Paper

Construction of Dairy Lagoons Below the Seasonal High Groundwater Table

Washington State Department of Ecology
Water Quality Program, Southwest Regional Office
Melanie Kimsey, Hydrogeologist
January 18, 2002

Issue

There are a number of dairies in Washington State that are located above a seasonally high (near surface) groundwater table, that have requested NRCS (Natural Resources Conservation Service) design assistance for their waste management system. To obtain the required certified nutrient management plan, all dairies must meet NRCS specifications for waste storage ponds. The NRCS Waste Storage Facility Pond Criteria (313-Practice Standard) requires that all ponds have a bottom elevation that is a minimum of 2 feet above the seasonal high groundwater table. NRCS has indicated that building a soil embankment waste storage pond 2 feet above the seasonal high groundwater table, using imported soil material, is cost prohibitive for these dairies.

The NRCS is requesting assistance from Ecology in developing alternatives for these dairies that are also consistent with water quality standards. The primary alternative that NRCS is asking Ecology to consider is a liquid manure storage pond and if it can be designed and built below the seasonal high groundwater table and still protect groundwater quality.

The technical justification provided by NRCS in support of this proposal only considers the hydraulic issues and does not address impacts to water quality. If the unsaturated zone is diminished or eliminated, the soil treatment capacity for pathogens is also eliminated. Lagoon leakage studies previously conducted by Ecology identify groundwater contamination in areas where there are direct discharges to groundwater.

Dairy lagoons are an important part of the operation and management of dairy wastes. The lagoons provide storage during the non-growing season and during times when it is not possible to land-apply wastewater at agronomic rates. Dairy manure contains elevated concentrations of total dissolved solids, BOD, total nitrogen, phosphorus, chloride, and microbiological pathogens. This is detailed in Table C-1.

Dairy Lagoon Wastewater Characterization

Table C-1. Typical Dairy Lagoon Wastewater Characterization.

Parameter	Average Concentration	Range
Total Dissolved Solids	4,232 mg/l	2,890 – 6,850 mg/l
BOD	5,980 mg/l	1,300 – 14,600 mg/l
Total Nitrogen	456 mg N/L	275 – 600 mg N/L
Phosphorus	71 mg/l	26 – 133 mg/l
Chloride	221 mg/l	139 – 399 mg/l
Total Coliform	3,678,000 cfu/100 ml	230,000 – 7,400,000 cfu/100 ml
Fecal Coliform	1,755,000 cfu/100 ml	200,000 – 5,800,000 cfu/100 ml

Derived from Department of Ecology (Ecology, 1994a).

Total coliform bacteria has a groundwater quality criterion of 1 cfu/100 ml. Without any pretreatment, these contaminants will be directly discharged to groundwater and will cause a violation of the groundwater quality standards.

Nitrogen Assimilation

Conventional parameters such as nitrogen and total dissolved solids can be managed by reducing the specific discharge from the lagoon and calculating the amount of assimilation that will occur in the aquifer at the downgradient property boundary. This exercise was conducted in 1993 when Ecology modeled the impacts of various lagoon designs, estimated the impacts to groundwater quality, and determined the appropriate design based on an acceptable level of degradation. This exercise resulted in a recommended liner permeability of 1×10^{-6} cm/sec, with the assumption that manure sealing would provide approximately an order of magnitude of additional protection resulting ultimately in a permeability of 1×10^{-7} cm/sec. These modeling results were used to establish lagoon design standards that are protective of groundwater quality.

Minimum Vertical Separation

Vertical separation between the point of discharge and the top of the water table is one of the most important variables in preventing the transmission of disease from animal and human wastewater. For dairy lagoons, the primary pathogen treatment mechanism is an adequate vadose zone. The vadose zone allows microorganisms to be attenuated. Dairy lagoons can impact groundwater quality if the microbiological pathogens are not treated in the unsaturated zone. The minimum vertical separation provides the space for treatment to occur. Soils generally function as attenuation zones by filtering the larger bacteria and adsorbing the smaller viruses onto the negatively charged particles in the soil. Survival is prolonged under saturated and cool conditions. Inactivation is related to the activity of the native soil microorganisms and their antagonistic effects. Maintaining an aerobic, unsaturated environment beneath the lagoon is essential to the attenuation and inactivation of bacteria and viruses.

Microbiological pathogens, such as bacteria and viruses, can remain viable for extended periods of time in the subsurface depending upon the environmental conditions. Survival rates for these organisms depend upon the soil moisture, rainfall, temperature, pH, and the organic content of the soil (Bitton and Gerba, 1984). Scientific studies have noted pathogens surviving for extended periods of time and travelling considerable distances in the subsurface. Enteric bacteria can survive up to 100 days in favorable conditions and viral movement has been documented 1312 feet in sandy soils. (Canter and Knox, 1985; Keswick and Gerba, 1980)

The degree of vertical separation is dependent upon site-specific characteristics, including soil texture. In fine-grained soils, a minimum of 2 feet may be adequate; but in coarser-grained soils, 10 to 12 feet may be necessary to remove all pathogens. Groundwater mounding can become a concern when the mound reduces the vertical separation by artificially raising the water table, such that pathogen attenuation is no longer effective (Cogger, 1989).

Therefore, if microbiological pathogens are not removed or treated in the unsaturated zone and they are allowed to migrate to groundwater, they can be transported great distances, potentially contaminating groundwater and affecting drinking water wells, surface water bodies, and shellfish habitat.

Dairy Lagoon Studies

Ecology's Environmental Assessment Program has conducted several studies that investigate the impacts of dairy lagoons on groundwater quality. Two of these studies consider lagoons that are located below the seasonal high groundwater table on an intermittent basis. Both of these dairies are located in Whatcom County (Ecology, 1991; 1994a; 1994b).

These reports conclude that the vertical separation distance between the bottom of the lagoon and the top of the water table may account for elevated groundwater concentrations for many constituents in downgradient wells. Bacterial concentrations were only detected intermittently in groundwater. However, there is a direct correlation between total coliform and fecal coliform concentrations when the water table elevation was higher than the bottom of the lagoon.

Consistency with Washington State's Regulatory Philosophy

A lagoon constructed below the seasonal high groundwater table is a direct discharge to groundwater. The liquid contained in a dairy lagoon is untreated manure. Ecology has not permitted the direct discharge of untreated wastewater or treated wastewater into groundwater for other activities. Ecology has not permitted the direct injection of wastewater into groundwater without treatment and monitoring constraints.

- The UIC (Underground Injection Control) program prohibits the direct injection of any wastewater, including stormwater, into groundwater.
- The reclaimed water standards have very stringent requirements for direct groundwater recharge, including a high level of treatment and rigorous reliability and redundancy requirements.

- The Department of Health on-site sewage system regulations require a minimum of three feet of vertical separation between the bottom of the septic system drainfield and the top of the water table.
- Ecology's stormwater manual recommends that all facilities be located at least three feet above the seasonal high water mark, bedrock (or hardpan), and/or impermeable layer.

Ecology and the State of Washington have established a regulatory precedent which prohibits a direct discharge of untreated waste into groundwater.

NRCS Proposal

In areas where a seasonal high groundwater table exists, the NRCS is advocating a single membrane-lined lagoon with a soil cover as a cost-effective dairy lagoon design.

This proposed design below the seasonal high water table is not a viable option because this constitutes a direct discharge of untreated manure into groundwater, since all liners leak. This option removes the vadose zone, which acts as the only treatment for pathogens. Based on the elevated concentrations of pathogens in dairy manure, and the fact that pathogens can remain viable in the subsurface for extended periods of time and travel considerable distances once in groundwater, a design of this nature would cause a violation of the groundwater quality standards. This option is also inconsistent with other regulatory approaches to treating wastewater that is discharged to the subsurface.

Options

Due to the persistent and mobile nature of pathogens, dilution is not an acceptable form of treatment. It is not possible to calculate an acceptable volume of wastewater discharge that will be protective of groundwater quality if the proposed direct discharge occurs. It is imperative that the wastewater either be contained or treated prior to being discharged to groundwater. Two main options exist for designing dairy lagoons in areas with seasonally high groundwater tables and providing required protection of groundwater quality from microbiological pathogens.

Option #1

The lagoon shall have a bottom elevation that is a minimum of 2 feet above the seasonal high groundwater table. In areas where the seasonal high groundwater table is less than 2 feet, additional soil should be used to create an above-ground lagoon.

This option provides a treatment zone for the attenuation and inactivation of pathogens, which is necessary to achieve compliance with the groundwater quality standards (Chapter 173-200 WAC). This option is similar to the NRCS Practice Standard 313 Waste Storage Structures. The main difference is that this proposed option does not allow the use of perimeter drains to artificially lower the water table.

Option #2

Construct a non-discharging lagoon by designing a double membrane lined lagoon with a leak detection system. This option achieves containment of the dairy wastewater and creates a non-discharging lagoon. A properly designed double membrane lagoon with a leak detection system should achieve compliance with the groundwater quality standards. This design could be situated below the seasonal high groundwater table.

The recommended design of the proposed double membrane lined lagoon with a leak detection system, the design considerations, and construction quality assurance are described below (Garin Schrieve, Department of Ecology, written communication, January 2002).

Recommended Design

The recommended default approach for double-liner systems where the bottom will be constructed below the seasonal high groundwater table is the following:

Two flexible membrane liners of minimum 30-mil thickness (60-mil if HDPE) separated by a suitable granular or geosynthetic leak detection layer. The leak detection layer must be capable of transmitting any leakage through the primary liner to a collection point or sump without becoming saturated to its full depth. Leakage accumulating in the collection point or sump should be periodically measured and removed such that potential for leakage through the secondary liner is minimized.

This recommendation is consistent with the draft Surface Impoundment Standards in *Minimum Functional Standards for Solid Waste Management* (formerly Chapter 173-304 WAC now proposed as Chapter 173-350 WAC). These standards call for liners consisting of either a single membrane liner with groundwater monitoring or a double membrane liner equipped with a leak detection and collection layer.

Other types of double-liner systems may be suitable for this application. However, due to concerns with cyclic hydration/desiccation of the secondary liner with the fluctuation of the groundwater table, soil liners and geosynthetic clay liners (GCL) may not be suitable for secondary liners in these types of systems. The design engineer should consult the literature in determining the applicability of these materials for a particular impoundment.

Further Design Considerations

It is beyond the scope of this report to discuss all the design elements that should be considered for each waste containment impoundment. However, some areas of particular concern for these types of systems include:

- **Membranes.** The type of membrane selected must perform well under the service loadings and be amenable to standard welding and seam testing techniques. The membrane must be protected from protruding objects in the subgrade, leak detection layer, and cover material (if used). Particular attention should be paid to the potential for damage of the membranes during placement of overlying layers.
- **Leak Detection Layer.** The layer must have the hydraulic capacity under the design loadings to transmit primary liner leakage without becoming saturated. The design must consider

reductions in transmissivity due to creep, blockage by soil particulates, or biological fouling. If soil or GCL is used as a liner, appropriate measures must be included to prevent migration of soil particles into the detection layer.

- *Foundation soils.* Foundation soils must be capable of supporting the pond without deformation that would jeopardize the liner system. Soils must provide suitable bedding for the membrane—consult ASTM standards on installation of geomembrane for specifics on acceptable subgrade conditions.
- *Uplift.* Special attention should be paid to the potential for damage to the liner system by groundwater uplift forces under various fill depths.
- *Static and seismic stability.* The stability of embankments and geosynthetics should be evaluated. Particular attention should be paid to material interfaces and uplift forces by groundwater which may reduce interface friction. Due to saturated soil conditions, special attention should also be paid to the susceptibility to liquefaction of the foundation soils. Design review by Ecology's Dam Safety Section is triggered for impoundments capable of containing 3,259,000 gallons.

Construction Quality Assurance

The importance of construction quality control and quality assurance for waste containment facilities cannot be overemphasized. The only way to ensure that the facility will perform as designed is to prepare and implement a Construction Quality Assurance Plan which lays out the program of inspection and testing that will be conducted during construction. The EPA guidance document, *Quality Assurance and Quality Control for Waste Containment Facilities* (EPA/600/R-93/182) represents the state of the practice for construction quality assurance at waste containment facilities and should be followed.

Additional Concerns

- Ecology acknowledges the agreement made with the NRCS in 1994 regarding the minimum vertical separation requirement and is not proposing to increase this requirement at this time. However, current scientific knowledge points to the use of maintaining a minimum vertical separation of 3 feet or greater, depending upon the soil type. Incorporating these elements into lagoon siting and construction designs would provide consistency with the regulatory requirements for minimum vertical separation established by the Department of Health for On-Site Sewage Systems and by Ecology for stormwater infiltration systems.
- There is a discrepancy between the construction standards for dairy lagoons and those standards required for all other waste impoundments. In order to obtain equal protection for groundwater and to comply with Chapter 173-240 WAC, manure lagoons should be designed, constructed, and installed consistent with the requirements for other waste impoundments.

References (for Appendix C)

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Appendix D: Sampling Guidelines

Manure Sample Collection

Understanding manure composition is an important part of the monitoring process. The results of the manure analyses will be used to determine the amount of manure to apply based on the Nutrient Management Plan. This section contains procedures for sampling both liquid and solid manure.

Quality Assurance information, guidance, and project plan templates can be found at Ecology's website: <http://www.ecy.wa.gov/programs/eap/quality.html>.

A Quality Assurance Project Plan (QAPP) should be developed which describes the sampling procedures and goals. Guidance on developing this plan can be found at: <https://fortress.wa.gov/ecy/publications/SummaryPages/0403030.html>

A laboratory should be used that routinely analyzes manure samples. Manure nitrogen analyses can be reported in two ways:

- Reported “as-received” is used to determine application rates. This is described as ammonium and total nitrogen in lbs/ton.
- Reported “dry-weight basis” is used to track manure nitrogen composition that can indicate consistency over time and is described as nitrogen in mg/kg.

(Sullivan et al., 1997; Bary et al., 2000)

All laboratories used must be accredited and must supply quality assurance results with the data. Details on the volumes of manure samples needed and sample containers can be arranged with the laboratory. Table D-1 describes the manure monitoring parameters and analytical methods. (Bary et al., 2000; Peters et al., 2003; Momohara, 2012).

Table D-1. Manure monitoring parameters and analytical methods.

Analyte	Analytical Method	Units
pH	SW-846-9045D	Standard Units
Dry matter/Solids	SM 2540G	Percent
Electrical conductivity	SW-846-9050	µmhos/cm
Ammonium-Nitrogen	SM-4500-NH ₃ B+ C, D, E, or G	mg/kg
Total Kjeldahl Nitrogen	Combustion: Micro-Dumas Method ¹ Analysis: SM-4500-N _{org} B or C	mg/kg
Phosphorus	SM 4500-P B+E or F; Digestion EPA 200.2+Analysis 200.7 or SW-846-6010	mg/kg
Potassium	Digestion SW-846-3050; Analysis SW-846-6010, 846-6020, EPA 200.7, or 200.8	mg/kg
Chloride	SW-846-9056	mg/kg
Total Carbon	SW-846-9060	mg/kg

¹ Hauck (1982) and Wolfgang (1983) <http://sisbl.uga.edu/udumas.html>

SM: Standard Methods for the Examination of Water and Wastewater, 20th Edition.

EPA: EPA Methods for Chemical Analysis of Water and Wastes, EPA 600/4-79-020.

SW 846 --U.S. Environmental Protection Agency (U.S. EPA). Test Methods for Evaluation of Solid Waste, Physical/Chemical Methods. EPA Publication SW-846.

Liquid Manure

The following are recommended procedures for sampling liquid manure.

- Before manure application:
 - Sample from the storage facility after agitating the storage facility for a minimum of 2 to 4 hours (Peters et al., 2003).
 - Collect at least 5 sub-samples, using a clean 5-gallon bucket.
 - Combine the subsamples in a clean container.
 - Mix thoroughly.
 - Pour samples into laboratory-supplied sample bottles.
 - Send samples to the laboratory.
- While manure is being applied: Use either collection method 1 or 2 as described below.

Method 1. Sample manure from application equipment

- Collect at least 3 composite grab samples from a sampling port in the manure application equipment during the application to the field (i.e., after ¼ of the field has received manure, ½ of the field, and ¾ of the field).
- Grab samples should be composited in a clean 5-gallon bucket.
- Mix thoroughly with a clean ladle or other mixing device.

- o Send samples to the laboratory.

Method 2. Sample manure from field during application

- o Collect subsamples in clean 5-gallon buckets located within the field to catch manure from the spreader.
 - o Combine the subsamples and mix thoroughly.
 - o Pour into laboratory-supplied sample bottles.
 - o Send samples to the laboratory. (Sullivan et al., 1997)
- Freeze samples if mailing to the laboratory or refrigerate if delivering directly.
 - Ensure that samples reach the laboratory within 48 hours of collection and that the laboratory can analyze them within established holding times.

Solid Manure

The following are recommended procedures for sampling solid manure (Bary et al., 2000; Peters et al., 2003):

- Sampling during loading: This method is recommended for sampling from a stack or bedded pack.
 - o Take at least 10 to 20 samples while loading several spreader loads.
 - o Combine to form one composite sample.
 - o Thoroughly mix the composite sample (about 5 gallons).
 - o Take a 1-pound/1- quart subsample using a 1-gallon plastic bag or other container, as recommended by the laboratory.
 - o Sampling directly from a stack or bedded pack is not recommended.
- Sampling during spreading
 - o Spread a clean tarp in the field.
 - o Catch the manure from one pass of the applicator.
 - o Repeat this process at several locations.
 - o Create a composite sample.
 - o Thoroughly mix the composite sample.
 - o Take a 1-pound/1-quart sample with a 1-gallon plastic bag or other container recommended by the laboratory.
- Freeze samples if mailing to the laboratory or refrigerate if delivering directly.
- Ensure that samples reach the laboratory within 48 hours of collection and that the laboratory can analyze them within established holding times.

Field Measurements

Field meters, such as an Agros Meter, can provide a quick and reliable field test for ammonium-N to compare with laboratory results (Sullivan et al., 1997; Sullivan et al., 1994; British Columbia Ministry of Agriculture, 1994). The Agros Meter can be used with both liquid manure and diluted solid manure.

Soil Sample Collection

Soil samples should be collected in a manner that is representative of the entire field and the sampling depth. Soil probes or augers should be used to collect samples. A shovel should not be used since it will not collect a uniform sample with depth.

Samples locations should be planned in advance using field maps. These locations should be recorded with a GPS and sampled consistently each sampling event to assess variability over time. At each location, samples should be collected using a grid or a random zig-zag pattern, making sure to sample where manure is routinely applied, and areas which represent average crop growth. (Sullivan and Cogger, 2003; Staben et al., 2003)

Soil samples should be collected by scraping away any crop residue or manure present at the soil surface. Only the soil should be collected for analysis. If the manure has been injected into the soils, then care should be taken to avoid the injection sites, and a larger number of soil cores should be collected to promote representativeness. (Sullivan and Cogger, 2003)

Composite samples should be collected in a bucket and then mixed thoroughly before a subsample is taken. Samples should be placed in containers provided by the laboratory and then kept refrigerated or frozen. Samples should be shipped to the laboratory within 48 hours of collection.

Number of Samples

Soil composition varies spatially. Soil samples should be composite samples consisting of 15 to 30 soil cores from each field. The number of subsamples needed to characterize conditions at each depth depends on the size of the field as shown in Table D-2.

Table D-2. Number of subsamples recommended for a representative composite sample, based on field size (Mahler and Tindall, 1997).

Field Size (acres)	Number of Subsamples
Fewer than 5	15
5 to 10	18
10 to 25	20
25 to 50	25
More than 50	30

Soil Sampling Depth

Samples should be taken at one-foot depth intervals from the land surface to one foot above the water table or to a confining layer (Ecology, 2000).

Sample Collection and Handling

Soil samples should be analyzed for the parameters listed in Table D-3.

Table D-3. Laboratory analytes and analytical methods for soil samples.

Analyte	Analytical Method
pH	SW-846-9045D
Organic Matter	Dry combustion method (USDA/NRCS, 2004)
Cation Exchange capacity	Effective cation-exchange capacity (USDA/NRCS, 2004)
Nitrate	SM 4500-NO ₃ E, F, or H
Ammonium-Nitrogen	SM-4500-NH ₃ B+ C, D, E, or G
Total Kjeldahl Nitrogen	SM 4500-N _{org} B; SM 4500-N _{org} C
Phosphorus	SM 4500-P B+E or F; Digestion EPA 200.2+Analysis 200.7 or SW-846-6010

SM: Standard Methods for the Examination of Water and Wastewater, 20th Edition.

SW-846: U.S. Environmental Protection Agency (U.S. EPA). Test Methods for Evaluation of Solid Waste, Physical/Chemical Methods. EPA Publication SW-846.

EPA: EPA Methods for Chemical Analysis of Water and Wastes, EPA 600/4-79-020.

Soil Survey Investigations Report No. 42, Soil Survey Laboratory Information Manual, Version 4.0, 2004, USDA/NRCS.

Groundwater Quality Monitoring

A groundwater monitoring plan, in the form of a Quality Assurance Project Plan, should be developed and approved consistent with the *Implementation Guidance for the Groundwater Quality Standards* (Kimsey, 2005) and the *Guidelines for Preparing Quality Assurance Project Plans for Environmental Studies* (Ecology, 2004b). This plan should include the following elements:

- Install a sufficient number of upgradient and downgradient monitoring wells to determine groundwater flow direction and impacts from the CAFO on groundwater quality. These wells should be tagged with an Ecology-issued, unique well ID number, and they must be constructed according to Chapter 173-160 WAC, Minimum Standards for the Construction and Maintenance of Wells.
- Groundwater sampling should follow Ecology's standard operating procedures for sampling monitoring wells (Marti, 2011a) and water supply wells (Marti, 2011b).
- A sampling plan should be developed which includes:
 - o Well location and construction information.
 - o Hydrogeologic description of the site.

- o Groundwater flow direction.
- o Identification of a background well.
- o Sample frequency -- Quarterly groundwater monitoring.
- o Well purging criteria.
- o Sample collection procedures.
- o Sample analysis by an accredited laboratory.
- o Field measurements of groundwater temperature, pH, conductivity, and dissolved oxygen.
- o Static groundwater elevations measured to a NAVD88 datum and groundwater flow direction determined.
- o Quality assurance samples (field blanks, equipment blanks, duplicates).
- o Groundwater quality data and water level data should be entered into Ecology's Environmental Information Management System (EIM).

The U.S. Geological Survey (variously dated) developed and maintains the *National Field Manual for the Collection of Water Quality Data*. This manual contains sampling procedures for the accurate assessment and management of surface water and groundwater resources.

Surface Water Monitoring

The Department of Ecology developed a guidance document that describes the important elements for characterizing water quality conditions of streams and lakes near dairy and concentrated animal feeding operations (CAFOs) (Plotnikoff et al., 2006). This document describes elements for developing a monitoring program that produces reliable data. Examples are provided for establishing monitoring sites, selecting water quality parameters, and analyzing data. An extensive list of references is included for planning more complex projects.

Sample Identification, Handling and General Considerations

Each sample container should be labeled with information regarding the farm name, date, and time collected. Manure application method and location should be recorded for long-term records.

All manure samples should be kept frozen and shipped on ice to the laboratory within 24 to 48 hours of collection. Samples may be refrigerated if delivered immediately to the laboratory.

Quality Assurance

Each CAFO permit will need a Quality Assurance Project Plan (QAPP) that includes all sampling and analysis as described in <https://fortress.wa.gov/ecy/publications/summarypages/0403030.html> . This guidance should be used in developing the QAPP to ensure reliable, representative results.

Laboratory analyses must be conducted by a laboratory accredited in Washington State.

A minimum of 10% blind duplicate samples are necessary to provide adequate quality assurance. For instance, if composite samples are collected at 10 fields in one day, a duplicate composite sample is needed at one of the fields.

Samples collected from an agitated lagoon should include a minimum of one duplicate per round.

One method for collecting a blind duplicate composite sample is to use two buckets at the field where a duplicate is collected. Each time a grab sample is collected during the manure application (3 times for each field), manure can be added to both buckets. At the end of the manure application to the field, each bucket should be treated the same as a typical sample, to ensure that the samples have different sample identification numbers.

A standard reference material should be submitted to the laboratory annually to evaluate accuracy of results (Peters et al., 2003).

Laboratory quality assurance results for all samples should be evaluated to ensure that results are reliable.

Appendix E: Glossary, Acronyms, and Abbreviations

Glossary

Agronomic rate: The rate at which a viable crop can be maintained and there is minimal leaching of chemicals downwards below the root zone. Crops should be managed for maximum nutrient uptake when crops are used for wastewater treatment.

Anthropogenic: Human-caused.

Application efficiency (Ea): The ratio of the average depth of irrigation water infiltrated and stored in the root zone to the average depth of irrigation water applied, expressed as a percentage. Also referred to as AE.

Average annual precipitation: The long-term or historic (generally 30 years or more) arithmetic mean of precipitation (rain, snow, dew) received by an area.

Bulk density: Mass of dry soil per unit volume, determined by drying to constant weight at 105°C, usually expressed as gm/cc or lb/ft³. Rock fragments 2 mm or larger are usually excluded or corrected for after measurement.

Capillary water: Water held in the capillary, or small pores of the soil, usually with soil water pressure (tension) greater than 1/3 bar. Capillary water can move in any direction.

Clean Water Act: A federal act passed in 1972 that contains provisions to restore and maintain the quality of the nation's waters. Section 303(d) of the Clean Water Act establishes the TMDL program.

Contamination: Water that has been degraded such that it does not meet the groundwater quality criteria listed in Chapter 173-200 WAC.

Crop rooting depth: Crop rooting depth is typically taken as the soil depth containing 80 percent of plant roots, measured in feet or inches.

Crop water use: Calculated or measured water used by plants, expressed in inches per day. Same as ET_c except it is expressed as daily use only.

Effective precipitation (Pe): The portion of precipitation that is available to meet crop evapotranspiration. It does not include precipitation that is lost to runoff, deep percolation, or evaporation before the crop can use it.

Effective rooting depth: The depth from which roots extract water. The effective rooting depth is generally the depth from which the crop is currently capable of extracting soil water. However, it may also be expressed as the depth from which the crop can extract water when mature or the depth from which a future crop can extract soil water. Maximum effective root depth depends on the rooting capability of the plant, soil profile characteristics, and moisture levels in the soil profile.

Effluent: An outflowing of water from a natural body of water or from a man-made structure. For example, the treated outflow from a wastewater treatment plant.

Evaporation pan: (1) A standard U.S. Weather Bureau Class A pan (48-inch diameter by 10-inch deep) used to estimate the reference crop evapotranspiration rate. Water levels are measured daily in the pan to determine the amount of evaporation. (2) A pan or container placed at or about crop canopy height containing water. Water evaporated from the device is measured and adjusted by a coefficient to represent estimated crop water use during the period.

Evaporation: The physical process by which a liquid is transformed to the gaseous state, which in irrigation generally is restricted to the change of water from liquid to vapor. Occurs from plant leaf surface, ground surface, water surface, and sprinkler spray.

Evapotranspiration (ET): The combination of water transpired from vegetation and evaporated from soil and plant surfaces. Sometimes called consumptive use (CU).

Field capacity: The amount of water retained by a soil after it has been saturated and has drained freely by gravity. Can be expressed as inches, inches per inch, bars suction, or percent of total available water.

Gross irrigation requirement (Fg): The total irrigation requirement including net crop requirement plus any losses incurred in distributing and applying water and in operating the system. It is generally expressed as depth of water in acre inches per acre or inches.

Growing season: The period, often the frost-free period, during which the climate is such that crops can be produced.

Hydraulic conductivity: The ability of a soil to transmit water flow through it by a unit hydraulic gradient. It is the coefficient k in Darcy's Law. Darcy's Law is used to express flux density (volume of water flowing through a unit cross-sectional area per unit of time). It is usually expressed in length per time (velocity) units, i.e., cm/s, ft/d. In Darcy's Law, where $V = ki$, k is established for a gradient of one. Sometimes called permeability.

Infiltration, infiltration rate: The downward flow of water into the soil at the air-soil interface. Water enters the soil through pores, cracks, wormholes, decayed-root holes, and cavities introduced by tillage. The rate at which water enters soil is called intake rate or infiltration rate.

Irrigation efficiency (Ei): The ratio of the average depth of irrigation water beneficially used to the average depth applied, expressed as a percentage. Beneficial uses include satisfying the soil water deficit, leaching requirement for salinity control, and meeting other plant needs. Generally used to express overall field or farm efficiency, or seasonal irrigation efficiency.

Irrigation: Applying water to the land for growing crops, reclaiming soils, temperature modification, improving crop quality, or other such uses.

Leaching fraction: The ratio of the depth of subsurface drainage water (deep percolation) to the depth of infiltrated irrigation water. (See Leaching requirement.)

Leaching requirement: (1) The amount of irrigation water required to pass through the plant root zone to reduce the salt concentration in the soil for reclamation purposes. (2) The fraction of water from irrigation or rainfall required to pass through the soil to prevent salt accumulation in the plant root zone and sustain production. (See leaching fraction.)

Leaching: Removal of soluble material from soil or other permeable material by the passage of water through it.

Mass Balance: An application of conservation of mass to the analysis of physical systems. By accounting for material entering and leaving a system, mass flows can be identified which might have been unknown, or difficult to measure, without this technique.

National Pollutant Discharge Elimination System (NPDES): National program for issuing, modifying, revoking and reissuing, terminating, monitoring, and enforcing permits, and imposing and enforcing pretreatment requirements under the Clean Water Act. The NPDES program regulates discharges from wastewater treatment plants, large factories, and other facilities that use, process, and discharge water back into lakes, streams, rivers, bays, and oceans.

Net irrigation water requirement: The inches of water, exclusive of effective precipitation, stored soil moisture, or groundwater, that is required for meeting crop evapotranspiration for crop production and other related uses. Such uses may include water required for leaching, frost protection, cooling, and chemigation.

Nonpoint source: Pollution that enters any waters of the state from any dispersed land-based or water-based activities, including but not limited to atmospheric deposition, surface-water runoff from agricultural lands, urban areas, or forest lands, subsurface or underground sources, or discharges from boats or marine vessels not otherwise regulated under the NPDES program. Generally, any unconfined and diffuse source of contamination. Legally, any source of water pollution that does not meet the legal definition of “point source” in section 502(14) of the Clean Water Act.

Nutrient management: Managing the application rate and timing of nutrients (contained in manure, wastewater, or fertilizers) to optimize crop use and reduce potential pollution of groundwater and surface water.

Parameter: Water quality constituent being measured (analyte). A physical, chemical, or biological property whose values determine environmental characteristics or behavior.

Pathogen: Disease-causing microorganisms such as bacteria, protozoa, viruses.

Point source: Sources of pollution that discharge at a specific location from pipes, outfalls, and conveyance channels to a surface water. Examples of point source discharges include municipal wastewater treatment plants, municipal stormwater systems, industrial waste treatment facilities, and construction sites where more than 5 acres of land are cleared.

Pollution: Contamination or other alteration of the physical, chemical, or biological properties of any waters of the state. This includes change in temperature, taste, color, turbidity, or odor of the waters. It also includes discharge of any liquid, gaseous, solid, radioactive, or other substance into any waters of the state. This definition assumes that these changes will,

or are likely to, create a nuisance or render such waters harmful, detrimental, or injurious to (1) public health, safety, or welfare, or (2) domestic, commercial, industrial, agricultural, recreational, or other legitimate beneficial uses, or (3) livestock, wild animals, birds, fish, or other aquatic life.

Pre-sidedress soil nitrate test: The pre-sidedress soil nitrate test (PSNT) is a springtime test to determine the amount of nitrogen present in the soils prior to the commencement of the growing season. This test provides a measure of the amount of nitrogen that needs to be applied to support crop growth.

Root zone: Depth of soil that plant roots readily penetrate and in which the predominant root activity occurs. Preferred term is plant root zone.

Surface waters of the state: Lakes, rivers, ponds, streams, inland waters, salt waters, wetlands and all other surface waters and water courses within the jurisdiction of Washington State.

Water table: The upper surface of a saturated zone below the soil surface where the water is at atmospheric pressure.

Acronyms and Abbreviations

AKART	All Known Available and Reasonable methods of prevention control and Treatment
ASTM	American Society of Testing and Materials
BMP	Best management practices
CAFO	Confined animal feeding operation
CFU	Colony forming unit
NMP	Nutrient management plan
Ecology	Washington State Department of Ecology
EPA	U.S. Environmental Protection Agency
GCL	Geosynthetic clay liner
HDPE	High-density polyethylene
N	Nitrogen
NPDES	National Pollutant Discharge Elimination System
NRCS	National Resources Conservation Service
NUE	Nitrogen use efficiency
POTW	Publically owned treatment works
PSNT	Pre-sidedress Soil Nitrate Test
RCW	Revised Code of Washington
SWDP	State Waste Discharge Permit
UIC	Underground Injection Control
USGS	U.S. Geological Survey
WAC	Washington Administrative Code
WRIA	Water Resource Inventory Area
WSU	Washington State University
WWTP	Wastewater treatment plant

Units of Measurement

ft	feet
g	gram, a unit of mass
mg	milligram
kg	kilograms, a unit of mass equal to 1,000 grams
mgd	million gallons per day
m	meter
mg/L	milligrams per liter (parts per million)
mg N/L	milligrams of nitrogen per liter (parts per million)

Unit Conversions

units	multiply by	to get
lbs N/acre	1.1198	kg N/ha
kg N/ha	0.893	lbs N/acre
mg/kg	equals	ppm
mg/L	equals	ppm
lbs/acre/yr	1.120849251	kg/ha/yr
kg/ha/yr	0.892179438	lbs/acre/yr
acre-ft	1,233,481	liter
acre-ft	43,560	cubic feet
acre-ft	325,851	gallons
gpd	3.78541	liters/day
1 acre	43560	square feet
1 acre	0.404686	hectare
1 cm	0.393701	in
°C	32	°F
1 gallon	3.78541	liters
1 inch	2.54	centimeter
1 foot	0.3048	meters
pounds	0.453592	kilogram
kilogram	2.20462	pound
hectare	2.47105	acre