AN ABSTRACT OF THE THESIS OF

<u>Jeff Feaga</u> for the degree of <u>Master of Science</u> in <u>Bioresource Engineering</u> presented on <u>February 7, 2003.</u>

Title: <u>An Eleven Year Study of Nitrate Leaching in Vegetable Production with Winter Cover Crops</u>

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Abstract approved: _

John S. Selker

The Willamette Valley of Oregon has high rates of winter precipitation that cause leaching losses of residual fertilizer nitrogen (N) as nitrate (NO_3^-). Cover crops may have potential to mitigate N loss. Shallow groundwater was sampled for 11 years from plots planted in summer vegetables with and without winter cover crop management. A randomized complete-block split plot design was used with cover crop system as main plot and three N fertilizer application rates as subplots (N0 for no fertilizer, N1 for intermediate and N2 for the full recommended N inputs). The summer crops were corn (*Zea mays* L. cv. Jubilee), broccoli (*Brassica oleracea* L. Botrytis Group cv. Gem), or snap beans (*Phaseolus vulgaris* L. cv. Oregon 91G) and the cover crop treatments were conventional fallow (called C plots) or fall-seeded cereal rye (*Secale cereale* L var. Wheeler), triticale (*Tritcosecale X* L. var. Celia), or common vetch (*Vicia Sativa*) / triticale mix cover crops during winter (called H plots). Cereal cover crops were used during the first eight years of the study while a legume / cereal mixture was used during the last three winters. Twenty-six 0.26 m^2 passive capillary samplers (PCAPS) continuously sampled water flux at a depth of 1.2 meters. Water was sampled from the PCAPS 249 times over the 11 year period.

A calculated water balance gave an average collection efficiency of 76%. Eleven-year flow-weighted average N in the form of nitrate (NO_3^--N) concentrations (mg L^{-1}) for fallow vs cover cropped plots were 16.7 and 11.9; 9.9 and 6.4; and 7.0 and 4.1 for N2, N1 and N0 plots, respectively. Calculated annual average NO_3^--N mass losses (kg ha^{-1}) for C vs. H plots during the same period were 76.3 and 45.9, 40.0 and 30.2, and 29.0 and 17.3 for N2, N1 and N0 plots, respectively. Analysis of variance indicated that cover crops, N

fertilization rate, and year had significant effects on NO_3^--N concentration and mass losses (P-value < 0.0001). The common vetch / triticale mix cover crop treatment had a significant effect on NO_3^--N concentration and mass losses (P-value < 0.0001) during the low rainfall winter of 2000 and the average rainfall winter of 2002, but NO_3^--N concentrations and mass losses during winter 2001 were among the highest observed throughout the entire study. These results may have been due to complicated environmental processes such as atmospheric N fixation, rainfall differences, and / or mineralization of organic matter. Further investigation is needed to test whether cereal cover crops are more effective NO_3^- scavengers than legume / cereal biculture cover crops.

Three conservative tracers were applied on three separate occasions over an eleven year period (Bromide (Br⁻) 1992; Chloride (Cl⁻) 1995; Br⁻ 2000) and breakthrough was observed using PCAPS. Breakthrough curves were fit to solutions to the Advection Dispersion Equation (ADE) to generate transport parameters. Following the 2000 Br⁻ application, soil was sampled in May, September, and December, 2001. Soils in 10 or 20 cm depth increments were taken from the surface to 1.2 meters total depth and were extracted to determine Br⁻ and NO_3^--N concentration profiles under the various treatments. Collection efficiencies of Br⁻ from PCAPS below H and C plots were significantly effected by cover crop treatment (P-value = 0.02) and were 35 and 47% of the applied mass, respectively. Soil concentration profiles of Br- and NO_3^--N were markedly different between H and C plots, and showed evidence of cover crop uptake, immobilization and mineralization of both chemicals. On average, dispersivity values were 3.74, 20.6, and 16.1 cm for the 1992, 1995, and 2000 applications, respectively. Fitted dispersivity values ranged from 2×10^{-2} to 78.8 cm for individual samplers. Peak tracer concentrations reached sampler depth after 0.91, 0.60, 0.79 pore volumes corresponding to 404, 95, and 367 days for the 1992, 1995, and 2000 applications, respectively. Frequent, large rain events such as those experienced during the 1995 Cl⁻ breakthrough resulted in higher pore water velocities, smaller retardation factors (R), and higher dispersivity values, while below average cumulative rainfalls with few large rain events had the opposite effect (1992 Br⁻ breakthrough). There were no significant

correlations between the log-normally distributed dispersivity values and calculated pore water velocities at the sampler level. Dispersivities at individual samplers showed no correlation in time. Observed Br⁻ soil profiles showed a ten-fold lower dispersion than predicted using PCAPS-based parameters. The results of this study suggested that even if fitting data sets well, the Advection Dispersion Equation is inappropriate for the description of solute spreading in structured soils under variable rainfall conditions.

An Eleven Year Study of Nitrate Leaching in Vegetable Production with Winter Cover Crops

by

Jeff Feaga

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

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Jeff Feaga, Author

ACKNOWLEDGEMENTS

CONTRIBUTION OF AUTHORS

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This thesis is dedicated to my family, who has always given unconditional support.

An Eleven Year Study of Nitrate Leaching in Vegetable Production with Winter Cover Crops

1. General Introduction

Crop production in Oregon and across the United States is very competitive and must be profitable. Conventional farming systems can be effective at maximizing these economic returns, but can have environmental impacts by leaching of pesticides and nitrogen (N) fertilizers to groundwater resources. These substances have a wide range of chemical characteristics that control their fate and transport in the environment. Nitrogen contamination of groundwater is usually associated with nitrate (NO_3^-), a highly soluble compound that is easily leached through the soil profile and transported with moving water. Nitrate can be toxic to infants (Weisenburger, 1993) and infirm people. Nitrate also adversely effects aquatic habitats by stimulating primary production of phototrophic organisms in surface waters, causing anoxic conditions and eutrophication. In response to the risk associated with consuming water contaminated with NO_3^-, the EPA set a 10 mg L^{-1} drinking water standard for N in the NO_3^- compound, or NO_3^--N.

A Mediterranean climate with seasonally high water tables combined with intensive agriculture makes the Willamette Valley a particularly vulnerable location for NO_3^- contamination (Owens, 1990). According to the United States Geological Survey (1998), 9% of wells in the Willamette Valley exceed the EPA 10 mg L^{-1} drinking water standard set for NO_3^--N. In an Oregon Department of Environmental Quality (DEQ) study in the southern Willamette Valley, 281 privately owned domestic drinking water wells in Lane County were sampled, showing that 22% of the wells exceeded the standard (Penhallegon, 1994). In a 2000-2001 study of 476 wells in the Southern Willamette Valley, 35 wells exceeded the drinking water standard, with 21% of the total wells exceeding 7 ppm NO_3^--N (DEQ, 2002).

In uncultivated natural systems, N limits productivity and is cycled through its various chemical forms without significant loss to aquifers. In agricultural systems, the

need to remove the N limitation on crop production results in accumulated free NO_3^-. High potentials for NO_3^- losses from Willamette Valley fields can be associated with the intense cultivation of vegetables and peppermint. These crops require large N inputs and are inefficient at utilizing applied N. Typically the quantity of N removed during harvest is only 30 to 70% of the total quantity of plant-available N available from fertilizer applications, mineralization of organic matter and dissolved in irrigation or rainwater (Hermanson et al., 2000).

In order to find the best management practices (BMP's) for minimizing N losses to groundwater, it is imperative to understand the role of soil variability and environmental factors, such as rainfall, controlling NO_3^- leaching processes. Environmental conditions are very important determinants of NO_3^- leaching potential. Rainfall timing, intensity, and quantity combine with the methods of crop management to make NO_3^- leaching losses variable from season to season and even from day to day. Oregon's Willamette Valley provides optimum conditions for NO_3^- leaching when soluble NO_3^- remaining after harvest is transported below the upper soil with heavy winter rains (Burket et al., 1996). Once NO_3^- leaches below rooting depth, it may enter the groundwater system. The use of winter cover crops is a strategy to capture NO_3^- though crop growth and N assimilation before NO_3^- moves past rooting depth. Climate affects the establishment of vegetable and cover crops and their ability to utilize applied N into plant organic matter and subsequently into soil organic matter after decomposition.

Soils have a wide range of morphologies and chemical properties that affect transport of water and NO_3^- to groundwater. The hydraulic conductivity of undisturbed soils shows great spatial variance and conductivity rates have been shown to have a lognormal distribution at field scales (Kutilek and Nielsen, 1994). It is important to consider areas of the field that exhibit the highest rates of conductivity to assess the vulnerability of a particular field NO_3^- leaching losses because it is estimated that 70% of leaching losses originate from less than 30% of the field land area (Kranz and Kanwar, 1995).

Preferential flow paths can also have a considerable effect on NO_3^- leaching potential; especially in climates were rainfall rates often exceed the infiltration capacity of the soil and cause saturated conditions. Low-lying fields that accumulate standing water from adjacent fields are particularly prone to leaching via preferential flow paths.

Engineers and soil scientists often use chemical tracers and mathematical models to predict and describe the transport and spread of chemicals in soil. Nitrate is not a useful tracer of water movement because of high background concentrations in agricultural soils, rapid chemical transformations, and changes in partitioning within the soil, microbes, and plant material. In addition, NO_3^- pulses usually move through the soil during and after the growing season and can overlap, making it difficult to generate clear breakthrough curves. To overcome these problems, other tracers such as bromide (Br^-) are useful. Bromide is not found in detectable concentrations in soils, it is easily analyzed at low concentrations in the laboratory, it is very soluble in water, and it does not sorb to soil particles or react with other chemicals. Bromide is a conservative tracer because it does not degrade. Bromide is readily taken up by plants as they assimilate soil water. Plant uptake of Br^- has been observed by many researchers (Chao, 1966; Owens, 1985; Kung, 1990; Jemison and Fox, 1991; Schnabel et al., 1995) and is sometimes considered to limit the utility of the chemical as a tracer. Kung (1990) observed that the redistribution of Br^- by plant uptake and subsequent reintroduction to the soil through mineralization of decaying leaves resulted in modifications of the Br^- pulse moving past root depth.

One of the most utilized analytical model for describing chemical transport in soil is the Advective Dispersion Equation (ADE). The ADE models chemical concentration as a function of time and space and associated chemical transport by both advection and chemical dispersion. The differential form of the ADE is presented as follows:

$$\frac{1}{R}\frac{\partial C}{\partial t} = D_{hd}\frac{\partial^2 C}{\partial z^2} - \bar{v}_w\frac{\partial C}{\partial z}$$ (1.1)

C = Concentration of tracer in soil water as a function of time and depth
D_{hd} = Hydrodynamic Dispersion Coefficient (encompasses both molecular diffusion and dispersion) cm^2/s
R = Dimensionless retardation coefficient
z = Stationary point of observation (sampler depth, 120cm)
$\bar{v}_w$ = Pore water velocity

Using the ADE, useful parameters describing the transport of a chemical tracer can be calculated either by measuring a tracer's changing concentration through time at a fixed position or by measuring a tracer's concentration at many points at a given time. An important parameter describing the magnitude of a pulse-applied tracer to spread out in the soil is termed hydrodynamic dispersion. Hydrodynamic dispersion (D_{hd}) is the sum of molecular diffusion (D_0) and mechanical dispersion (D_m) within the system. Mechanical dispersion, usually called simply dispersion, is a function of dispersivity (α) and the pore water velocity (V). Mathematically, the components of D_{hd} are:

$$D_{hd} = D_0 + D_m$$ (1.2)

$$D_m = \alpha \bar{v}_w$$ (1.3)

D_0 = Molecular diffusion (cm^2/s)
D_m = Mechanical dispersion (cm^2/s)
α = Dispersivity (cm)

The dispersivity is considered to be a characteristic parameter of the soil media (Selker et al., 1999; Beven et al., 1993), and therefore should be independent of the pore water velocity (Wierenga and Van Genuchten, 1989). Assuming that α is a static soil parameter implies that variations in pore water velocity alone will determine variability of tracer spread in a particular soil location. Due to the high pore water velocities measured in many experiments, the D_m component has been assumed to dominate D_0

within the system to the point where D_0 is assumed to be negligible (Van Ommen et al., 1989). In practice, many researchers have excluded the contribution of D_0 entirely from the tracer breakthrough analysis. For many studies that control water inputs, this assumption is valid due to the short duration of the study and the high velocities of the system. It is possible that the importance of D_0 varies in systems where conditions range between periods of rainfall and downward percolation of soil water and long periods of relatively dry soil when water remains relatively immobile in the soil matrix.

There were two objectives of this thesis: 1) Quantify NO_3^- leaching losses from soils in vegetable systems under various N fertilizer rates and winter cover crop treatments; and 2) To determine recovery of chemical tracers and describe chemical transport in a Willamette silt loam. Through analysis of tracer breakthrough, the relation of transport parameters in time and space as well as the importance of molecular diffusion in chemical transport were assessed.

2. Eleven Years of Nitrate Leaching Under Vegetable Production With Cereal and Common Vetch Cover Crops

2.1 Abstract

The Willamette Valley of Oregon has high rates of winter precipitation that cause leaching losses of residual fertilizer nitrogen (N) as nitrate (NO_3^-). Cover crops may have potential to mitigate N loss. Shallow groundwater was sampled for 11 years from plots planted in summer vegetables with and without winter cover crop management. A randomized complete-block split plot design was used with cover crop system as main plot and three N fertilizer application rates as subplots (N0 for no fertilizer, N1 for intermediate and N2 for the full recommended N inputs). The summer crops were corn (*Zea mays* L. cv. Jubilee), broccoli (*Brassica oleracea* L. Botrytis Group cv. Gem), or snap beans (*Phaseolus vulgaris* L. cv. Oregon 91G) and the cover crop treatments were conventional fallow (called C plots) or fall-seeded cereal rye (*Secale cereale* L var. Wheeler*)*, triticale (*Tritcosecale X* L. var. Celia), or common vetch (*Vicia Sativa*) / triticale mix cover crops during winter (called H plots). Cereal cover crops were used during the first eight years of the study while a legume / cereal mixture was used during the last three winters. Twenty-six 0.26 m^2 passive capillary samplers (PCAPS) continuously sampled water flux at a depth of 1.2 meters. Water was sampled from the PCAPS 249 times over the 11 year period.

A calculated water balance gave an average collection efficiency of 76%. Eleven-year flow-weighted average N in the form of nitrate (NO_3^--N) concentrations (mg L^{-1}) for fallow vs cover cropped plots were 16.7 and 11.9; 9.9 and 6.4; and 7.0 and 4.1 for N2, N1 and N0 plots, respectively. Calculated annual average NO_3^--N mass losses (kg ha^{-1}) for C vs. H plots during the same period were 76.3 and 45.9, 40.0 and 30.2, and 29.0 and 17.3 for N2, N1 and N0 plots, respectively. Analysis of variance indicated that cover crops, N fertilization rate, and year had significant effects on NO_3^--N concentration and mass losses (*P*-value < 0.0001). The common vetch / triticale mix cover crop treatment had a significant effect on NO_3^--N concentration and mass losses (*P*-value < 0.0001) during the low rainfall winter of 2000 and the average rainfall winter of 2002, but NO_3^--N concentrations and mass losses during winter 2001 were among the highest observed

throughout the entire study. These results may have been due to complicated environmental processes such as atmospheric N fixation, rainfall differences, and / or mineralization of organic matter. Further investigation is needed to test whether cereal cover crops are more effective NO_3^- scavengers than legume / cereal biculture cover crops.

This long-term study demonstrates the effectiveness of nutrient and cover crop management to regulate groundwater NO_3^--N concentrations. With such management, fertilizer application rates could decrease the potential for NO_3^- contamination of drinking water aquifers.

2.2 Introduction

Nitrate (NO_3^-) contamination of groundwater by agricultural practices is a national concern. In the United States Midwestern Corn Belt, agriculture dominates land use and a majority of the population relies on groundwater for drinking (Klocke et al., 1999). Nitrate leaching in Mid-Atlantic and Southern states has been intensely researched due to high density farm operations using intensive row cropping practices, the prevalence of highly permeable sandy coastal plain soils, and the need to regulate nutrient inputs to water courses exhibiting eutrophic conditions (Staver and Brinsfield, 1998; Ritter et al., 1998). Many of these studies have measured and quantified NO_3^--N concentrations in soil water percolating to depths below the root zone and into aquifers. As a result of substantial nitrogen (N) inputs, shallow groundwater aquifers in agricultural areas can contain substantial quantities of NO_3^- and often exceed the EPA 10 mg L^{-1} drinking water standard set for N in the NO_3^- compound, or NO_3^--N (Brandi-Dohrn et al., 1997; Meisinger et al., 1991; Staver and Brinsfield, 1998).

Nitrate leaching processes and the methods of N management are regional. In Western Oregon's Willamette Valley, a Mediterranean (dry summer maritime) climate with nearly 100 cm of rain falling between the months of October and April can make for conditions particularly conducive to NO_3^- leaching. The United States Geological Survey (1998) in a 1993 study found that nine percent of 70 domestic wells in the Willamette Valley exceed the EPA 10 mg L^{-1} drinking water standard for NO_3^--N. A study in Marion County where our study was conducted showed that 28 of the 82 tested wells exceeded the EPA limit (Petit, 1988). In a 2000-2001 study of 476 wells in the Southern Willamette Valley, 35 wells exceeded the drinking water standard, with 21% of the total wells exceeding 7 ppm NO_3^--N (DEQ, 2002).

The Willamette Valley has extensive cultivation of irrigated peppermint and row vegetables, which have high N requirements and low N efficiency. Often, significant amounts of plant available N remain in the soil profile after harvest. With the onset of the rainy season in October or November, soluble NO_3^- is quickly lost and plant residues left on the surface are quickly mineralized. Rainfall characteristics and mild winter temperatures make winter cover crops a valuable best management practice (BMP) to

reduce N loss from row crop systems in western Oregon (Brandi-Dohrn et al., 1997; Burket et al., 1997).

Additions of NO_3^- in aquifer recharge water do not necessarily have an immediate effect on deep groundwater aquifer concentration, such as in wells. Owens et al. (1985) applied a Bromide (Br^-) (as a surrogate for NO_3^-) tracer to two Ohio pastures and found that up to 104 weeks were needed to reach peak concentrations within nearby groundwater springs. For aquifers at risk of NO_3^- contamination, land managers are beginning to realize that the delayed effect of N inputs to raise groundwater NO_3^-- N concentrations obligates that action be taken to develop BMP's before there is a problem with drinking water. Some of the most typical strategies include: determining the timing, rate, and method of N applications, accounting for and managing mineralization rates of crop residue during noncropping periods, using diversified crop rotations, and using various tillage practices (Dinnes, 2002).

The quantity of inorganic N available for leaching has generally been found to be more dependent on cropping and fertilizer application history than fertilizer applications of the current year (Macdonald et al., 1989; Martinez and Guiraud, 1990). Long-term use of cover crops can change the timing of plant available N supply because of uptake and subsequent mineralization. Successful growing operations often rotate crops and management in response to changing demand, therefore studies of systems supporting a stable monoculture may be unrealistic for some regions. Within the context of rotational cropping, few NO_3^- leaching studies have been operated for periods long enough to accurately measure the effect of field management changes on NO_3^-- N concentration in leachate. The relation to NO_3^- leaching from environmental factors such as rainfall timing, amount and intensity is hard to discern during short studies. Experiments with controlled precipitation are of limited relevance to real world processes.

Finding the most appropriate methods for measuring management effects on NO_3^- content within mobile soil water has been a challenge for researchers. Deep well testing is not an ideal method due to the time lag of the groundwater response and other groundwater inputs diluting the NO_3^- and making it difficult to identify the NO_3^- source (Owens et al., 1985; Owens and Edwards, 1992). Sampling methods in the upper vadose

zone are the most capable of timely detection of NO_3^- management effects, but present their own difficulties. Many studies use destructive sampling such as coring. Such methods, particularly with repetition, can change infiltration and drainage characteristics of the soil and disrupt the hydrology of long term study sites. Most sampling devices are incapable of measuring both soil water flux and soil water quality. Zero tension pan lysimeters require positive pore water pressures before water collection can occur. This characteristic leads to lateral flow away from the sampler with collection efficiencies ranging from 10 to 58% (Jemison and Fox, 1992). Suction cup samplers require high negative pressures for operation, making estimates of chemical mass impossible because the volume of soil from which water is extracted is uncertain. In addition, suction cup samplers and soil cores do not well represent water flowing along preferential flow paths (Brandi-Dohrn et al., 1996).

Passive capillary wick samplers have much higher collection efficiencies than pan lysimeters, and are able to sample matrix and preferential water (Boll et al., 1991; Brandi-Dohrn et al., 1997). Brandi-Dohrn et al. (1996) showed that Br^- concentration measured by PCAPS differed from concentrations measured using suction cup samplers during the rising and falling limb of the tracer's breakthrough curve. This was due to the PCAPS sampling of flux concentrations while suction cup samplers obtain matrix concentrations.

2.2.1 Cereal Cover Crops

The use of winter cover crops is a strategy that has potential as a BMP for N as well as soil fertility and conservation. Winter cover crops that are incorporated into the ground before spring planting add organic matter, smother weeds and improve soil tilth (Sattell et al., 1998). Cover crops can reduce erosion on slopes and impede the formation of crust layers due to raindrop effects on bare earth, improving infiltration and water-holding capacity. Besides improving soil quality, cover crops scavenge residual N in the soil after fall harvest, reducing NO_3^- leaching to groundwater (Meisinger et al., 1991; Brandi-Dohrn et al., 1997; Staver and Brinsfield, 1998; McCracken et al., 1994). Researchers have observed varying effectiveness for cover crops to assimilate residual

soil NO_3^- and decrease NO_3^--N concentrations within soil water. In an N_{15} lysimeter study in France an Italian rye grass cover crop reduced NO_3 leaching from 110 kg N ha^{-1} under bare fallow to 40 kg N ha^{-1} (Martinez and Guiraud, 1990). Nitrate -N concentrations in the soil water were reduced from 48 to 25 mg L^{-1}. In a cover crop study using a corn/rye rotation on the Atlantic Coastal plain of Maryland, NO_3^--N concentrations measured by 6 foot deep observation wells were reduced from 17 mg L^{-1} to 12 mg L^{-1} (Meisinger et al., 1990).

There are several desired characteristics of winter cover crops used for managing residual soil NO_3^-. The cover crop must be easy to establish and grow adequately in cooler fall temperatures in order to establish a root network. After being killed in the spring to prepare for planting, the cover crop would ideally make N available to the growing plant through mineralization. In theory, this additional plant available N could supply a considerable portion of the summer crop's N requirements and fertilizer additions could be reduced. The most common and effective cover crops used for N scavenging are cereal crops such as ryegrass and other forage crops which can typically accumulate between 25 and 50 kg ha^{-1}yr^{-1} of N (Wagger and Mengel, 1988; Shennan, 1992; Shipley et al., 1992; Ditch et al., 1993).

Despite the effectiveness of cereal crops to accumulate N, research findings question the ability of disked cereal rye residue to provide available N to the summer crop. Brandi-Dohrn (1997) found that inorganic N contents measured for three consecutive Septembers under cereal rye plots were not significantly different, suggesting that the cereal rye was not contributing to the plant-available N pool through mineralization. On the same experimental plots, Burket (1997) measured crop yields harvested from plots managed under fallow treatment, cereal rye and clover. Consistent with other studies such as McCracken et al. (1989), Mitchell and Teel (1977) and Raimbault et al. (1991), Burket found no yield improvements for corn grown in cereal cover plots versus fallow plots, suggesting that mineralization was not enhancing crop growth. These observations could be a result of high C:N ratios in cereal rye residue, or possibly phytotoxic compounds released in the decomposing cereal rye residue (Raimbault et al., 1991).

2.2.2 Leguminous Cover Crops

Leguminous winter cover crops such as crimson clover, winter pea, and vetch are appealing because while scavenging NO_3^- from the previous crop, additional N may be supplied to the system by fixation of atmospheric N (N_2). Leguminous cover crops can fix N_2 to varying degrees depending on the type of legume, growing region, and accumulated biomass. Leguminous crops are more effective at supplying mineralized N to the growing crop due to greater N concentrations in the biomass (Sainju et al., 1997). Nonetheless, it is common for mineralization to lag behind the period of rapid plant uptake, as was observed with a hairy vetch cover mineralizing after corn silking (after rapid uptake). This observation may be due to dry conditions slowing mineralization rates (Brown et al., 1993).

Studies have shown that most winter legume covers do not scavenge soil NO_3^- as efficiently as cereal crops. McCracken et al., (1994) found that rye reduced NO_3^- leaching by 94% compared with 48% using hairy vetch. To explain the apparent inability of typical leguminous covers to scavenge NO_3^-, Sainju et al. (1998) measured the densities and length of rye, hairy vetch, and crimson clover. Rye had a greater root density and length earlier in the winter than did the leguminous crops. A positive correlation was made between root count and crop biomass and N uptake. A negative correlation was established between root count and NO_3^--N concentration in the soil. Because of temperature, rye crops resume growth earlier in the spring than vetch (McCracken et al., 1994). Meisinger et al. (1991) hypothesized that legumes may achieve N requirements through fixation rather than soil water uptake, making legumes less effective at scavenging residual NO_3^-.

The use of cereal / legume bicultures as winter cover crops is a promising alternative to using monocultures. Bicultures can utilize the scavenging ability of a cereal crop, while the legume decreases the C:N ratio of the residue (Ranells and Wagger, 1997). Sullivan et al., (1991) found that rye had higher concentrations of N grown with hairy vetch than grown without. The moderate C:N ratio associated with bicultures can prevent early mineralization of a legume biculture, and prevent immobilization, which is a risk with rye monoculture (Ranells and Wagger, 1997).

The objectives of this study were to use observations over eleven growing seasons in Western Oregon to: 1) estimate the concentration and mass flux of NO$_3^-$ below the root zone following corn, broccoli, and snap beans grown with and without a winter cover crop 2) Describe the relationship of NO$_3^-$ leaching to rainfall patterns and climatic conditions 3) Determine NO$_3^-$ leaching characteristics under common vetch / triticale mix winter cover crops.

2.3 Methods

2.3.1 Site Description

The study site was at the North Willamette Research and Extension Center (NWREC) located near Aurora, Oregon (45° 17' N and 122° 45' W), at 46 m elevation above sea level. The Willamette Valley has a Mediterranean climate characterized by wet winters and dry summers. A meteorological station has been operated at the NWREC site since 1963. The Pacific Northwest Cooperative Agricultural Weather Network has maintained the meteorological station since October 1, 1998. Weather records from 1963 to 1990 give an annual average precipitation total of 103.6 cm (40.8 inches) (Oregon Climate Service, 2003). Thirty year monthly precipitation and temperature averages as well as the average precipitation from 1992 to 2002 are shown in Figure 2.1. Data are arranged according to months of the water year, which begins in October and ends in September of the posted year. The months included in a water year more closely coincide with the timing of an agricultural growing year. During the dry, sunny summer months, potential evapotranspiration (ET) far exceeds rainfall. Precipitation totals for the years of study are summarized in Table 2.1, showing the driest year had 1/3 the rainfall of the wettest.
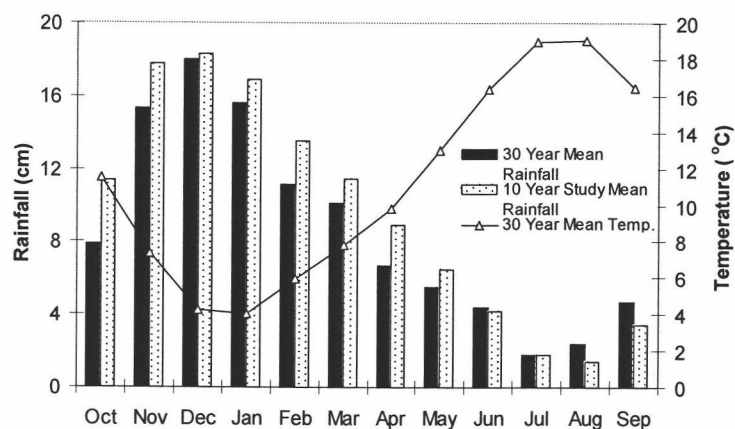
Figure 2.1 Precipitation and temperature characteristics by month at the NWREC (Oregon Climate Service, 2003).

Table 2.1 Total yearly precipitation at the NWREC by water year (October of previous year to the September of posted year) during study period (Oregon Climate Service, 2003).

Water Year	Rainfall (cm)	Relative to 30 Year Average (%)
1993	111.5	108
1994	79.3	77
1995	132.6	128
1996	165.0	159
1997	176.3	170
1998	110.6	107
1999	130.0	125
2000	92.0	89
2001	59.7	58
2002	112.1	108
2003*	100.0	96

* Does not include last four months of water year.

Soils at the 0.98 hectare study site are slightly sloped towards the south (<3%). The soils are of glaciolacustrine genesis and are classified as a Woodburn variant loam with a small strip of Willamette variant loam bisecting the field. Description of organic carbon, pH, bulk density, particle size distribution, saturated hydraulic conductivity and moisture retention characteristics for the NWREC study plot are described in Brandi-Dohrn (1996). Soil bulk density measurements were repeated in 2001; values with depth are presented in Table 3.1 in Chapter Three.

Field management with respect to soil tillage remained the same over the 11 year study period. There was evidence of burrowing rodents at the site, most commonly in the border areas between the experimental plots. Efforts were made to control the rodents with varying success throughout the study. The rodents may have changed conductivity and bulk density values both spatially and temporally. The southeast corner of the field had a slight depression and often became ponded in the winter (Figure 2.2). Soil sampling here showed a compacted layer at 0.7 m depth with low hydraulic conductivity.

2.3.2 Experimental Design

Prior to the initiation of the cover crop studies, the site had been under continuous wheat production and winter fallow for seven years. Starting in 1989, the experimental design changed to a randomized complete block split plot, with cover crop system as main plot and N fertilizer rate as the subplot. The main plot treatments followed summer crops as conventional winter fallow (called C plots) or as a fall planted cover crop (called H plots). The field was divided into 40 plots with dimensions of 18 m by 9 m. Eight of the plots had lysimeters installed in 1992. Each plot was divided into three subplots, two with dimensions of 9 m by 6 m and the control measuring 18 m by 3 m. Nitrogen fertilizer treatments at the sub-plot level were managed as N0, N1, or N2 which represented no addition of fertilizer, an intermediate rate, and the Oregon State University Extension Service recommended amount to achieve maximum yield, respectively. A field diagram is provided in Figure 2.2.

Summer vegetables were irrigated using a hand move sprinkling system. Irrigation sets took place once per week. Water was applied for variable set times according to the time of the year; however a quantitative model of evapotranspiration was not used to fine tune irrigation amounts. Irrigation procedures were consistently managed by the same person throughout the entire study.

Figure 2.2 Field diagram of 0.98 ha experimental site at the NWREC showing sampler location.

Summer vegetable and winter cover crop type changed throughout the experiment and included sweet corn (*Zea mays* L. cv. Jubilee), broccoli (*Brassica oleracea* L. Botrytis Group cv. Gem), and snap beans (*Phaseolus vulgaris* L. cv. Oregon 91G). Crop type and planting and harvesting schedules are summarized in Table 2.2. Cover crops included cereal rye (*Secale cereale* L var. Wheeler), a commonly used cover crop, triticale (*Tritcosecale X* L. var. Celia), a rye / wheat hybrid, and a common vetch (*Vicia Sativa*) / triticale mix during the last three winter seasons of the study. Common vetch is a legume and commonly contains as much as 56-135 of N per hectare (Sattell et al., 1998).

For this analysis, the leaching or crop year was defined as beginning at planting of summer vegetable crops. Therefore, the leaching year includes the summer vegetable crop and the following winter cover crop as a unit. For example, the period of time from the planting of corn on the 30th of May 1996 until May 1997 was defined as 1996. Data

is often subdivided according to the periods defined by the cover crop system used (cereal cover crop and common vetch / triticale mix periods).

To prepare for spring planting, soil was moldboard plowed to a depth of 30 cm. Several weeks later, soil was further worked with a disk-harrow combination and a seedbed was prepared using a spike toothed harrow and packing rollers. Fertilizer was applied two times, the first at planting and the second about one month later (Table 2.3). At harvest in late summer, vegetable crops were removed from 4.5 m transects of two rows within each subplot. Total plant biomass (wet and dry) and total plant N were determined for each crop from a 0.5 kg sub-sample. The remainder of the standing crop was mowed and incorporated back into the soil.

Table 2.2 Crop type and planting and harvesting dates during the experimental period.

Leaching Year	Spring Crop	Seeding	Harvest	Winter Crop	Winter Crop Timing
1992	Sweet Corn	May 20th	Aug	Cereal Rye	
1993	Broccoli	June 9th	Aug 30th	Cereal Rye	
1994	Sweet Corn	July 7th	Sep	Cereal Rye	Seeding late September or early October of posted year
1995	Broccoli	July 11th	Aug 31st	Triticale	
1996	Sweet Corn	May 30th	Sep 5th	Triticale	
1997	Broccoli	July 3rd	Sep 16th	Triticale	
1998	Sweet Corn	June 1st	Sep 9th	Cereal Rye	Harvest late March or early April of following year
1999	Snap Beans	June 10th	Aug 17th	Triticale	
2000	Sweet Corn	May 23rd	Sep 6th	Common vetch / Triticale	
2001	Snap Beans	June 13th	Aug 29th	Common vetch / Triticale	
2002	Sweet Corn	May 14th	Aug 21st	Common vetch / Triticale	

About a month after the summer crop was mowed, cover cropped plots were prepared for planting the cereal cover using a moldboard plow and disk. To prepare plots for common vetch / triticale cover crops, water was added to the soil surface to prepare a soft seed bed. Seeds were broadcast over the surface by hand. Seeding densities and fertilization rates for each vegetable and cover crop are given in Table 2.3.

In late March or early April the success of the cover crop was determined by taking a spatially random 1.0 m^2 sample from each of the sub-plots. A 0.5 kg sub-sample was saved from each random sample to determine total N and dry weight biomass. The cover

crop was killed by mowing, allowed to decompose for several weeks, and turned into the soil with a moldboard plow to begin preparation for another summer vegetable crop. Though not mimicking a typical commercial growing operation, this management practice ensured that all N inputs were accounted for. In this analysis the assumption is made that mineralization rates for the mowed crops reached steady state by the time lysimeters were installed in the fall of 1992 (Brandi Dohrn et al., 1997). Not until the winter of 2000 were N inputs potentially modified when N fixing common vetch was introduced to the rotation.

Table 2.3 Seed planting and fertilizer application densities for vegetable and cover crops during the experimental period.

Crop	Seeding density (Vegetable rows 102 cm apart)	N1 Fertilization (2 applications)	N2 Fertilization (2 applications)
Corn	62,000 plants ha^{-1}	56 kg ha^{-1}	224 kg ha^{-1}
Broccoli	30 cm spacing	140 kg ha^{-1}	280 kg ha^{-1}
Snap beans	51 cm spacing	67 kg ha^{-1}	134 kg ha^{-1}
Cereal Rye	75 kg ha^{-1}	No fertilizer	No fertilizer
Triticale	75 kg ha^{-1}	No fertilizer	No fertilizer
Common Vetch / Triticale	90 kg ha^{-1} Common Vetch 20 kg ha^{-1} triticale	No fertilizer	No fertilizer

2.3.3 Instrumentation

In the summer of 1992, 32 passive capillary samplers (PCAPS) were installed at the NWREC plots. Samplers represented four fallow plots (C) and four cover cropped plots (H). Each plot received four PCAPS, one each in the N1 and N2 sub-plots, and two PCAPS on opposite ends of the N0 control sub-plots. The PCAPS installation was completed to maximize statistical comparability made possible by the complete block split plot design (Figure 2.2). However, due to flooding, four of the subplots were abandoned in block one leaving only three complete replications. The current experimental design utilizes 26 samplers. Published results prior to 1997 from this field

site included data from the six consistently flooded PCAPS, but these data were omitted for this analysis.

PCAPS employ a fiberglass wick that equilibrates to the soil water pressure of the overlying soil. The samplers are therefore capable of intercepting soil water fluxes from soils with pressures ranging from 0 to 0.8 m. As a result of this design, PCAPS can collect water moving through the soil over a range of conditions from saturated to draining and approaching field capacity. PCAPS sample the soil column from a known cross-sectional area, enabling researchers to correlate rainfall, groundwater losses, and ET losses. The wick samplers used at the NWREC were constructed of a 15 kg epoxy coated fiberglass box. The box was covered with a one mm thick stainless steel panel with dimensions of 31 x 84.5 cm. This panel was divided into three sections, each having a calibrated wick attached to the flat surface and entering into the sampler where water was collected in three separate 3.78 L collection vessels. High density polyurethane (HDPE) tubing connected each collection vessel to the soil surface. Though all joints on the sampler were well-sealed using silicone, a HDPE tube was fitted that could drain the fiberglass box should it take on water during periods of high groundwater levels.

PCAPS were installed at a depth of 1.2 meters measured from the soil surface to the wick covered panel oriented parallel to the surface. To ensure that field hydraulic characteristics above the samplers were retained, the units were installed laterally using deep dug trenches. PCAPS were wedged upward to ensure close contact with the overlying soil. A PCAPS unit and installation trench is shown in Figure 2.3. A much more detailed description of wick sampler design can be found in Knutson and Selker (1996) and Brandi-Dohrn (1996). Sampler installation and operation at the NWREC is detailed in Brandi-Dohrn (1996).
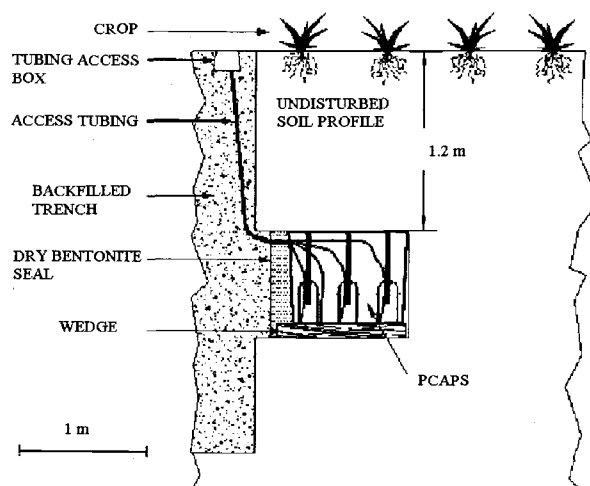
Figure 2.3 Cross section of PCAPS unit and installation trench. Three separate collection vessels and the access tubing for pumping from the surface are shown (Diagram from Brandi-Dohrn, 1993).

2.3.4 Sample Processing

A vacuum pump was used to sample leachate collected from the PCAPS. Samples were taken after cumulative rainfall between sampling events neared 2.5 cm, at which point the collection vessels would be reaching capacity. During late spring and summer it was not necessary to sample as frequently because potential ET rates exceed inputs from rainfall and irrigation. Brandi Dohrn (1996) determined that sampler bias between the three separate PCAPS collection vessels stabilized after the first year of operation; therefore groundwater volumes from each vessel were combined on-site. One sample was taken from these mixed volumes for analysis. Each sampling event provided one sample from each PCAPS unit. Samples were put into HDPE vials and frozen until ion analysis. A Dionex 2000i ion chromatograph with a Dionex AS4A-5C separator column and an AG4A-SC guard column was used to determine anion concentrations. Anions of interest included NO_3^-, Br^-, and Cl^-. Groundwater was sampled from each PCAPS 249 times, starting with the first collection on the 4th of November 1992 until the last on the 26th of April 2003.

2.3.5 Calculations and Data Analysis

PCAPS are capable of measuring both water flux and concentration of dissolved constituents. Therefore, reliable measurements can be made to calculate average concentrations weighted with respect to water flow. Flow-weighted averages can be calculated for multiple samplers and / or samples can be averaged over periods of time ranging from one sampling event to events encompassing multiple growing seasons.

The equation for flow weighting the ion concentration of multiple samplers from the same collection event is:

$$\bar{c} = \frac{\sum_{i=1}^{n} c_i V_i}{\sum_{i=1}^{n} V_i} \tag{2.1}$$

$\bar{c}$ = Mean flow-weighted concentration for multiple samplers
c_i = Concentration as measured by each of n samplers
V_i = Volume of percolation as measured by each of n samplers
n = Number of samplers collecting water

The equation for finding a flow-weighted average ion concentration over multiple sampling events using flow-weighted averages calculated for multiple samplers is:

$$\bar{c}(t) = \frac{\sum_{t=1}^{N} \bar{c}_t \bar{V}_t}{\sum_{t=1}^{N} \bar{V}_t} \tag{2.2}$$

$\bar{c}\,(t)$ = Mean flow-weighted concentration over multiple sampling events
$\bar{c}_t$ = Mean flow-weighted concentration measured for multiple samplers calculated for N sampling events
$\bar{V}_t$ = Average volume of percolation collected during each sampling event calculated for N sampling events
N = Number of sampling events

The equation for average cumulative flux of ions over *t* sampling events is:

$$M = \frac{\sum_{t=1}^{N} \frac{1}{n_t} \left(\sum_{i=1}^{n_t} c_{i,t} V_{i,t} \right)}{A_{PCAPS}}$$ (2.3)

M = Mass flux

$c_{i,t}$ = Concentration as measured by n samplers during time interval $t_{i+1} - t_i$

$V_{i,t}$ = Volume of percolation measured by n samplers during time interval $t_{i+1} - t_i$

N = Number of sampling events

n_t = Number of samplers at sampling event t

A_{PCAPS} = Surface area of PCAPS (2619.5 cm^2)

The experimental design allowed a statistical comparison using analysis of variance (ANOVA) with a general linear model as a randomized complete split-plot block design over years with winter cover crop treatment as main plots and N rate as subplots. ANOVA was made for NO_3^--N concentration and mass during years using only cereal winter cover crops (1992-1999), years using a legume / cereal mix cover crop (2000-2002), and the entire study period. A separate ANOVA analysis was also made for each of the 11 years in the study.

An important statistical assessment of a sampler and its variability is the calculation of the number of PCAPS required to estimate the annual recharge at the site. This number was generated using the mean collection confidence interval generated from the variability of the PCAPS collection efficiencies during the time periods used in a water balance (Equation 2.4). Where $\bar{y}$ is the mean yearly collection efficiency, n is the number of samplers, t is the t statistic with n-1 degrees of freedom and a probability of exceedance of $\alpha/2$, and S is the sample standard deviation. A α value of 0.05 was chosen to give the 95% confidence interval. The sample standard deviation was

$$\bar{y} \pm t_{\alpha/2, n-1} \left(\frac{S}{\sqrt{n}} \right)$$ (2.4)

estimated from the pooled standard deviation of the 78 values provided by the 26 samplers during the three periods in the water balance. For analysis using the t statistic to be valid, measured collection efficiencies must be independent and follow a normal distribution. It was found that the distribution appeared normal, but was not significant at the 0.05 significance level using the Filliben test for normality.

2.4 Results / Discussion

2.4.1 Variability of PCAPS

The reliability of PCAPS was validated using a water balance before an analysis on water quality was made. The hypothesis was that the collection efficiency of the PCAPS would not change or decrease during the experiment. Therefore, the relative collection efficiency of each sampler compared to the mean collection efficiency of all the samplers would not be expected to change. In other words, a sampler that collected water volumes greater than the average at the beginning of the experiment would be expected to do the same at the end of the experiment. Collected volumes from sample events # 5 through # 67 were compared to sample events # 121 through # 172, which corresponded to time periods between the Dec 3rd 1992 through April 11th 1995 and Dec 3rd 1997 through April 20th 2000, respectively. Water collected from the pool of 26 samplers during the first period was summed and ranked by the % deviation from the mean collection amount. The % deviation from the same sampler number during the second group of sample dates are plotted alongside the original rankings (Figure 2.4). Sample collection efficiency varied over the experiment, but the majority of samplers consistently collected greater or less than the mean amount. It appears from these results that there were not any large changes in the structure of soil or flow characteristics in the soil column over the samplers during the course of the experiment.
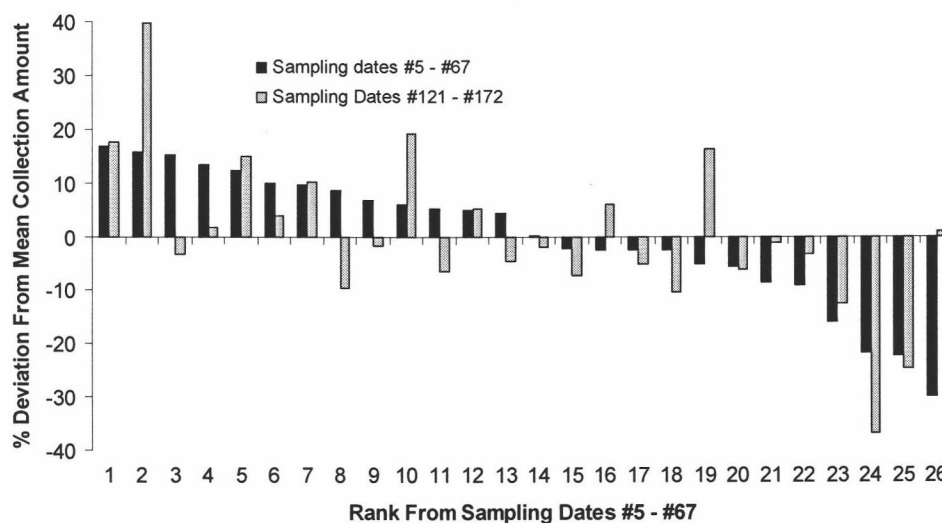
Figure 2.4 PCAPS collection efficiency compared to the mean. PCAPS were expected to consistently sample volumes greater or less than the mean amount.

To determine the reliability of the PCAPS to represent the actual quantity of percolation, a water balance was calculated for three periods of the eleven year study (Table 2.4). The three periods used for the analysis were the months of December through March of the winter of 1993-1994, 1999-2000, and 2001-2002. These periods were chosen because they took place during years of near average precipitation (108%, 89%, and 108% of the 30 year average, respectively), represented the beginning and end of the experiment, and did not include times when fields were often flooded.

Table 2.4 Water balance and collection efficiency of PCAPS for three periods throughout the study.

Period Sampling event	Precipitation	ET	Percolation expected	Percolation observed	Collection efficiency
		------cm------	------	----------	%
# 27 - # 37	55.5	12.9	42.5	20.6	49
# 158 - # 169	48.7	9.3	39.4	31.9	81
# 196 - # 219	64.0	10.5	53.5	53.3	100

To perform the water balance, PCAPS collected water volumes were compared to the theoretical percolation calculated as precipitation minus ET. Evapotranspiration information was obtained on a daily basis and was based on the 1982 Kimberly Penman Equation reference value (Pacific Northwest Cooperative Agricultural Weather Network Website, 2003). For simplification, change in soil storage was not considered as the three periods were during the wettest winter months and the soil profile was at moisture contents near saturation. Surface runoff and lateral subsurface flow were also not considered because of the slight 3% slopes of the field and the inability to accurately measure these parameters.

Collection efficiencies were 49%, 81% and 100% for the winters of 1993-1994, 1999-2000, and 2001-2002, respectively. The low efficiency for the first period could be due to several days of high rainfall during the last week of February 1994 that exceeded the capacity of the samplers and allowed water to bypass the lysimeters. The 100% collection efficiency realized during the last period could be attributed to our assumption that runoff was negligible. On two occasions in winter 2001, water was observed to flow from portions of the adjacent fallow field to the south east and onto the study site where it flooded plots supporting operating PCAPS. Brandi-Dohrn et al. (1996) calculated the collection efficiency of the NWREC PCAPS units for several periods during the early stages of the experiment and found efficiency ranges from 66% to 81%. These values are comparable to the 76% average collection efficiency calculated in this water balance.

The variation in the water balance collection efficiency allowed the determination of the minimum number of samplers needed to estimate the mean annual recharge at the PCAPS field site. With a 15% bound on the mean a minimum of 20 samplers were required. A minimum of six samplers were required to estimate the mean annual recharge with a 30% bound on the mean.

2.4.2 Nitrate Leaching

2.4.2.1 Long term concentrations and leaching patterns

Nitrate - N concentrations and cumulative NO_3^-- N losses for N0, N1 and N2 fertilizer treatments measured by PCAPS through time are shown in Figure 2.5, Figure 2.6, and Figure 2.7, respectively. Differences in NO_3^-- N concentration due to fertilizer application level are apparent. Nitrate - N concentration and cumulative NO_3^--N mass were consistently higher under fallow crop rotations (C) than under cover crop treatments (H). Concentrations of $N0_3^-$- N were often greater than the EPA 10 mg L^{-1} standard set for water intended for drinking, especially for plots receiving N2 fertilizer additions.

Nitrate- N concentration varied through time, and periods of high and low NO_3^-- N leaching losses reflect the seasonal pattern of rainfall in the Willamette Valley. Nitrate - N concentrations were typically the highest during the fall and early winter. As winter approached and fall rains began, the soil profile became saturated and most of the NO_3^- left after harvest was leached below the sampling depth. Nitrate - N concentrations decreased during this time due to the high volumes of water diluting the NO_3^-- N. The highest NO_3^-- N mass flux rates in the fall and early winter period were related to periods of highest NO_3^-- N concentration but lagged in time behind them.

Water collected from unfertilized N0 plots contained NO_3^- throughout the study and patterns of fluctuating NO_3^-- N concentration and leaching losses were the same as for N1 and N2 plots. Nitrogen was present below unfertilized plots due to the reservoir of N available for mineralization contained in soil minerals and humus. Weathering and mineralization made these forms of N plant-available. More plant-available N could have been added to the system through groundwater pumped irrigation water. Finally, N could have been added to the system through atmospheric N fixation during the last three seasons using the common vetch / cereal mix cover crop. Background (N0) NO_3^-- N concentrations and mass losses were not subtracted from N1 and N2 plots for this analysis, but it is important to realize the dynamic N cycle under control plots.
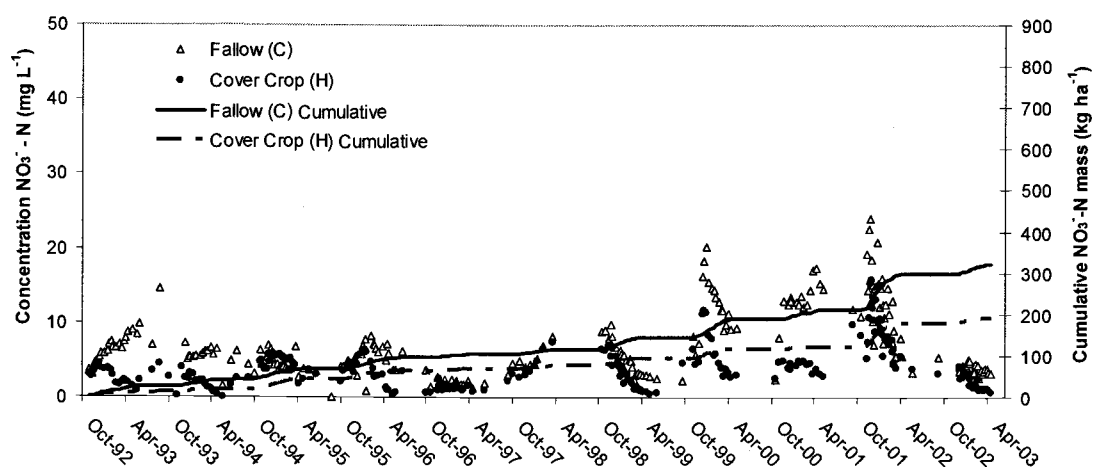
Figure 2.5 Flow-weighted NO_3^--N concentrations and cumulative mass leaching with time for N0 plots.
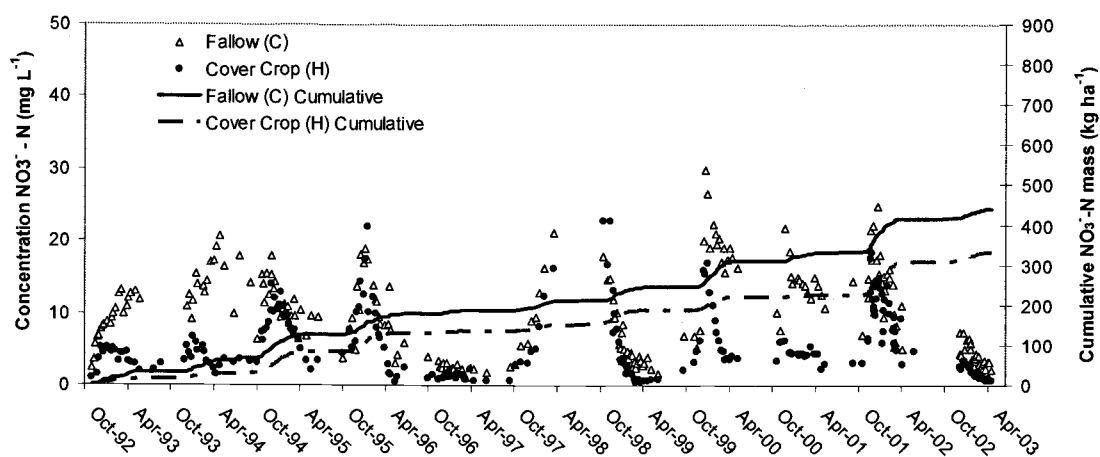
Figure 2.6 Flow-weighted NO_3^--N concentrations and cumulative mass leaching with time for N1 plots.
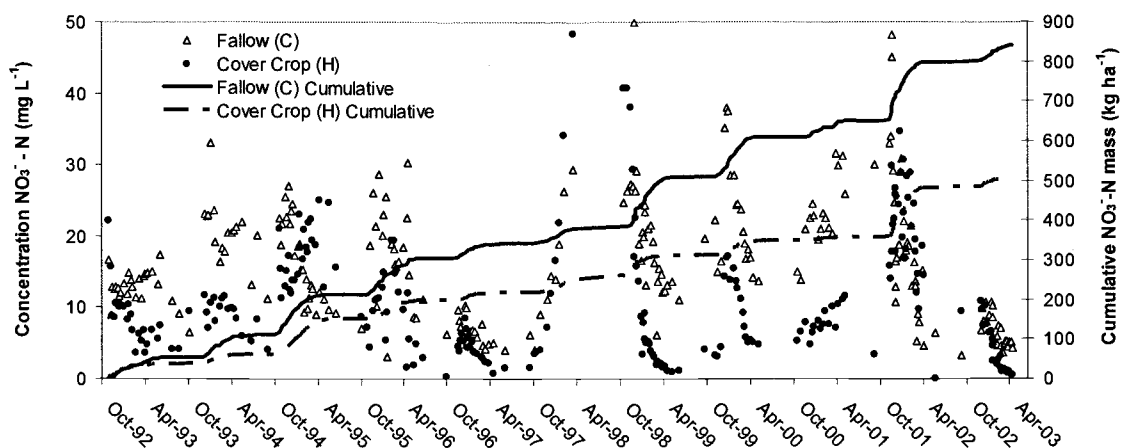
Figure 2.7 Flow-weighted NO$_3^-$-N concentrations and cumulative mass leaching with time for N2 plots.

2.4.2.2 Effect of cover cropping

Flow-weighted average NO$_3^-$- N concentrations and mass losses for each year of the study as well as periods using different cover crop treatments are summarized in Tables 2.5 and 2.6. Flow-weighted average NO$_3^-$-N concentrations (mg L^{-1}) for fallow vs cover cropped plots over the entire study were 16.7 and 11.9; 9.9 and 6.4; and 7.0 and 4.1 for N2, N1 and N0 plots, respectively. Calculated average annual NO$_3^-$-N mass losses (kg ha$^-$) for fallow vs. cover cropped plots over the entire study were 76.3 and 45.9, 40.0 and 30.2, and 29.0 and 17.3 for N2, N1 and N0 plots, respectively.

Annual flow-weighted average NO$_3^-$- N concentrations throughout the study for fields fertilized at recommended agronomic N rates (N2 plots) ranged from 5.0 to 23.3 mg L^{-1} and 6.7 to 24.9 mg L^{-1} for cover cropped and fallow fields, respectively. Annual average NO$_3^-$- N mass losses on N2 plots during this same period ranged from 21.0 to 125.7 kg ha^{-1} and 37.8 to 150.4 kg ha^{-1} for cover cropped and fallow fields, respectively. The distribution of calculated yearly flow-weighted NO$_3^-$- N concentrations and NO$_3^-$- N mass losses on N2 plots are shown in boxplot form in Figure 2.8.

Table 2.5 Average NO_3^-- N concentration measured under various fertilizer treatments. Cover cropped plots are divided into the periods using cereal cover crops and leguminous cover crops.

| Year (winter) | N2 | | | N1 | | | N0 | | |
| | C Plots | H Plots | Δ due to cover | C Plots | H Plots | Δ due to cover | C Plots | H Plots | Δ due to cover |
	mg L^{-1}		%	mg L^{-1}		%	mg L^{-1}		%
1992	13.4	9.6	28	8.3	4.6	45	6.5	2.8	56
1993	22.1	11.2	49	13.3	4.3	67	6.0	2.2	63
1994	18.0	16.1	10	11.9	8.7	27	5.0	4.5	9
1995	18.2	12.2	33	11.4	9.1	21	6.0	3.7	38
1996	6.7	5.0	26	2.3	1.2	47	2.3	1.3	43
1997	15.6	15.5	0	10.0	6.2	37	5.0	4.0	20
1998	20.1	11.5	43	6.4	5.5	14	6.1	3.5	43
1999	24.9	11.1	56	17.5	8.9	49	12.9	6.3	51
2000	26.4	8.2	69	15.0	4.5	70	13.7	3.9	71
2001	21.3	23.3	-9	13.9	10.2	27	12.7	9.1	28
2002	7.0	3.8	52	4.4	3.4	24	4.4	2.0	46
1992 - 1999 Cereal	17.1	11.4	34	9.8	6.1	37	6.0	3.6	41
2000 - 2002 Legume / Cereal	16.1	13.1	19	10.0	6.9	32	9.4	5.6	41
1992 – 2002 Entire Study	16.7	11.9	29	9.9	6.4	36	7.0	4.1	41

Table 2.7 shows the resulting P-values from analysis of variance comparing the differences in flow-weighted average NO_3^-- N concentration and mass over years of the experiment using only cereal cover crops, legume / cereal mix cover crops, and the entire experiment. ANOVA results showed that cover crop treatment, N fertilizer rate, and year all had a significant effect (P value of ≤ 0.001) on NO_3^-- N concentration and mass during each of the different time periods. Using cover crops reduced the NO_3^-- N concentration by 29, 36, and 41% for N2, N1 and N0 plots, respectively. Reductions in yearly NO_3^-- N mass leached due to cover crops were 40, 25, and 40% for N2, N1 and N0 plots, respectively. Year also had a significant effect on variance of NO_3^-- N concentration and mass and could be explained by differences in crop type, winter

Table 2.6 Average NO_3^-- N mass measured under various fertilizer treatments. Cover cropped plots are divided into the periods using cereal cover crops and leguminous cover crops.

Year	N2			N1			N0		
	C Plots	H Plots	Δ due to cover	C Plots	H Plots	Δ due to cover	C Plots	H Plots	Δ due to cover
	kg ha⁻¹		%	kg ha⁻¹		%	kg ha⁻¹		%
1992	51.3	36.2	30	32.2	13.4	58	25.5	9.2	64
1993	56.2	24.6	56	29.2	13.2	55	15.0	6.6	56
1994	107.6	89.4	17	65.9	55.8	15	25.5	26.4	-4
1995	90.0	47.5	47	49.4	46.0	7	27.9	18.8	33
1996	37.8	21.0	44	10.9	6.9	37	8.9	6.5	27
1997	38.0	34.8	8	22.6	14.8	35	11.7	8.9	24
1998	128.9	58.5	55	36.0	36.8	-2	29.9	18.0	40
1999	98.3	34.2	65	65.4	32.7	50	43.1	21.7	50
2000	40.9	10.6	74	21.0	7.5	64	22.9	6.0	74
2001	150.4	125.7	16	85.1	79.5	7	84.9	56.8	33
2002	39.9	21.9	44	22.2	25.1	-13	23.2	11.5	51
1992 - 1999 Cereal Avg.	76.0	43.3	43	39.0	27.5	29	24.0	14.6	39
2000 - 2002 Legume / Cereal Avg.	76.9	52.7	31	42.7	37.3	13	43.7	24.7	43
1992 - 2002 Entire Study	76.3	45.9	40	40.0	30.2	25	29.0	17.3	40

rainfall conditions, and cover crop establishment during each season; however, no statistics were used to test the significance of these factors. ANOVA also showed that several significant combination effects were expressed. The strong effect of year*cover crop (P-value < 0.0001) indicated that the relationship between these factors were not truly linear as would be the case in an ideal complete block split plot experimental design.

ANOVA of NO_3^-- N concentration and mass were made separately during each of the 11 study years to test the effect of cover crops and N fertilizer rates on an annual basis (Table 2.8). Cover crop treatment had a significant effect (P-value < 0.05) on both NO_3^--N concentration and mass losses four out of eleven years. Nitrogen fertilization rate had a significant effect (P-value < 0.05) on both NO_3^-- N concentration and mass losses ten

out of eleven years. Levels of significance were not as high for annual ANOVA tests as they were for ANOVA tests calculated with data combined over multiple years because statistical power was lost using the smaller yearly data sets. Nonetheless, *P*-values generated by both annual and over years ANOVA reveal how effective cover crops were to significantly reduce NO_3^-- N concentration and mass losses compared to the fallow control treatments.
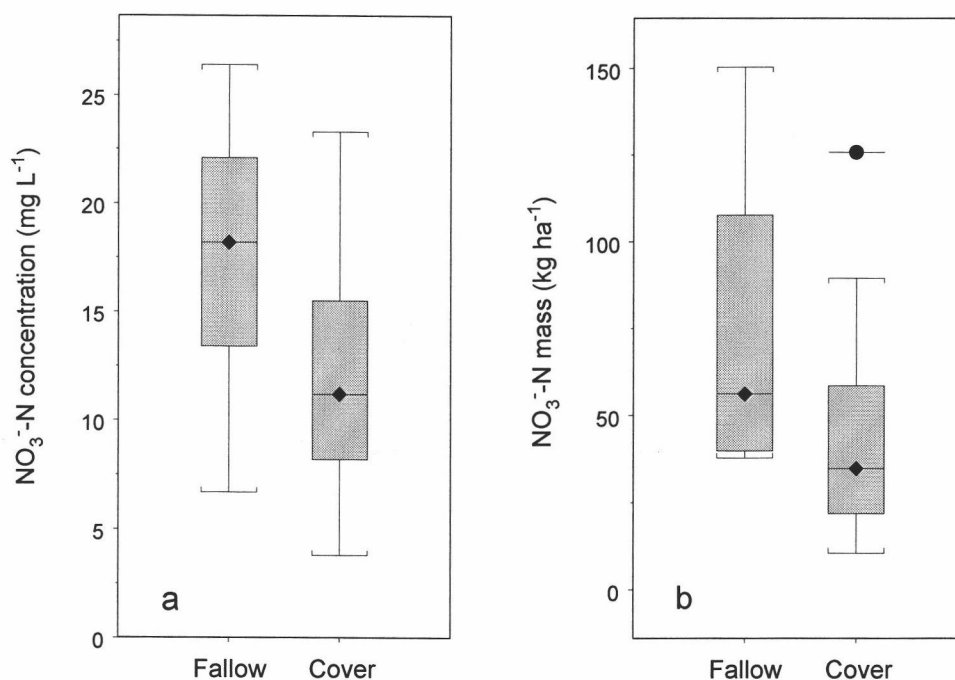
Figure 2.8 Range of flow-weighted average NO_3^-- N concentration (a) and mass losses measured under N2 plots throughout the entire 11 year study period, (b). Box and whisker plot shows a box with borders at the 25 and 75% quartiles and the statistical median in the center. The whiskers extend to the farthest points that are not outliers (i.e., that are within 3/2 times the interquartile range). A dot is shown for every point more than 3/2 times the interquartile range.

Table 2.7 Summary of analyses of variance showing the sources of effects on NO_3^--N concentration and mass losses during key periods of the study.

Time Period	Source of Variation	NO_3^--N Concentration		NO_3^--N Mass	
		df	P-value	df	P-value
Cereal Cover Crop Years (1992-1999)	Year	7	< 0.0001	7	< 0.0001
	Cover crop	1	< 0.0001	1	< 0.0001
	N fert. rate	2	< 0.0001	2	< 0.0001
	Block	3	0.0919	3	0.0202
	Year * Cover crop	7	0.0114	7	0.0686
	Cover * N fert. rate	2	0.1263	2	0.0042
	Year * N fert. rate	14	0.2919	14	0.0065
Common Vetch / Cereal Mix Cover Crop Years (2000-2002)	Year	2	< 0.0001	2	< 0.0001
	Cover crop	1	< 0.0001	1	0.0010
	N fert. rate	2	< 0.0001	2	< 0.0001
	Block	3	0.4814	3	0.0317
	Year * Cover crop	2	< 0.0001	2	0.5786
	Cover * N fert. rate	2	0.6313	2	0.2630
	Year * N fert. rate	4	0.0182	4	0.0005
Entire Study Period (1992-2002)	Year	10	< 0.0001	10	< 0.0001
	Cover crop	1	< 0.0001	1	< 0.0001
	N fert. rate	2	< 0.0001	2	< 0.0001
	Block	3	0.2457	3	0.0055
	Year * Cover crop	10	< 0.0001	10	0.1600
	Cover * N fert. rate	2	0.1071	2	0.0027
	Year * N fert. rate	20	0.0584	20	< 0.0001

Apparent was the observation that cover crop treatment significantly effected NO_3^--N concentration and mass (as determined by yearly separated ANOVA) during every year that rainfall was less than the average (water years 1994, 2000, and 2001; corresponding to leaching years 1993, 1999, and 2000). Conversely, very rainy winters such as 1994 and 1996 did not show differences in NO_3^--N concentration and mass due to cover crop treatment. This trend was not always the case, however, as water year 1996 had 159% of the usual rainfall, but still had a significant cover crop effect on NO_3^--N mass losses during leaching year 1995. It is inferred from these observations that high rates of NO_3^- leaching to deeper depths would be expected during years where cover crop establishment is weak because of poor growing conditions and / or consistent rainfalls

Table 2.8 Summary of analyses of variance showing the significance of cover crop treatment and N fertilizer rate as a source of variation for NO3-- N concentration and mass losses each year of the study. Cover crop tests had 1 degree of freedom and N fertilizer rate had two degrees of freedom.

Year	Cover Crop as a Source of Variation		N Fertilizer Rate as a Source of Variation	
	NO_3^--N Concentration	NO_3^--N Mass	NO_3^--N Concentration	NO_3^--N Mass
		P-value		
1992	0.0599	0.0669	0.0257	0.0430
1993	0.0138	0.0071	0.0063	0.0022
1994	0.5475	0.5139	0.0006	0.0002
1995	0.1103	0.0482	0.0017	0.0004
1996	0.2149	0.1214	0.0030	0.0017
1997	0.1356	0.1115	0.0002	0.0004
1998	0.1228	0.1023	0.0086	0.0107
1999	0.0006	0.0010	0.0108	0.0177
2000	< 0.0001	< 0.0001	0.0037	0.0129
2001	0.4007	0.1855	0.0097	0.0068
2002	0.0163	0.0651	0.0969	0.1977

begin before crop roots are well established. Staver and Brinsfield (1998) observed improved N scavenging for cover crops planted after corn harvest due to the limited number of leaching events in early autumn.

When there is normal rainfall in early winter with below average rainfall in late winter, the bulk of NO_3^- mass left after harvest may not reach sampler depth and will over-winter in the soil profile. At these depths, NO_3^- does not undergo rapid chemical change nor can it be used by growing crops during the following summer because few vegetable roots penetrate to this depth. This situation can result in high NO_3^-- N concentration and mass losses the following winter as the residual NO_3^- of two growing seasons leach to sampling depth during one winter.

The leaching of two season's excess NO_3^- is a possible explanation for the extremely high NO_3^-- N concentration and mass losses recorded during leaching year 1994 as the previous water year (October 1993 to September 1994) received only 77% of the long-term average precipitation. This process may also explain NO_3^-- N observations in 2001 under common vetch / triticale mix cover crops and is discussed in the following section.

Careful observation and statistical testing of the relationship of rainfall amount, timing, and vegetable and cover crop establishment are needed to completely understand the relationship between cover crop NO_3^- uptake and climate. These types of comparisons and correlations are difficult, however, due to the challenge of maintaining an experimental control.

2.4.2.3 Assessment of NO_3^- leaching under legume / cereal mix cover crops

The final years of the study using common vetch / triticale cover crops resulted in observed flow-weighted NO_3^-- N concentrations and mass losses that were significantly affected by cover crop treatment and N fertilization rate (ANOVA over three years, *P*-value < 0.0001) (Table 2.7). These results were similar to those observed under cereal cover crops, but observations of NO_3^-- N concentration and mass losses were very different each of the three years (Figure 2.9). Figure 2.10 shows average mass of NO_3^- lost during this same period. Yearly observations may reflect complicated environmental factors such as atmospheric N fixation, rainfall differences, and / or mineralization of organic matter. During 2000 and 2002, separated yearly analysis of variance shows that cover crop treatment significant effected NO_3^-- N concentration. Cover crop treatment could not explain variance in NO_3^-- N concentration or mass in 2001.

Nitrate – N concentrations during 2001 were generally high for fallow plots at all fertilizer treatments, and were the absolute highest concentrations measured during the 11 year study under cover cropped plots at all three N fertilization rates. There are several possible explanations for the high NO_3^-- N concentrations and mass losses in 2001
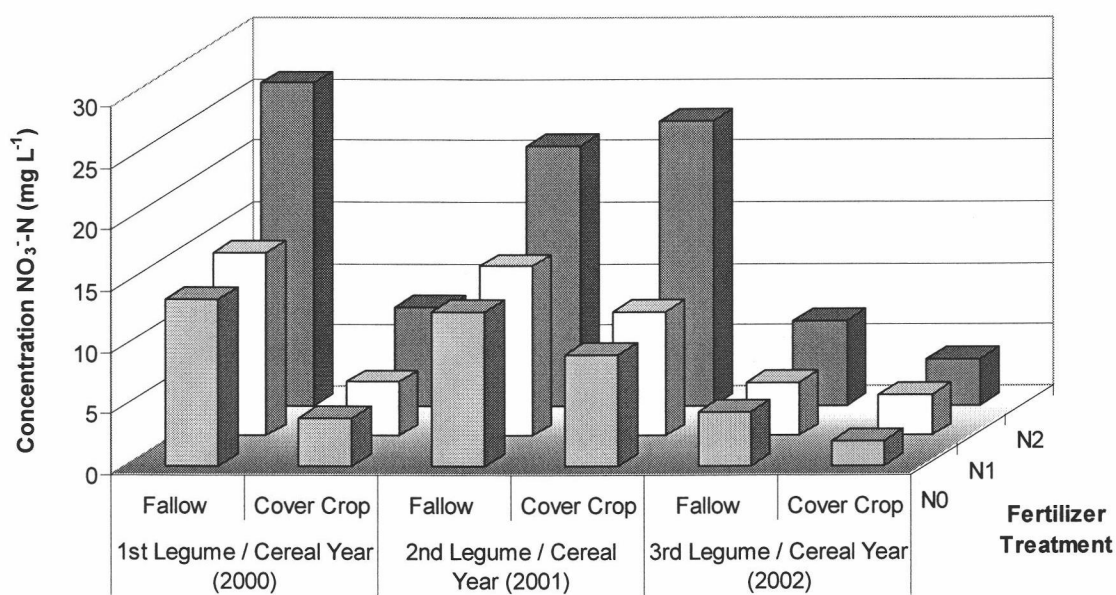
Figure 2.9 Flow-weighted average NO_3^--N concentrations during three season using common vetch / triticale mix cover crops.
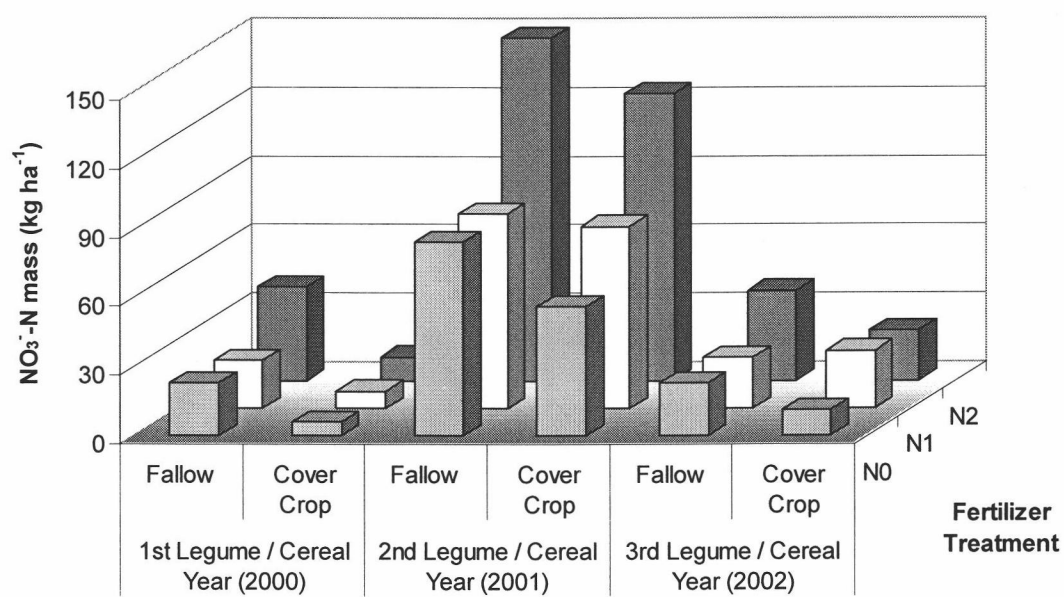
Figure 2.10 Average NO_3^--N mass losses during three season using common vetch / triticale mix cover crops.

compared to the other years using common vetch / triticale mix cover crops. Because mineralization rates were not determined nor organic matter contents in the soil measured over time, it is impossible to positively attribute observations in 2001 to an increase in NO_3^- made available through atmospheric fixation of N by the common vetch. A probable explanation for the variable observations is the difference in rainfall patterns experienced during the last three years of the study. High concentrations and little mass loss during the year 2000 using common vetch could be explained by the extremely dry conditions in water year 2001 when only 58% of the long-term average rainfall was collected. The second year of common vetch / triticale cover crops had typical rainfall accumulations, with 108% of the long-term precipitation average. Similar to 1994, it is possible that the leaching of two season's worth of NO_3^- can explain the observations made in 2001.

Though results were variable under common vetch / triticale mix cover crops, NO_3^--N concentration and mass losses were affected by winter cover crop treatment. It is possible that the bulk of NO_3^- scavenging ability of this mix cover crop was due to the triticale portion of the cover crop mixture. Further investigation is needed to test whether cereal cover crops are more effective NO_3^- scavengers than common vetch / triticale biculture cover crops.

2.5 Conclusions

The analysis of water collected below the root zone at 1.2 meters depth indicated that the use of cover crops and N fertilizer application rates had a significant effect (*P*-value <0.0001) on NO_3^-- N concentration and mass losses over the 11 year study. Fall planted cereal cover crops used during the first eight years as well as common vetch / triticale mix cover crops planted during the last three years successfully scavenged residual NO_3^- remaining in the upper soil column after summer vegetable growth. Further investigation under legume / cereal cover crops is necessary to evaluate the relation of NO_3^- leaching to legume mineralization.

Despite the differences in average NO_3^-- N concentration and mass losses achieved using cover crops, all plots fertilized at recommended N rates were higher than the EPA's

10 mg L^{-1} drinking water standard set for NO_3^-- N. Similar NO_3^- leaching characteristics would be expected under row vegetable growing operation throughout the Willamette Valley. These results stress the importance of employing nutrient management strategies before NO_3^-- N concentrations increase in drinking water aquifers.

From the yearly variability of NO_3^-- N concentrations and mass losses, it was clear that environmental factors, especially timing and quantity of rainfall, had pronounced effects on NO_3^- leaching. Seed germination and establishment of the winter cover crop is important for successful winter scavenging. Large rain events before good cover crop root establishment can leach NO_3^- to depths unreachable by crop roots and reduce the efficiency of the cover crop to scavenge residual N contained in the soil water.

This long-term study makes a strong demonstration for the effectiveness of winter cover crop management as an option to limit NO_3^- leaching losses to groundwater. It is likely that organic matter levels would eventually increase in soils under long-term winter cereal crop rotations. Long term studies, on the order of decades, are needed to reveal the possible improvements for crop growth and the opportunity of applying N fertilizer credits from increased mineralization rates. The highly correlated response of NO_3^-- N concentration to application rates infer that nutrient management strategies must continue to achieve increases in N efficiencies through application methods and timing without compromising crop yield. With such management, fertilizer application rates could possibly be decreased and contamination of drinking water aquifers by agriculture would be reduced.

Acknowledgements

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3. Cover Crop Effects on Bromide Tracer Recovery and Soil Nitrate Distribution

3.1 Abstract

A conservative tracer was used to test the hypothesis that the mechanism by which cover crops scavenge residual nitrate (NO_3^-) is through uptake and retention during the winter with redistribution to the soil surface in spring. Bromide (Br^-) was applied in December 2000 to plots planted in summer vegetable plots with (called H plots) and without (called C plots) winter cover crop management. Three fertilizer treatments (N0 for no fertilizer, N1 for half and N2 for vegetable crop recommendation) were managed at the subplot level. During winter, H plots had a common vetch (*Vicia Sativa*) / triticale (*Tritcosecale X* L. var. Celia) mix cover crop while C plots were left fallow. Snap beans (*Phaseolus vulgaris* L. cv. Oregon 91G) were planted on plots during the summer of 2001. Twenty-six passive capillary samplers (PCAPS), each with a collection area of $0.26 m^2$, were installed at 1.2 meters depth. Flux concentrations of Br^- within percolating soil water were measured. Soil was sampled to 1.2 meters depth during May, September, and December of 2001 using a hand auger and extracted to determine vertical Br^- and NO_3^-- N concentration profiles under the various treatments. PCAPS recovered peak Br^- concentrations were 14.3 and 23.1 mg L^{-1} while fitted peak concentrations were 10.9 and 14.7 mg L^{-1} for H and C plots, respectively. Average Br^- collection efficiencies from PCAPS below H and C plots were 35 and 47 % of the applied mass, respectively. Cover crop treatment had a significant effect on Br^- collection efficiency (*P*-value of 0.02). Vertical concentration profiles of Br- and NO_3^-- N between H and C plots showed evidence of cover crop uptake, retention and mineralization of both chemicals. Nitrate - N concentrations reflected increasing N fertilizer application rates at the subplot level. PCAPS collected and soil extracted Br^- mass were added after the May auger sampling to calculate an 83% and 95% tracer recovery for the cover cropped and fallow fields, respectively. Through uptake and retention, cover crops can assimilate NO_3^- and therefore reduce the amount of NO_3^- available to leaching processes.

3.2 Introduction

The use of winter cover crops is a strategy that has potential as a best management practice (BMP) for nitrogen (N) as well as soil fertility and conservation. Winter cover crops that are incorporated into the ground before spring planting protect the soil surface, add organic matter, smother weeds, and improve soil tilth (Sattell et al., 1998). Cover crops maintain organic matter content, reduce erosion on slopes and impede the formation of crust layers due to raindrop effects on bare earth, improving infiltration and water-holding capacity during dry summer months. Besides improving soil quality, cover crops scavenge excess N left in the soil after fall harvest, reducing NO_3^- leaching to groundwater. (Meisinger et al., 1991; Brandi-Dohrn et al., 1997; Staver and Brinsfield, 1998; McCracken et al., 1994).

Despite the effectiveness of cereal crops to accumulate N, research findings question the ability of disked cereal rye residue to provide available N to the summer crop. Burkett (1997) measured crop yields harvested from plots managed under fallow treatment, cereal rye and clover. Consistent with other studies such as McCracken et al. (1989), Mitchell and Teel (1977) and Raimbault et al. (1991), Burkett found no yield improvements for corn grown in cereal cover plots versus fallow plots, suggesting that mineralization was not enhancing crop growth.

Leguminous winter cover crops such a crimson clover, winter pea, and vetch are appealing because while scavenging NO_3^- from the previous crop, additional N may be supplied to the system. In a three year study by Ranells and Wagger (1997) comparing grass cover crops and legume monocultures, the top 90 cm of soil contained more total inorganic N in the legume plots. Other studies showing higher inorganic N concentration after legume cover crop cultures include Sainju et al., (1998) and Brown et al., (1993).

Bromide⁻ is often used as a surrogate tracer for NO_3^- because it is easily analyzed, has low background concentrations and is not reactive or adsorbed in soils (Kung, 1990). Hargrove and Bausch (1973) used leaching rates of Br⁻ to compare leaching rates from N fertilizers. The ability of plants to take up Br⁻ from soil water is well recorded (Chao, 1966; Owens et al., 1985; Kung, 1990; Schnabel et al., 1995). Plants passively assimilate Br⁻ while taking up soil water. Because of crop uptake, the ability of Br⁻ as a tracer

describing water movement has been questioned. Tracer breakthrough characteristics in soil can be modified if Br⁻ is taken up by plants and redistributed on the surface through plant decay, leaf fall, or ingestion by grazing animals (Owens et al., 1985; Kung, 1990). Plant uptake of Br⁻ in surrogate studies could result in overestimation or underestimation of leaching depending on the reactivity and absorption qualities of the modeled chemical (Kung, 1990).

The objectives of this study were to 1) Quantify the mass recovery of a surface applied Br⁻ tracer under cover cropped and fallow plots 2) Observe Br⁻ and NO_3^-- N concentration profiles attained through manual soil sampling from cover cropped and fallow plots 3) Describe the effect of cover crop uptake and mineralization on the distribution of Br⁻ and NO_3^- in the vadose zone.

3.3 Methods

3.3.1 Site Description

Complete descriptions of the location, climate, soil characteristics and field management are given in Chapter Two. During the two winters included in this analysis, cover cropped (H) plots were planted in a biculture of common vetch (*Vicia Sativa*) and the cereal crop triticale (*Tritcosecale X* L. var. Celia) while C plots were left fallow. Plots were in corn (*Zea mays* L. cv. Jubilee) during the summer of 2000 preceding this analysis and snap beans (*Phaseolus vulgaris* L. cv. Oregon 91G) were planted on all plots during the summer of 2001.

Soils located within the 0.98 hectare study site are slightly sloped towards the south (<3%). The soil is classified as a Willamette silt loam and is of glaciolacustrine genesis. Description of organic carbon, pH, bulk density, particle size distribution, saturated hydraulic conductivity and moisture retention characteristics for the NWREC study plot are described in Brandi-Dohrn (1996). Soil bulk density measurements were repeated in 2001 (Table 3.1).

Table 3.1 Results of 2001 bulk density measurements as a function of depth and crop treatment.

Depth Increment	Bulk Density (g/cm^3) (Standard deviation in parenthesis)		
	Average	Cover Crop Average	Fallow Average
0 – 10	1.07 (0.16)	1.14 (0.20)	1.02 (0.12)
10 – 20	1.28 (0.17)	1.35 (0.25)	1.22 (0.07)
20 – 30	1.44 (0.15)	1.39 (0.07)	1.48 (0.19)
30 – 40	1.56 (0.21)	1.57 (0.30)	1.55 (0.15)
40 – 50	1.57 (0.27)	1.63 (0.36)	1.52 (0.19)
50 – 60	1.42 (0.12)	1.42 (0.18)	1.43 (0.06)
60 – 70	1.29 (0.15)	1.29 (0.18)	1.30 (0.15)
70 – 80	1.28 (0.18)	1.20 (0.13)	1.34 (0.20)
80 – 90	1.23 (0.12)	1.20 (0.14)	1.25 (0.12)
90 – 100	1.27 (0.17)	1.25 (0.21)	1.28 (0.16)
100 – 110	1.30 (0.13)	1.32 (0.13)	1.28 (0.14)
110 – 120	1.44 (0.25)	1.50 (0.31)	1.39 (0.22)

3.3.2 Instrumentation and Sample Processing

A complete description of the PCAPS lysimeter design for sampling soil water flux is given in Chapter Two. The procedures for PCAPS generated sample processing and calculating flow-weighted averages are also provided.

3.3.3 Tracer Application

A Br$^-$ tracer was applied on December 12, 2000. The fall planted common vetch / triticale mix cover crop was already well established on H plots. In order to remain consistent with previous studies, a 29.6 g/L Br$^-$ solution was used for the tracer application (Hess, 1995). The total amount of Br$^-$ applied over the 31 x 84.5 cm^2 PCAPS surface dimensions was 3877 mg. The tracer solution was prepared by dissolving KBr in water. The tracer solution was pressurized to 414 kPa using CO_2 and applied to the field using a three meter wide manually mobilized sprayer mounted to a bicycle wheel. A 0.5 mm depth of the solution was sprayed above each pair of PCAPS from a height of one meter. The total spray area was 6m x 7.5m. Figure 3.1 shows the spray pattern for the Br$^-$ application.
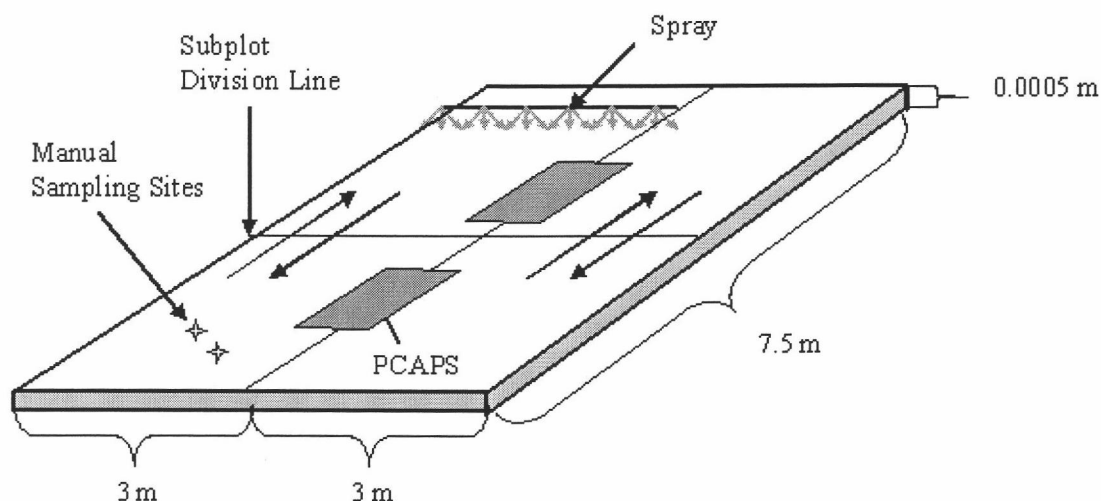
Figure 3.1 Spray pattern for applying 2000 Br⁻ tracer. Previous tracer applications were only three meters wide and were centered on the longitudinal axis of the PCAPS. The general location of 2001 manual soil sampling sites are also shown.

3.3.4 Soil Sampling

Following the Dec 12th, 2001 application of the Br⁻ tracer at the NWREC, manual soil sampling was used to observe spatial concentrations of the Br⁻ plume. Dates for the three sampling events were the 5th and 6th of May 2001, 5th-7th of September 2001, and the 15th of December 2001, respectively. For each sampling event, two holes were augured (1 ¼" diameter) on each of the subplots (N0, N1, N2) equipped with working PCAPS. All augured holes were well within the area of tracer application and about 1.5 meters from the buried samplers (Figure 3.1). It was imperative to sample far enough from the PCAPS to prevent any hydrologic changes that could result from destructive sampling. Each of the 40 holes was augured to 1.2 meters, the same depth as the flux plane of the PCAPS collection wicks. Each augured hole was divided into six layers, each 20 cm deep. All layers from the soil profile were separately pulled to the surface, removed from the auger, placed into heavy duty freezer bags and stored in a cooler containing dry ice to stop microbiological action in the soil. Holes were back filled with bentonite to ensure that the remaining auger hole did not serve as a conduit for preferential flow. Samples were stored frozen until they could be analyzed.

The last two sampling events were identical to the first except that each 1.2 meter hole was divided into twelve layers (instead of six), each 10 cm deep. This change was made so that the Br$^-$ plume and resident concentrations of NO$_3^-$- N within the soil water could be determined with better resolution. The upper 30 cm were not represented during the September sampling event due to large quantities of decaying bean crop left from the recent mowing and a dry, uneven soil surface that was deemed difficult to characterize using an auger sampling tool. Samples were not taken from fallow plots containing PCAPS 21 and 22 in both September and December. In September the soil was too compacted to sample, while in December the area was ponded with water.

3.3.5 Soil Extractions and Processing

After thawing, soil samples were manually homogenized within the bag. Bromide and NO$_3^-$- N concentrations may be variable over small distances, so this mixing step was important to ensure that each sample removed from each layer for extraction was representative of the entire layer. A sub sample was taken from each of the bags to yield approximately 20 grams of oven dry soil. Wet weights were recorded, and samples were placed in tin drying pans and oven dried at 105 °C overnight. Dry weight was measured to calculate gravimetric water content. Soil Br$^-$ and NO$_3^-$ were extracted from each of the oven dried sub samples using the method used by Dick and Tabatabai (1979). Though Br$^-$ and NO$_3^-$ are negatively charged and readily soluble in water, the extracting solution contained 100 mg L^{-1} P as (H$_2$PO$_4$)$_2$ to ensure that all ion binding sites were saturated and that the ions of analytical interest were dissolved in solution. Samples were mechanically shaken in a 250 ml Nalgene® container for one hour. After shaking, samples were allowed to settle for five minutes. The supernatant was gravity filtered using Whatman No. 42 filter paper. The clear extract was stored frozen until it could be analyzed with a Dionex AS4A-SC separator column and an AG4A-SC guard column (Dionex Corp. Sunnyvale, CA) for Br$^-$ and NO$_3^-$ content. Concentrations of Br$^-$ and NO$_3^-$- N within the extracted solution were used to back calculate the concentrations of these ions in the soil water and to determine the percent of original applied tracer mass recovered.

To compute the total mass recovery, the average Br⁻ content from each 20 cm depth unit of the augured soil sample was converted to a mass Br⁻ / mass soil basis. This ratio was applied to the soil mass in the corresponding soil volume below the PCAPS. The soil mass in each of the volumes below the PCAPS was estimated using the bulk density measurements with depth and the volume of soil contained in a cube with a length and width equal to the dimensions of the PCAPS surface panel and a depth of 10 or 20 cm as defined by the soil sampling depth units.

Bromide collection efficiency data were tested by analysis of variance (ANOVA) with a general linear model as a randomized complete split-plot block design over tracer application years with winter cover crop treatment as main plots and N rate as subplots. A separate ANOVA analysis was also made for each tracer application year.

3.4 Results

3.4.1 Br⁻ Tracer Breakthrough and PCAPS Mass Recovery

Rainfall characteristics during the period following the application of the Br⁻ tracer are given in Figure 3.2. Water year 2001 (measured from October 2000 to September 2001), received only 58% of the average rainfall. Winter 2001 was somewhat wetter than average with 108% of the typical water year accumulation. Winter 2002 received 100% of the average at the time of this analysis with four months yet to be measured in water year 2003. Several rain events during December and January of water year 2002 resulted in saturated conditions and overland flow on some of the plots. Though matrix flow was the dominant flow regime, preferential flow existed at the site and was an important process in chemical transport and dispersion (Chapter Four).

An average concentration was fit to the flow-weighted average Br⁻ concentration calculated from the 14 samplers under C plots and the 12 samplers under H plots (Figure 3.3b). The best fit line was determined using Microsoft Excel's solver package to minimize the sum of squared differences between measured data and the Advection Dispersion Equation (ADE) modeled solution. This analysis of tracer breakthrough is
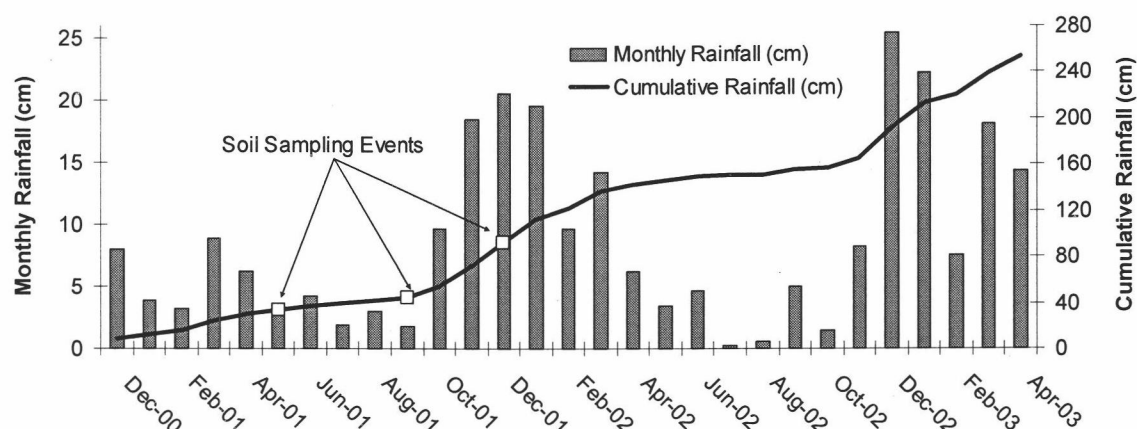
Figure 3.2 Precipitation characteristics during the period of Br- tracer breakthrough. Monthly rainfall totals are given from December 2000 through April 2003. The three soil sampling events are shown.

qualitatively based; a quantitative description of transport parameters generated using the ADE is provided in Chapter Four.

Figure 3.3b shows that the real and fitted concentrations were higher below fallow plots than cover cropped plots. PCAPS measured peak Br$^-$ concentrations were 14.3 vs. 23.1 mg L^{-1} and fitted peak concentrations were 10.9 vs. 14.7 mg L^{-1} for H and C plots, respectively. Figure 3.3a shows the cumulative mass recovery of Br$^-$ collected by PCAPS after 2.8 pore volumes of water were collected. Overall average mass recovery in percent for the 2000 Br$^-$ application was 41%. Individual samplers measured recovery efficiencies that ranged between 56 and 18 % (Figure 3.4). Mass recoveries were 35 and 47% for the H and C plots, respectively. Analysis of variance showed that cover crop treatment had a significant effect (P-value = 0.02) on Br$^-$ collection efficiency (Table 3.2). Though crop analysis was not made to determine the Br$^-$ content within the biomass, uptake of Br$^-$ by the common vetch / triticale mix cover crop is the most probable explanation for the observed differences in tracer recovery.

Using a water balance, it was possible to find the water collection efficiency for PCAPS. In Chapter Two, an average collection efficiency of 76% was calculated over three seasons spanning the eleven years of PCAPS operation. Brandi-Dohrn et al. (1996) also found an average water collection efficiency of 76% over the first 566 days of
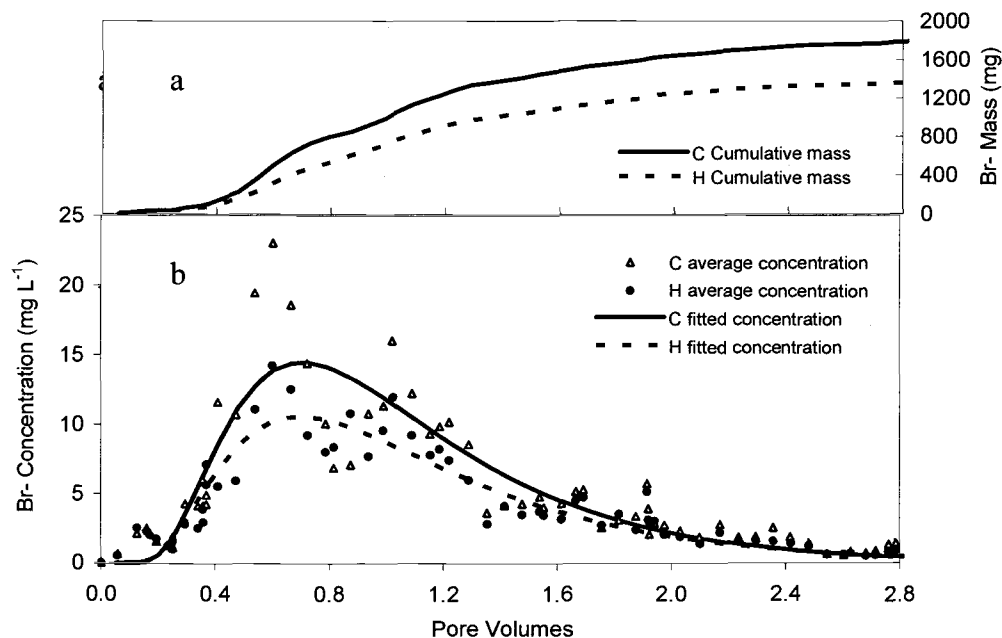
Figure 3.3 Difference in cumulative Br- mass collected by PCAPS from 2000 Br-application for conventional (C) and cover cropped (H) plots (a). Characteristic breakthrough curves for C and H plots (b). Differences in Br- collection efficiency were attributed to Br- uptake by cover crops.

PCAPS operation at the same site. Assuming 76% collection efficiency during this period of Br⁻ recovery, average Br⁻ collection efficiencies would be increased to 46 and 61% for the H and C plots, respectively.

With time, Br⁻ tracer recovery under cover cropped plots could reach unity with fallow plots as the process of crop assimilation and mineralization will eventually relinquish Br⁻ to the soil water and leach to sampler depth. Recycling of Br⁻ through mineralization aside, it is possible that recovery ratios calculated using PCAPS may increase in the future if considerable diffusion of Br⁻ occurred into small pores of the soil matrix. This would have the effect of creating a long tail on the Br⁻ breakthrough curve (Haggerty et. al., 2000). This may be hard to measure, however, as Br⁻ concentrations drop to near the detection limit defined by the chromatograph analysis technique.
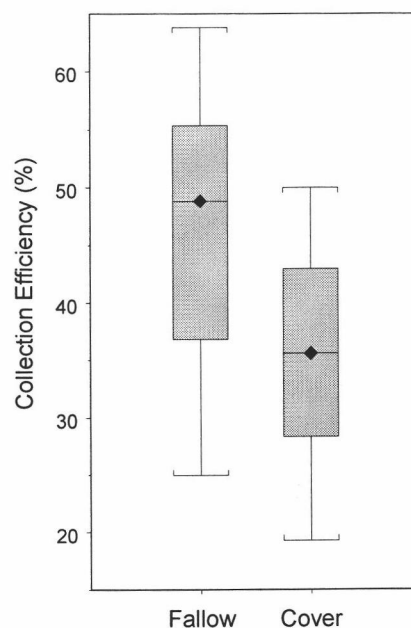
Figure 3.4 Distribution of tracer recovery efficiencies achieved using PCAPS during breakthrough of the December 2000 Br⁻ tracer application. Box and whisker plot shows a box with borders at the 25 and 75% quartiles and the statistical median in the center. The whiskers extend to the farthest points that are not outliers (i.e., that are within 3/2 times the interquartile range). A dot is shown for every point more than 3/2 times the interquartile range.

Table 3.2 Summary of analyses of variance showing the sources of effects on Br-collection efficiency by PCAPS following the December, 2000 tracer application.

Time Period	Source of Variation	Br⁻ Collection Efficiency	
		df	*P*-value
2000 Br⁻ tracer application	Cover crop	1	0.0214
	N fert. rate	2	0.4489
	Block	3	0.2140
	Cover * N fert. rate	2	0.9635

3.4.2 Cover Crop Effects on Soil NO_3^- and Br⁻ Concentration Profiles

Extraction of soil samples showed differences in chemical distribution between fallow and cover cropped treatments. Bromide and NO_3^-- N concentration profiles were drawn for at least 36 different 1.2 meter holes for the months of May and September 2001. The May sampling was one month after the incorporation of the cover crop into

the soil by disking the plots. Bromide concentration profiles for plots of similar cover crop treatment were averaged for each sampling date and are shown in Figure 3.5. For NO_3^-- N profiles, concentrations correlated with fertilizer inputs, however only N2 plots are shown (Figure 3.6). Extraction analysis of soil samples from mid-December 2001 indicated that heavy rains in November and early December moved the Br^- and NO_3^- - N pulses past the 1.2 meter level. Considerable differences were not seen between May profiles and September profiles, and can be explained by the lack of deep percolation during the dry summer months.
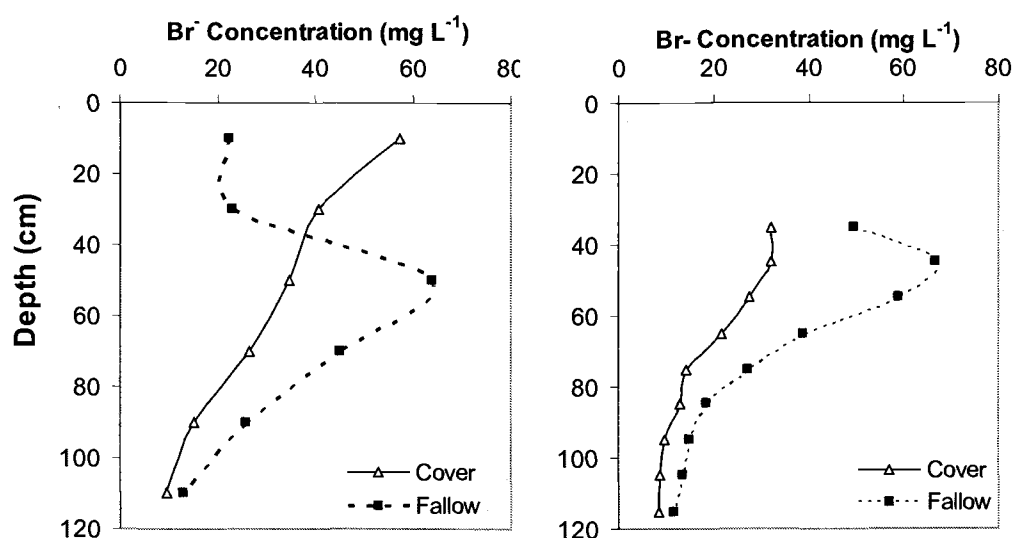
Figure 3.5 Bromide concentration in soil water during May 2001 (left) and September 2001 (right). The difference between cover cropped and fallow treatments are evidence that Br was retained by the cover crop during the winter.
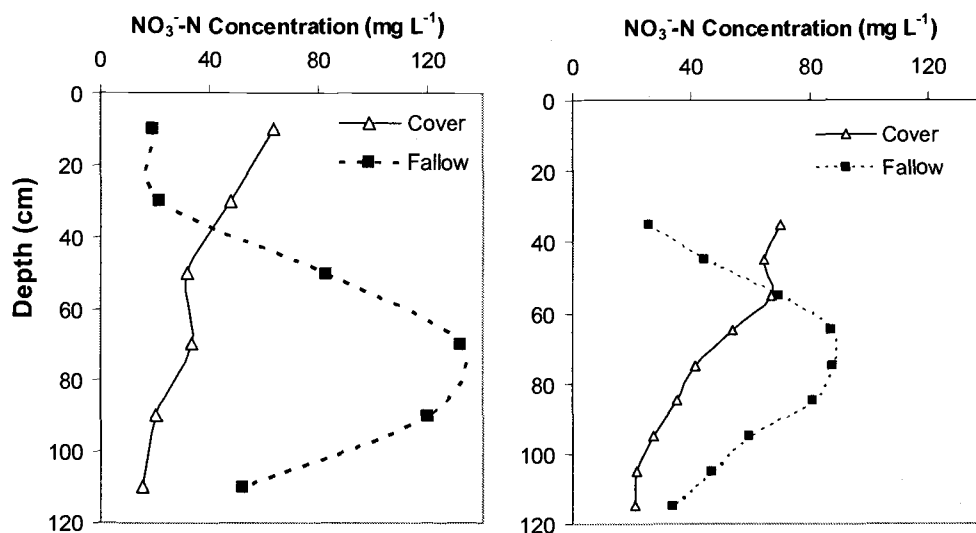
Figure 3.6 Nitrate - N concentration in soil water during May 2001 (left) and September 2001 (right). The difference between cover cropped and fallow treatments are evidence that NO_3^--N was retained by the cover crop during the winter.

The May and September destructive tracer recovery events indicated that the cover cropped fields had much higher concentrations of NO_3^- - N and Br^- near the ground surface, evidence that the common vetch triticale mix cover crop had successfully scavenged dissolved chemicals from the soil water during the winter months. It appeared that the elevated concentration levels of soluble Br^- and NO_3^-- N and at the time of sampling were attributed to redistribution of these chemicals through decomposition and mineralization of the cover crop in the soil (Kung, 1990). It is possible that a portion of the N mineralized from the common vetch came from fixed atmospheric N. Nitrogen contents in the plant and N fixing rates were not quantified; therefore it is impossible to positively identify the source of elevated N concentrations in the upper soil.

The lack of a NO_3^--N or Br^- pulse under cover cropped plots versus a clearly defined pulse peaking at around 50 cm under fallow fields supports the hypothesis that the legume cereal mix successfully scavenged N. The higher NO_3^-- N concentrations were at a depth of around 70 to 80 cm, while Br^- pulse was at a depth of 40 to 50 cm. The NO_3^-- N plume originated as post harvest NO_3^- remaining in the soil after the fall 2000 corn harvest, while the Br^- tracer was not applied until Dec 12th.

It is probable that the redistributed NO_3^- and Br^- remaining under the cover cropped plots were available for uptake by the following summer bean crop. Nitrate and Br^- in the fallow plots would be incapable of assimilation the following summer as a large proportion of their mass was below root depth.

3.4.3 Adjusted Br⁻ Mass Balance at a Point in Time

PCAPS tracer recoveries during the 2000 Br^- application were expanded to include mass recovered through soil extraction. Tracer recovery was calculated only for the May 2001 sampling event because the September 2001 event did not include the upper 30 cm of the profile. The May soil sampling recovered an average of 82% and 92% of the applied Br^- mass in the cover cropped and fallow plots, respectively. The recoveries calculated by PCAPS collection were much lower than the soil sampling mass recovery ratios. At the time of the May soil sampling, some Br^- mass had already been collected by the PCAPS at 1.2 meters of depth. The average cumulative mass of Br^- collected by PCAPS between the date of tracer application and the 4th of May (the day before soil sampling) was 71 mg and 97 mg (of the total 3877 mg applied over each PCAPS cross sectional area) for the cover cropped and fallow plots, respectively.

The PCAPS collected Br^- mass and the mass collected through soil collection and extraction were added together to calculate a total Br^- balance for the 6th of May 2001. This adjusted Br^- mass recovery was 83% and 95% for the cover cropped and fallow plots, respectively. Jaynes et al. (2001) also recorded high total recovery ratios for a Br^- tracer when masses from soil residue extractions and tile effluent losses were added. The high recovery ratios of soil extracted Br^- versus PCAPS collected Br^- can be explained by rainfall patterns. Little rain fell following tracer application during the winter of 2000, minimizing leaching and allowing the Br^- to stay in a well defined plume that was well-represented by the soil sampling design.

3.5 Conclusions

Recovery of NO_3^- and an applied Br^- tracer indicate that the common vetch / triticale mix cover crop assimilated Br^- and NO_3^- during the winter of 2000. The retention of Br^- and NO_3^- within the upper soil profile illustrated the process by which cover crops reduce leaching of applied chemicals. Statistically significant cover crop effects on Br^- collection efficiency suggest scavenging by the cover crop.

Acknowledgements

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4. Apparent Transport Parameters Determined Using Three Conservative Tracer Applications

4.1 Abstract

The dispersivity (α) component of mechanical dispersion is typically assumed to be a characteristic of a soil and independent of soil water pore velocities. Molecular diffusion (D_0) is often ignored when estimating total hydrodynamic dispersion (D_{hd}) due to the assumption that mechanical dispersion (D_m) within soil systems dominates the spreading during transport. To analyze the temporal and spatial variability of transport parameters as well as the importance of D_0 in an unsaturated soil system with naturally alternating wet and dry periods, two conservative tracers were applied at three separate times: bromide in 1992 and 2000 and chloride in 1995. Tracer breakthrough was monitored using twenty-six passive capillary samplers (PCAPS) installed under undisturbed soil at 1.2 m depth. Breakthrough curves were fit to solutions to the Advection Dispersion Equation (ADE) to calculate transport parameters. On average, α values were 3.74, 20.6, and 16.1 cm for the 1992, 1995, and 2000 applications, respectively. Dispersivity ranged from 0.02 to 78.8 cm for all individual samplers. Peak tracer concentrations reached sampler depth after 0.91, 0.60, 0.79 pore volumes corresponding to 404, 95, and 367 days for the 1992, 1995, and 2000 applications, respectively. Dimensionless retardation (R) values were 0.98, 0.74, and 0.83, and pore water velocities were 1.2×10^{-2}, 2.2×10^{-2}, and 2.0×10^{-2} cm h^{-1}, for the 1992, 1995, and 2000 applications, respectively. Frequent, large rainfalls such as those occurring during the 1995 chloride breakthrough resulted in higher pore velocities, smaller R, and higher α values, while consistent, below average rainfalls had the opposite effect (1992 bromide breakthrough). There were no significant correlations between α values and calculated pore water velocity for individual samplers. Experimental Peclet number calculation indicated that tracer transport was controlled by ped characteristic length scales and not the silt grain size of the soil. Dispersivities at a given location did not show a correlation in time. Most samplers did not consistently generate α's above or below the mean value, though samplers generating very high or

low α's were more consistent. Spatially, α values significantly fit a log-normal distribution. Observed Br$^-$ profiles generated though soil sampling and extraction could not be predicted using PCAPS calculated α values, and α values calculated spatially were lower than those measured through time using PCAPS because they did not reflect the higher dispersion caused by heavy rainfalls during the second half of the tracer breakthrough. These findings show a weakness in the current conceptual model describing D_m as dependent on a static α, and that in vadose environments diffusion cannot be dismissed in computing spreading rates of contaminant plumes. In these experiments, differences between cycles of molecular diffusion and mechanical dispersion dominated periods were more important to determined solute dispersion than were values of α.

4.2 Introduction

Estimating realistic parameter values with predictive capacity is a challenge when modeling natural systems. For solute transport models within porous media, engineering estimates of parameters are often based on easy to measure characteristics such as the hydraulic conductivity and particle size of the media. The Advective Dispersion Equation (ADE) is a common model used to simulate solute transport in porous media because it has few parameters.

To apply the ADE, estimates of hydrodynamic dispersion (D_{hd}) must be made. Hydrodynamic dispersion is the sum of molecular diffusion (D_0) and mechanical dispersion within the system. Mechanical dispersion (D_m) is the product of the dispersivity (α) and the pore water velocity (v). Dispersivity is considered to be a characteristic parameter of the soil media (Selker et al., 1999; Beven et al., 1993), and therefore should be independent of the pore water velocity (Wierenga and Van Genuchten, 1989).

Researchers have attempted to describe the relationship between α, other parameters in the ADE, and the soil environment. Studies within deep groundwater and saturated soils have shown a positive correlation between α and the scale length of tracer travel (Gelhar et al., 1983; Jaynes et al. 1988). Variations in α seem to be less understood in unsaturated conditions were measured D_{hd} and α have not been well-correlated with depth (Porro et al., 1993). Wierenga and Van Genuchten (1989) found no increase in the dispersion with depth in unsaturated water columns. A slight decrease of α with depth was found under drip and ponded irrigation (Jaynes and Rice, 1993). Persson and Berndtsson (1999) studied the effect of water application on transport parameters and found that the α decreased when water was applied more often. This relation was attributed to less variability in the pore water velocity with frequent water applications.

For unsaturated conditions, Beven et al. (1993) compiled an extensive review of dispersion coefficients, α values and pore water velocities obtained from tracer studies of various area and length scales. In general, α values have fallen between centimeters for packed column experiments to many meters in deep groundwater studies.

Numerous studies have recorded the importance of preferential flow within soil systems (Jaynes et al. 1988; Kung et al. 2000; Sigua et al. 1993; Villholth et al. 1998; Hamdi et al., 1994). Soil systems showing preferential flow exhibit much different chemical transport properties than soils showing predominately flow through the soil matrix. Studies have found that preferential flow can deliver nitrate (NO_3^-) and/or pesticides to shallow groundwater at short transport times. Hamdi et. al. (1994) recorded separate peaks for the preferential and matrix components of the tracer breakthrough. Water can follow preferential flow paths during times when water inputs exceed the infiltration capacity and saturated conditions develop. Due to the spatial heterogeneity of hydraulic conductivity (K) and infiltration (I), saturated conditions may be localized and difficult to spatially characterize. Kung et al. (2000) observed early tracer breakthrough due to preferential flow though no saturated conditions or ponding was observed.

A majority of tracer studies showing preferential flow flooded the soil surface or used irrigation systems to apply water at rates nearing the infiltration capacity of the soil and thus selected for conditions that favored preferential flow. These observations may be applicable to other irrigated sites, sites with low permeability soils, and areas with short duration, high intensity rainfall events such as in the Midwest. It is likely that these results do not represent the processes in many other regions with different rainfall patterns or well-drained soils.

Because of the high velocity and short travel times observed in studies using saturated and or continuous irrigation sets, it is plausible that flow characteristics and fitted dispersion parameters could differ from experiments relying on natural rainfall. For example, time of peak tracer breakthrough is typically shorter than the time that would be needed under natural conditions.

Due to the high pore water velocities employed in many experiments, D_m has been much greater than D_0 within the system, to the point where D_0 could be assumed to be negligible. Mathematically, $D_{hd} = D_0 + \alpha V$ is reduced to $D_{hd} = \alpha V$, where the product of αV is D_m (Van Ommen et al., 1989). However, if one is concerned with the behavior of the slowest moving solutes, such an assumption is not acceptable (Haggerty et al., 2000).

In soil systems were pore water velocities are variable, with times of slow movement and some preferential flow, diffusion into soil peds may be important (Villholth et al., 1998).

It is likely that diffusion would be important in natural systems experiencing cyclic rainfall throughout the year such as Oregon's Willamette Valley. During periods of little or no water movement solutes could move by diffusion through the smaller water-filled soil pores. The effect of such conditions would be a long tail on the tracer breakthrough and may be expected to have high fitted D_{hd} depending on the measured concentrations within the breakthrough tail relative to the concentration within the bulk tracer breakthrough (Haggerty et al., 2000).

Several studies have pointed out the log-normal distribution of spatially measured transport parameters within soil systems. Boll et al. (1997) used wick and gravity samplers to measure conservative tracer breakthrough under a variety of soil types and water application rates. Dispersion, pore water and solute velocities best fit a lognormal distribution under macropore dominated systems, but fit a normal distribution under matrix flow conditions. Under a week of continuous flooding and sampling using suction cup samplers, Jaynes et al. (1988) found that the pore water velocity and dispersion of a Br⁻ tracer best fit a lognormal distribution. Saturated conditions and preferential flow through macropores was also observed. Gupte et al., (1996) measured Br⁻ breakthrough from drainage tiles and found a lognormal distribution for measured dispersivities.

The objectives of this study were to use wick samplers and soil sampling to examine: 1) the characteristics of tracer breakthrough for three pulse applied conservative tracers under natural rainfall conditions, 2) the relation between fitted transport parameters for each of the applications under predominately matrix flow conditions where D_0 is important 3) the spatial distribution of fitted α values 4) the ability of the ADE to spatially predict tracer concentration.

4.3 Methods

4.3.1 Site Description

A complete description of the location, climate, soil characteristics and field management is given in Chapter Two. A diagram of the field design is also given. Summer vegetables were irrigated using a hand move sprinkling system. Irrigation sets took place once per week. Water was applied for variable set times according to the time of the year; however a quantitative model of evapotranspiration was not used to fine tune irrigation amounts. Irrigation procedures were consistently managed by the same person throughout the entire 11 year study.

4.3.2 Field Lab Methods

A complete description of the PCAPS lysimeter design for sampling soil water flux is given in Chapter Two. The procedures for PCAPS generated sample processing and flow weighting calculations are also provided.

Water content was measured during the summer of 2002 using a neutron probe model 503-DR Hydroprobe from Campbell Pacific Nuclear International. The unit was calibrated at nine access tube sites throughout the field, and measurements were made at 17 sites in total. Access tubes were located approximately 2.4 m from the center of the PCAPS. For the calibration, soil cores were taken at 10 cm increments down to 160 cm depth and neutron probe signal was compared with laboratory measured water contents. Soil water content was determined in the field 6 times and measurements were always made the day before irrigation events.

Conservative tracers were applied over the lysimeter plots three times during the 11 year study. The first Br tracer was applied on the 4th of November, 1992. A Cl tracer was applied on the 28th of October 1995. The final Br application took place on the 12th of December, 2000. In order to remain consistent during the study, 29.6 g/L Br and Cl solutions were used for each tracer application. The solutions were prepared by dissolving KBr or KCl in water. Each of the tracers was applied to the field using a three meter wide sprayer mounted to a bicycle wheel. The solution was pressurized using CO_2

and applied at 414 kPa. A 0.5 mm depth of the solution was sprayed above each pair of PCAPS from a height of one meter. The first two applications covered a 3 m x 7.5 m area. Due to poor tracer mass recovery obtained from the first two applications, the last Br tracer spray area was doubled to 6 m x 7.5 m. The assumption being that the larger tracer area would minimize the effect of lateral solute movement on tracer recovery. The spray pattern for the last Br application is shown in Chapter Three (Figure 3.1).

A complete description of the soil sampling and laboratory extraction procedure for Br are given in Chapter Three.

4.3.3 Advective Dispersion Equation

The advection dispersion equation (ADE) with linear adsorption / exclusion was used to model observed breakthrough curves of the three tracer applications. The differential form of the ADE for a tracer moving through the soil is:

$$\frac{1}{R}\frac{\partial C}{\partial t} = D_{hd}\frac{\partial^2 C}{\partial z^2} - \overline{v}_w\frac{\partial C}{\partial z}$$

(4.1)

Here, concentration as a function of time and space is controlled by both advection and dispersion. A one-dimensional vertical solution to the ADE for a surface application whose mass is as a delta-function at $x = 0$ (infinitely concentrated at the surface) is:

$$C\left(z,t\right)=\frac{M}{\theta AR\sqrt{4\pi D_{hd}t/R}}\left[\exp\left(-\frac{\left(z-\bar{v}_w t/R\right)^2}{4D_{hd}t/R}\right)+\exp\left(-\frac{\left(z+\bar{v}_w t/R\right)^2}{4D_{hd}t/R}\right)\right]$$ (4.2)

Where:

$C\ (z,t)$ = Concentration of tracer in soil water as a function of time and depth

M = Mass of tracer applied over sampling area

θ = Volumetric fraction of soil occupied by water, taken to be field capacity 0.344

A = PCAPS horizontal flux area

D_{hd} = Hydrodynamic Dispersion Coefficient (encompasses both molecular diffusion and dispersion)

R = Dimensionless retardation coefficient

z = Stationary point of observation (sampler depth)

$\bar{v}_w$ = Mean pore water velocity

The semi-infinite form of the model utilizing two superimposed advecting Gaussian solutions effectively accounts for the fact that there is no flux with time due to diffusion out of the soil surface into the atmosphere.

The ADE was fitted to measured tracer concentrations using Microsoft Excel's Solver package to minimize the sum of squared differences between measured data and the modeled solution. Three ADE parameters were fit to the measured data: M, mass of Br⁻ added at time zero, $D_{hd,}$ R, the retardation coefficient. It was necessary to fit M to use the ADE but the fitted value was not imperative to describe the spreading of the solute or the temporal aspects of the center of mass. It was necessary to fit R because assigning a default value of one would force all breakthroughs peaks to occur at one pore volume. A breakthrough at one pore volume would theoretically be expected with a conservative tracer in homogenously packed porous media. Observations of tracer breakthrough did not support this. Fitting an R value can help describe solute behavior in systems showing preferential flow, unstable wetting fronts, and ion exclusion.

Unlike modeling situations where conventional time units can be used, such as the case of an advecting solute in a river with constant velocity, we used time units of relative pore volumes passed. This was necessary because solute movement for each of the tracers was driven by natural precipitation. Due to unsteady dry and wet conditions, the solute remained in the soil column for long periods of time with no movement. To

calculate the depth of water equivalent to exactly one pore volume, P.V., at a depth of the PCAPS, z, we used.

$$P.V. = z\theta_f$$ (4.3)

Where:

θ_f = Field capacity or volumetric fraction of soil occupied by water, measured at 0.344.

The P.V. was a constant value of 41.3 cm throughout the experiment. A unitless parameter applicable specifically to the field site, Relative pore volumes, p.v., and was calculated using:

$$p.v. = \frac{\sum_{i=1}^{n} V/A}{P.V.}$$ (4.4)

Where:

V = Cumulative volume of water collected during sampling events
n = Number of sampling events since tracer application

Hydrodynamic dispersion (D_{hd}) is the cumulative effect of all processes causing the spread of solutes in porous media and can be expanded as follows to show the contribution of molecular diffusion (D_0) and mechanical dispersion (D_m).

$$D_{hd} = D_0 + D_m$$ (4.5)

Where:

$$D_m = \alpha \bar{v}_w$$ (4.6)

Where:

α = Dispersivity

The pore water velocity was not fitted, but was calculated using the cumulative volume of water collected during the period that breakthrough characteristics were measured for each of the tracer pulses. Pore water velocity could be calculated for any desired time increment, t (from average velocity over entire breakthrough to maximum velocity during periods of high rainfall and percolation):

$$\overline{V}_w = \frac{\left[\sum_{i=1}^{n} V/A\right]\frac{1}{\theta_f}}{t} \quad \text{Or} \quad \overline{V}_w = \frac{p.v.\,{}^{*}\,z}{t} \qquad \text{(4.7) and (4.8)}$$

Where:

n = Number of sampling events in period of interest

The sum of least squared differences method was not capable of isolating D_0 and D_m during the fitting process. The fitting process was only able to fit an effective spreading of the tracer plume and generate a value for D_{hd} by using the pore water velocity and fitting an intermediate dispersivity (α^{*}) for the system.

$$D_{hd} = \alpha^{*}\,\overline{V}_w \qquad (4.9)$$

Where:

α^{*} = Intermediate fitting parameter reflecting the dispersivity in the system

The intermediate parameter was mathematically indiscriminate of the contribution of spreading due to D_0 and D_m.

Conventional time was eliminated from the ADE using pore volumes and the total distance of the soil column. The use of pore volumes instead of conventional time provided a linear scale for the time variable required to fit the ADE to observed data.

$$D_{hd}t = \alpha^{*}\overline{V}_{w}t \tag{4.10}$$

$\overline{V}_{w}t$ = distance wetting front has traveled = $z\,p.v.$

The real dispersivity of the system (as well as the mechanical dispersion) was calculated using the diffusion coefficient for Br⁻ adjusted to account for the inherent tortuosity in the soil media. An estimate of the unitless tortuosity coefficient, k, is given in Bear (1972) is 0.4. This is nothing but an algebraic rearrangement of Equation 4.5 and 4.6.

$$\alpha = \frac{D_{hd} - \left(D_{0}^{*}k\right)}{\overline{V}_{w}} \tag{4.11}$$

D_{0}^{*} = Diffusion coefficient for Br⁻ with tortuosity constant
k = Tortuosity coefficient

4.3.4 Spatial Analysis Using the ADE

Chemical concentration is a function of time and space. Manual soil sampling provided an opportunity to test the ability of the ADE to predict the spatial chemical concentration profiles at a point in time. Transport parameters were calculated using the spatial concentration profile (spatial parameters) and compared with PCAPS calculated parameters (temporal parameters). Both the May 2001 and September 2001 sampling dates provided a concentration profile encompassing the rising and falling limb of the breakthrough curve. Therefore these dates could be used to generate unique solutions using the advection dispersion equation. The December sampling event took place during the time of or after the Br⁻ peak concentration had passed the 1.2 meter depth, therefore, the profile could not be used for analysis because it could not provide a unique set of solution parameters with only the falling limb of the tracer breakthrough.

To fit the parameters of α, initial solute mass, and retardation, the cumulative number of pore volumes passing through the 1.2 meter flux plain up to the soil sampling date had to be determined using the PCAPS. These values, used as the time variable in the ADE,

were not very different for the day 143 and day 280 (May and September) of 2001 soil sampling events as little water was collected by PCAPS between these months.

4.4 Results

4.4.1 Tracer Breakthrough and Transport Parameters Measured Using Stationary PCAPS

Tracer breakthrough was measured using each of the 26 PCAPS for each of the three tracer applications. Bromide concentration was analyzed following the 1992 Br⁻ application for 47 sampling events between 4/12/1992 and 1/11/1994. Chloride concentration was analyzed following the 1995 application for 43 sampling events between 4/11/1995 and 20/4/1997. The final Br⁻ application of 2000 was analyzed over 73 events between 19/12/2000 and 26/4/2003. Rainfall patterns during the time following the application of each of the tracers varied with respect to monthly and total accumulation (Figure 4.1). Months recording high precipitation had more periods of intense rain than low rainfall months.
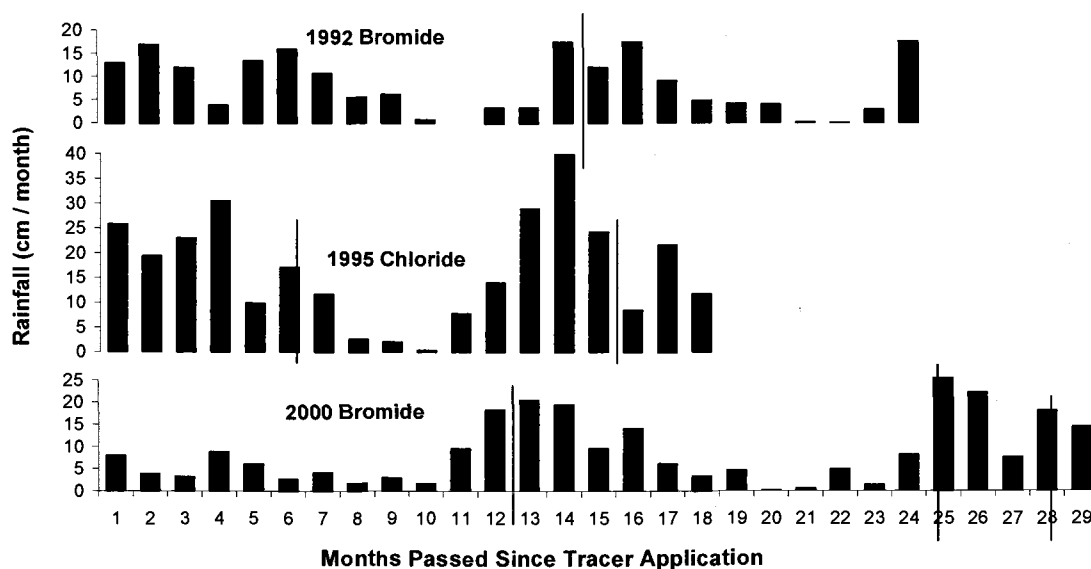
Figure 4.1 Monthly precipitation following each of the tracer applications. Monthly rainfall totals are given from November 1992 through October 1994, November 1995 through April 1997, and December 2000 through April 2003 for the 1992, 1995 and 2000 tracer applications, respectively. Vertical lines show points in time when integer numbers of pore volumes were collected by PCAPS.

The neutron probe provided water content with depth throughout the 2002 growing season to early fall (Figure 4.2). Measurements were made before weekly irrigation sets. The soil column was driest at the surface and wettest at 100 to 140 cm. Water content observations did not rule out the possibility of short periods of upward water movement caused by tension forces. Upward water gradients would make solute travel distances greater, allowing for more solute spread and longer travel times. Possible upward water gradients do not interfere with the study goal to characterize α in field soils, as all plots were comparable under similar water contents.
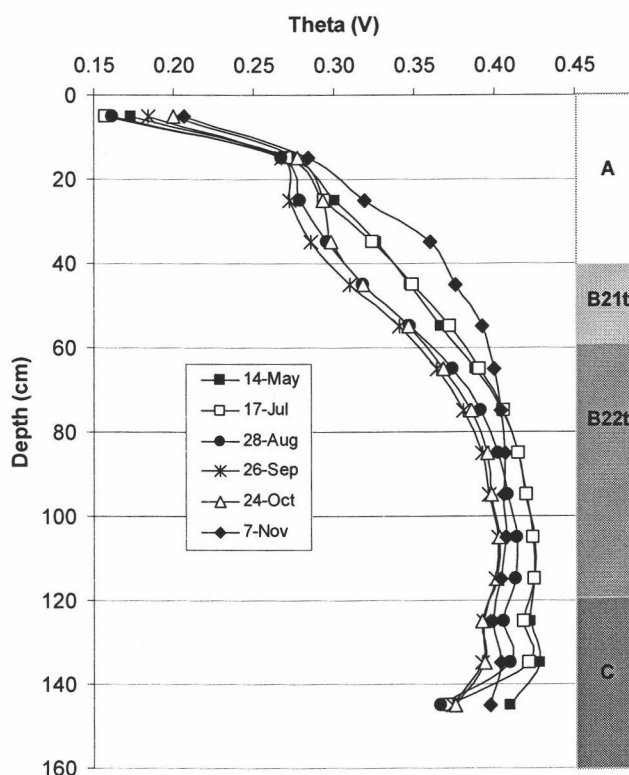
Figure 4.2 Average soil water content with depth calculated from 17 locations during summer months as determined by neutron probe. Soil horizons with depth are shown at right (Data from Y. Hagimoto, 2003, personal communication).

The ADE was fit to all 78 breakthrough curves generated during the study. For each of the tracer applications, a flow-weighted average concentration for all 26 samplers was calculated and plotted against the average number of pore volumes collected. This provided an average breakthrough curve for each of the tracers (Figure 4.3 a,b, and c).

Differences were observed between the timing and shape of the three average breakthrough curves. The average breakthrough generated from the 1992 Br⁻ application had the latest time of peak breakthrough and its shape had little skew. For breakthrough curves measured through times at a point in space, the magnitude of skew is proportional to the amount of hydrodynamic dispersion in the system. The 1995 Cl⁻ application resulted in the fastest time to peak breakthrough and had the most positively skewed fitted concentration curve, indicating that the plume was the most dispersed. The 2000 Br⁻ application average breakthrough curve was skewed slightly less than the 1995 tracer application.

Average α values for the three tracer applications, calculated from the flow-weighted average breakthrough curves, were 3.7 cm, 20.6 cm and 16.1 cm for the 1992, 1995, and 2000 events, respectively. Dispersivity values for all tracer applications ranged between 0.02 cm and 79 cm. The distribution of fitted dispersivities for each tracer application is shown in a boxplot (Figure 4.4).

Peak tracer concentrations reached sampler depth after 0.91, 0.6, 0.79 pore volumes or 404, 95, and 367 days for the 1992, 1995, and 2000 events, respectively. Average retardation (R) values, defined as the ratio of the water velocity over the solute velocity, were 0.98, 0.74, and 0.83 for the 1992, 1995, and 2000 events, respectively. Pore water velocities were 1.2×10^{-2}, 2.2×10^{-2}, and 2.0×10^{-2} cm h⁻¹, for the 1992, 1995, and 2000 applications, respectively. Average transport parameters and their ranges for each of the tracer applications are summarized in Table 4.1, and include the parameters calculated through soil sampling discussed further in the text.
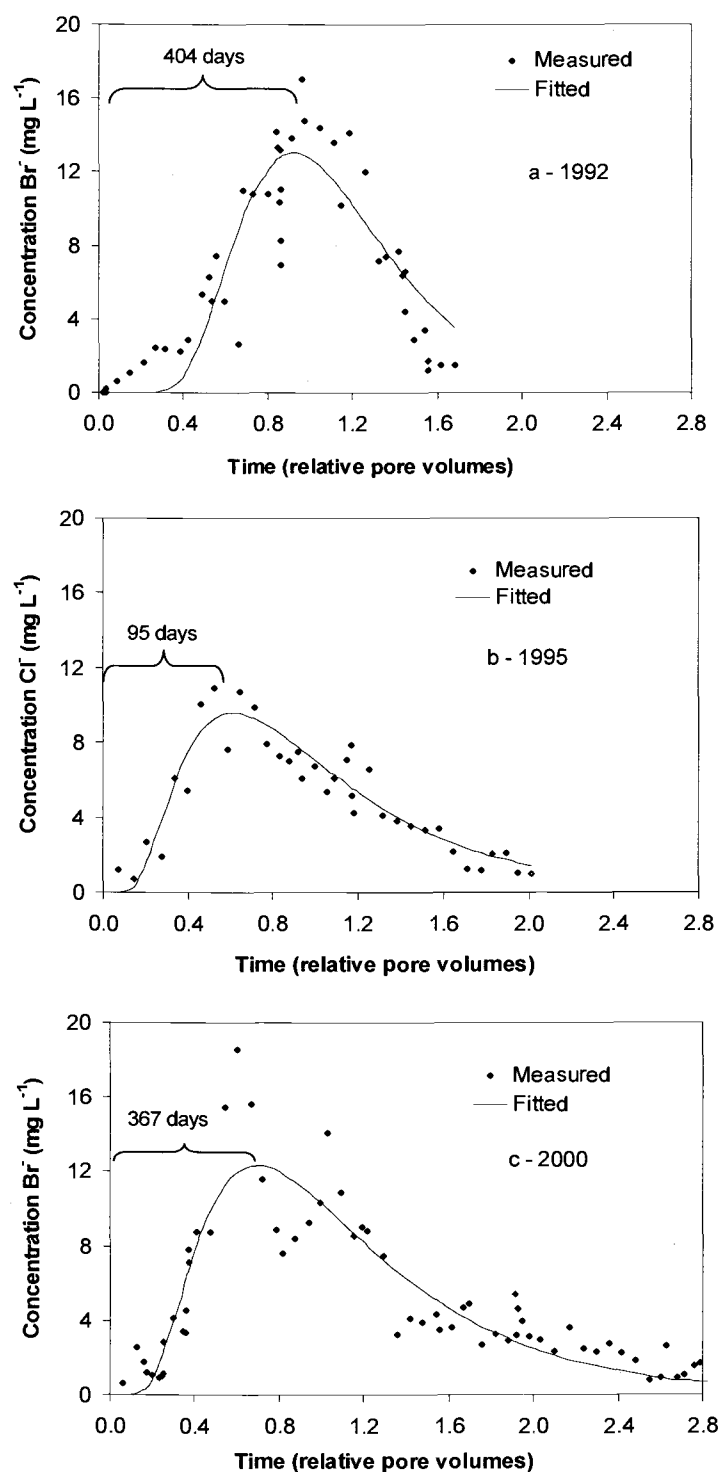
Figure 4.3 Average breakthrough curve for each of the tracer applications. The 1992 Br,
1995 Cl and 2000 Br applications are represented by graph a, b, and c, respectively.
Time in days to peak concentration is given.
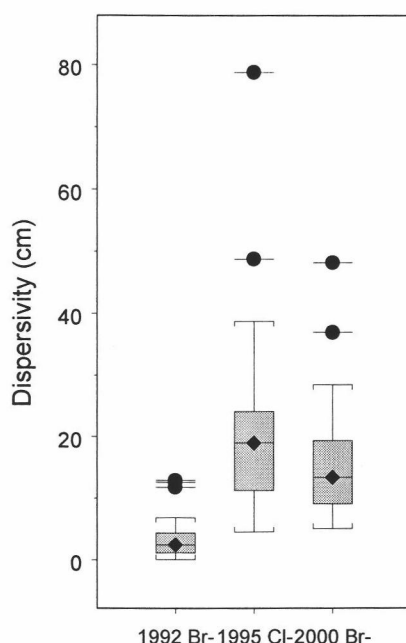
Figure 4.4 Distribution of α values calculated for each of the tracer applications. Dispersivity values varied for each tracer application. Box and whisker plot shows a box with borders at the 25 and 75% quartiles and the statistical median in the center. The whiskers extend to the farthest points that are not outliers (i.e., that are within 3/2 times the interquartile range). A dot is shown for every point more than 3/2 times the interquartile range.

Average mass recovery in percent for the 1992 Br⁻, 1995 Cl⁻, and 2000 Br⁻ applications were 32, 45, and 41% respectively. Differences in Br⁻ collection efficiencies for the 2000 tracer application below fallow and cover cropped plots were 47 and 35 %, respectively. Winter cover crops had a significant effect on collection efficiency at $P = 0.021$. The difference may have been due to scavenging and retention by the winter cover crop (Chapter Three). Cover crop treatment did not have a significant effect on α values (Table 4.2). Though cover crops removed 12 % of the Br⁻ mass relative to the fallow plots, the attenuated Br⁻ pulse under cover cropped plots was still able to generate transport parameters comparable to fallow plots.

Table 4.1 Average and range of transport parameters for each conservative tracer application. Parameters fitted from tracer breakthrough measured through time by PCAPS lysimeters and spatially by soil sampling are shown.

Statistical Group	Summary of Transport Parameters			
	D_m (cm^2/h)	α (cm)	V (cm h^{-1})	R
1992 Br⁻ Application	0.045 (2.29 x 10^{-4}, 0.164)	3.74 (0.024, 12.9)	0.012 (0.007, 0.014)	0.98 (0.60, 1.24)
1995 Cl⁻ Application	0.455 (0.073, 1.84)	20.6 (4.48, 78.8)	0.022 (0.015, 0.026)	0.74 (0.23, 1.33)
2000 Br⁻ Application	0.308 (0.089, 0.747)	16.1 (5.04, 48.1)	0.020 (0.011, 0.027)	0.84 (0.51, 1.32)
2000 Br⁻ Application * May 2001 Soil Sampling	0.025 (0.002, 0.077)	2.29 (0.123, 6.79)	0.012 (0.006, 0.015)	0.75 (0.44, 0.99)
2000 Br⁻ Application * May 2001 Soil Sampling	0.01 (-0.15, 0.109)	1.55 (-2.13, 13.5)	0.006 (0.003, 0.008)	0.79 (0.40, 1.17)

* Calculated only with holes augered from fallow plots.

Table 4.2 Summary of analyses of variance showing the sources of effects on fitted α values for all three applications grouped together and the 2000 Br- application alone. Cover crop treatment did not have a significant effect on α though it did have a significant effect on bromide collection efficiency (Chapter 3).

Time Period	Source of Variation	Dispersivity (α)	
		df	P-value
1992, 1995, and 2000 tracer applications	Year	2	< 0.0001
	Cover crop	1	0.5847
	N fert. rate	2	0.6551
	Block	3	0.2000
	Year * Cover crop	7	0.6312
	Cover * N fert. rate	2	0.3784
	Year * N fert. rate	14	0.3648
2000 tracer application	Cover crop	1	0.2765
	N fert. rate	2	0.7815
	Block	3	0.4587
	Cover * N fert. rate	2	0.8239

4.4.2 Relationship of Rainfall, Pore Water Velocity, and Molecular Diffusion on Tracer Breakthrough

Tracer application year was the only variable having a significant effect (P-value < 0.0001) on α over the course of the three tracer applications. ANOVA results are given in Table 4.2. The PCAPS used for these measurements were stationary in space and management on the vegetable plots was constant throughout the study. It is unlikely that soil structure below the plow layer changed significantly in the relatively short 11 year study period. By deduction, rainfall patterns are the only remaining variable that could have affected the difference in measured transport parameters between years.

Many studies observe early tracer peak concentrations that are contributed to preferential flow paths in the soil. This soil hydrologic system appeared to be dominated by matrix flow, although evidence of preferential flow existed at the site. Steady rising and falling concentrations were observed for most breakthroughs. Observations of early-arriving highly concentrated tracer spikes were not widespread for the 78 breakthroughs.

Despite the predominance of matrix flow, preferential flow was an important flow regime at the site. Observations from individual samplers during the 1995 Cl⁻ application breakthrough showed early arriving tracer. During the rainy December of winter 2001, several intense storms caused ponding on the soil surface. In seven days, from the 11th to the 18th of December, almost 8 cm of rain fell. Water was sampled from PCAPS four times during this period. Analysis of Br⁻ concentration revealed peaks and troughs in the breakthrough curve between 0.6 and 1.2 pore volumes passed. Before the storm events, Br⁻ concentrations were beginning to reach their fitted peak concentration of about 16 mg L⁻¹ (Figure 4.4c). Saturated conditions subsequently lowered Br⁻ concentrations to about eight mg L⁻¹ as the ion was washed from locations near the preferential flow paths and dilution ratios were raised (Villholth et al. 1998). After the intense rain events, Br⁻ concentrations returned to about 14 mg L⁻¹ as matrix flow once again dominated.

Providing other evidence of preferential flow, during near freezing mid-winter rain events in 2001, some samplers supplied "new" water that was considerably colder to the touch than water pumped from other samplers during the same event. Typically, temperatures in December and January are around 4°C (40°F) while soil temperatures are

temperatures in December and January are around 4°C (40°F) while soil temperatures are higher, so water using complete soil matrix flow paths would be expected to have temperatures moderated by higher soil temperatures before reaching sampler depth.

Pore water velocity values were compared with calculated α values to reveal possible correlations. A linear correlation (r-value 0.99) existed between the average pore water velocities calculated for the 1992, 1995 and 2000 tracer applications and their corresponding α values. There was no correlation, however, between pore water velocities and α values when individual sampler-generated values were compared. Since the variable α values calculated for each tracer application could not be explained by average pore water velocity, these results suggest that α values are more likely a function of cumulative monthly rainfall, rainfall timing and rainfall intensity. Average α had only an indirect correlation to pore water velocity due to the associative relationship of these parameters.

Two breakthrough curves measured by individual PCAPS during the 1992 and 1995 application breakthroughs are shown in Figure 4.5 a and b. These breakthroughs illustrate the general effect rainfall patterns had on tracer transport. Figure 4.5a shows the measured breakthrough for sampler number 17 from the 1992 Br⁻ application. The breakthrough had showed very little hydrodynamic dispersion (represented by skew) and generated an α value of 0.02 cm when D_0 was considered (Equation 4.11). Though this breakthrough was an extreme example, all of the breakthroughs from the 1992 Br⁻ application showed little D_m and a greater relative contribution of diffusion. Referring to rainfall characteristics in Figure 4.1, there was unusually low rainfall during the winter of application and during the second winter. Though short-term intensity values were not available for this period, average pore water velocities were considerably lower and less variable than values calculated for the other tracer periods. Much of the rainfall during the second year of the 1992 application served only to replenish the soil water storage reservoir in the column and thus did not contribute greatly to Br⁻ movement.
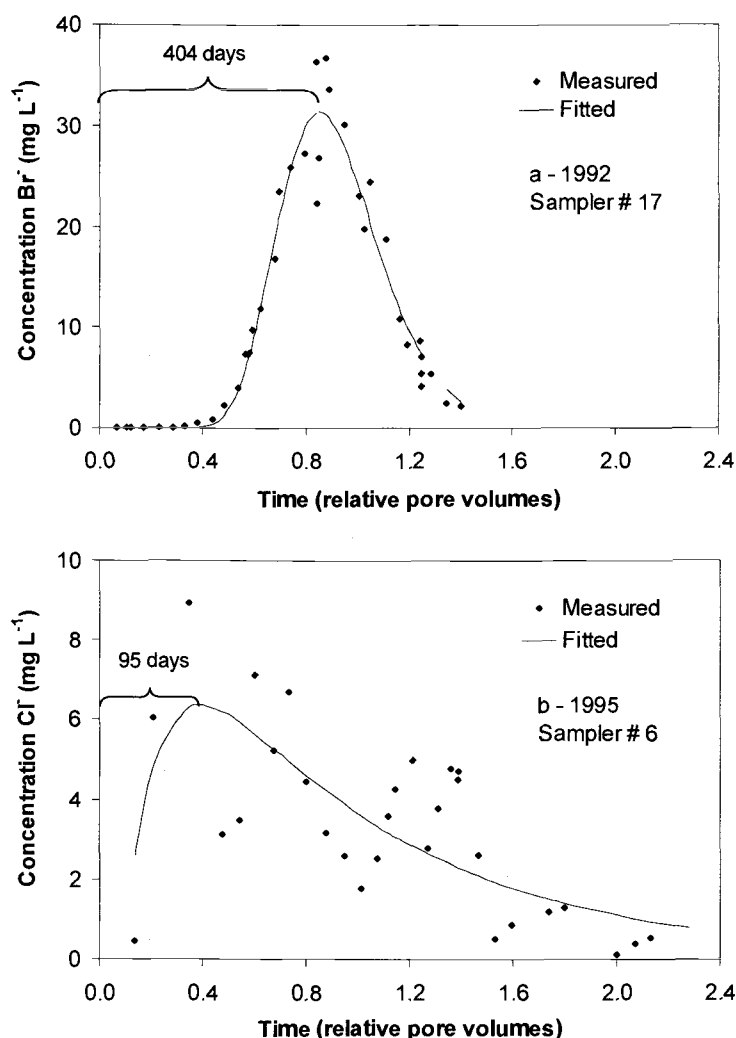
Figure 4.5 Two extreme breakthrough curves generated from individual samplers. Time in days to peak concentration is given. Figure a shows the breakthrough from the number 17 sampler during the 1992 Br- tracer collection. This breakthrough had a α value of 0.02 cm. Sampler number six of the 1995 Cl- application measured the breakthrough shown in Figure b. This breakthrough had an α value of 49.9 cm and had a long tail with measurable Cl- after 2.0 pore volumes passed.

The highly dispersed breakthrough measured by sampler number 6 during the 1995 Cl⁻ application is shown in Figure 4.5b. It is clear that heavy rainfall, especially soon after tracer application, had a large effect on breakthrough shape and fitted transport parameters. This highly skewed breakthrough had a fitted α value of 49.9 cm and was a result of the heavy rainfalls (159% and 170% of the thirty year average during water

years 1996 and 1997, respectively) received immediately after tracer application and during the following year. Highly concentrated Br⁻ was collected at 0.2 to 0.4 pore volumes passed, showing evidence of preferential flow. Average pore water velocities were the highest for this tracer application. Periods of preferential flow between periods of diffusion resulted in large dispersion values. Although peak concentrations occurred at 0.6 pore volumes, Cl⁻ continued to be collected at detectable concentrations even after 2.0 pore volumes had been collected.

Dispersivity values from the 2000 Br⁻ application fell between the 1992 and 1995 tracer applications. Very little rain fell during the first winter following the Br⁻ application; however, the second winter received above average rain and included some intense rain events that caused some preferential flow. Evidence from soil sampling indicated that little Br⁻ dispersion occurred during the first winter, yet the second winter succeeded in spreading the Br⁻ and resulted in an average fitted α value of 16.1 cm.

4.4.3 Peclet Number Analysis

The Peclet number provided a quantitative method to assess the relative contributions of D_m and D_0 to overall observed tracer spread (D_{hd}). The theoretical Peclet number is calculated as the ratio between the magnitude of the average pore water velocity, v_w, times a characteristic length, d, expressed at the mean grain size, to the D_0 coefficient (Bear, 1972).

$$Pe = \frac{\bar{v}_w d}{D_0}$$

(4.12)

The characteristic length may be expanded to include other scalars, for example, the ped size of structured soils (Bear, 1972).

Bear (1972) showed the relationship between D_{hd} and D_0 as a function of the Peclet number and defined the graphical relationship by dividing the curve into several zones. As the zone number increased from one through five, the importance of D_0 diminished. In the vadose zone, conditions usually fall into zone two to three (Selker, 1999) and D_0 and D_m are both important. Peclet number in zone two to three are between 0.4 and ten.

For this Peclet analysis, as in the calculation of the true α (Equation 4.11), the rate of D_0 for Br$^-$ was adjusted for tortuosity by multiplying the published value for D_0 in pure water by 0.4 (Bear, 1972). This resulted in a D_0 value of 2.88 x 10^{-2} cm^2 hr^{-1}. Using this rate of D_0, a grain size for silt of 0.001 cm, and an average pore water velocity calculated for the 2000 Br$^-$ application of 0.02 cm hr^{-1}, a Peclet number of 6.9 x 10^{-4} was calculated. This small Peclet number indicated that all spreading of the tracer plume should be attributed to D_0. Indeed, D_0 was an important process for solute spreading, especially during times of low pore water velocity such as in the summer when the tracer was below the depth of irrigation inputs.

It is unreasonable to believe that D_0 is the dominant process at all times throughout the year. During the winter, pore water velocities were the greatest and there was evidence of preferential flow during large rain events. The Peclet number increased to 5.6 x 10^{-3} during short periods (three to four days) when pore water velocities were higher due to large rain events. This Peclet number still remained too small for D_m to be an important spreading process. It appears that these results do not seem to well-describe solute spreading in this soil system, at least when we consider the fundamental silt grain size.

Below the plow layer, ped structures increase the characteristic diameter controlling solute spreading. If one assumes that the fundamental structure of the soil consists of peds with dimensions on the order of centimeters, Peclet numbers reach the magnitude where D_m is important. Using the average pore water velocity and a ped size of two cm or five cm gives Peclet numbers of 1.4 and 3.5, respectively.

Hydrodynamic dispersion (D_{hd}) was fit during the least squares fitting process used to find α. Bear (1972) showed that an experimental Peclet number could be calculated as shown in Equation 4.12.

$$Pe = \frac{D_{hd}}{D_0} \qquad (4.13)$$

Experimental Peclet numbers from each of the tracer applications are shown in Figure 4.6. From this analysis we see that Experimental Peclet numbers were much higher than the theoretical Peclet number calculated with the average pore water velocity and the silt grain size diameter. It appears that tracer spread in this soil system was indeed determined by characteristic length scales on the order of cm's and not by the fundamental silt grain size. With the greater range of Peclet numbers calculated experimentally, solute spread would result from both D_0 and D_m processes.
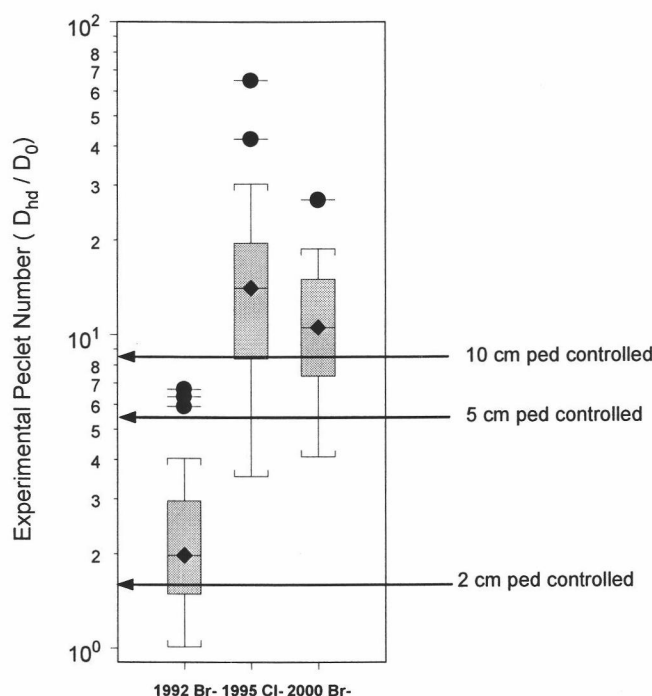
Figure 4.6 Range of experimental Peclet numbers calculated from 78 tracer breakthrough curves. Arrows point to the Peclet number that would result from water moving at the average velocity measured during the 2000 Br$^-$ tracer breakthrough with the posted soil structure diameters controlling solute transport.

4.4.4 Spatial and Temporal Distribution of Fitted Dispersivity Values

Spatial and temporal variability of α within porous media was analyzed using the 78 fitted breakthrough curves. In order to make each set of 26 dispersivities fitted from each tracer application comparable, the α values were normalized by dividing each sampler's

fitted α by the mean α calculated from all 26 samplers during each tracer event. Normalizing removed the effect of environmental conditions such as tillage, rainfall, and pore water velocity on α values. Normalizing transformed all dispersivities to a distribution with a mean of one.

Dispersivities measured by individual samplers were compared to reveal whether values remained constant over time in relation to the other samplers. It is traditionally supposed that the α of porous media is a property of the soil and therefore should not change in time in absence of physical disturbance (Selker, 1999; Beven et al., 1993). First, α data sets for each of the tracer applications were paired by sampler number and the correlation coefficient between annual values was calculated. The dispersivities of the three tracers showed no correlation with correlation coefficients of -0.003, 0.020, and 0.175 for comparison of Br 1992 vs Cl 1995, Br 1992 vs Br 2000, Cl 1995 vs Br 2000, respectively.

It appeared that areas of the field measuring extremely low or high α values may have exhibited more consistent α values. Sampler 17 had the lowest α for all of the tracer applications. The number 6 sampler had the highest α during the 1995 Cl⁻ application and the second highest α during the 2000 Br⁻ application, but measured an average α during the 1992 Br⁻ application. These observations were likely attributed to soil structure. The consistently low dispersivities measured by sampler 17 for all applications could be due to a well packed homogenous soil with little secondary structure to promote preferential flow. Flow through soils above sampler 17 would be expected to be entirely through the soil matrix because they showed very little chemical spread in time. Figure 4.5a shows the 1992 Br⁻ tracer breakthrough from sampler 17, which corresponded well to these expectations. In contrast, higher dispersivities similar to those fitted by sampler number 6 from the 1995 Cl⁻ application (Figure 4.5b) could be due to some established preferential flow paths within the soil.

Statistical hypothesis t-tests were another method to identify any temporal correlation between fixed position samplers. Fitted dispersivities from the three tracer pulses were compared by determining whether a particular location consistently fit dispersivities above or below the mean. The null hypothesis was that the sample of three normalized α

values calculated by each PCAPS came from a population with a mean value of one. The alternative hypothesis was that the α values came from a population with a different mean, either lower or higher (two-tailed test). *P*-values were determined with n-2 degrees of freedom. At a probability level of 0.1, only six samplers significantly came from a population with a mean not equal to one. Five of these were from a population with a mean normalized α lower than one, while only one came from a population with a significantly higher mean α. From this analysis it would appear that locations of rapid plume spreading are more transient than low-spreading sites.

To understand the spatial distribution of α values throughout the study, fitted α values were statistically tested to find whether the observations fit a normal or a log-normal distribution. Normalized α values from each sampling event as well as the pooled 78 dispersivities from all three tracer events were plotted in histograms. The distribution of all 78 normalized dispersivities revealed a distribution that appeared log-normal. Figure 4.7 shows the distribution of the log of the 78 normalized α values, which on first inspection appears roughly bell shaped.
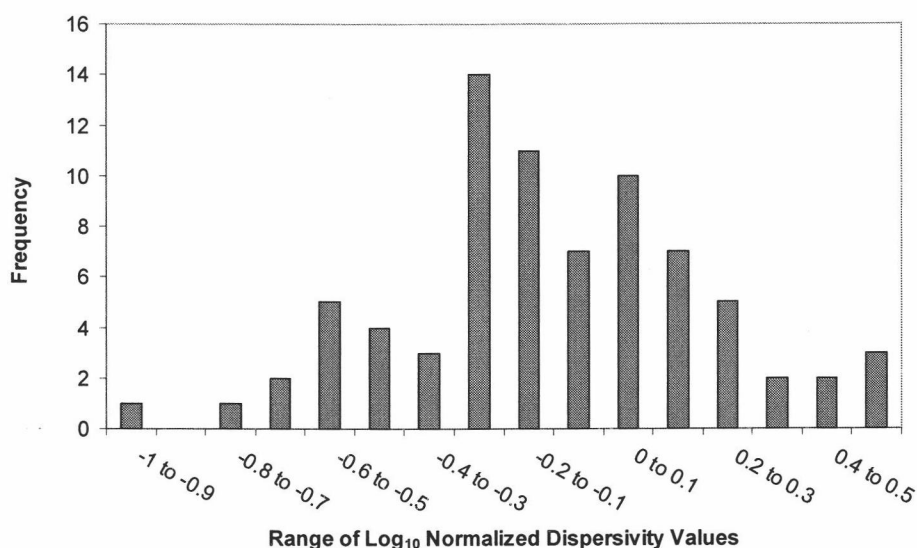
Figure 4.7 Histogram of log10 α values from all three tracer applications (α normalized by dividing by the mean value for each application).

The Filliben test statistically compared the measured distribution to a theoretical normal distribution (Filliben, 1975). The Filliben test standardizes a set of numbers to enable their distribution to be compared with that of a Gaussian distribution. This process uses the distribution about the median of a set of numbers to find a normal probability plot correlation coefficient, r, which can then be assigned a level of significance according to the error level chosen. To use the Filliben test, the Log_{10} values of the normalized dispersivities were calculated. For this Filliben analysis, it was necessary to omit the value calculated by the number 17 sampler from the 1992 application. The α value of this sampler was too small to include in the test for normality because the Log_{10} value of this near zero number had a very large absolute value. The Filliben test revealed that the Log_{10} values of the normalized dispersivities fit a normal distribution. The test was applied to each set of 26 values generated during each tracer application as well as the 78 pooled α values (Table 4.3).

Table 4.3 Normal probability plot correlation coefficient values for Filliben test for normal or log-normal distribution.

Statistical Group	Normal Probability Plot Coefficient		
	Number	Correlation Coefficient	Significance at 0.05 error
1992 Br Application adjusted α values	26	0.883	NO
1995 Cl Application adjusted α values	26	0.869	NO
2000 Br Application adjusted α values	26	0.913	NO
Entire pool adjusted α values	78	0.902	NO
1992 Br Application Log_{10} adjusted α values	25*	0.983	YES
1995 Cl Application Log_{10} adjusted α values	26	0.986	YES
2000 Br Application Log_{10} adjusted α values	26	0.987	YES
Entire pool adjusted Log_{10} α values	77*	0.995	YES

* # 17 sampler not included in distribution. With #17 included, the 1992 and pooled values did not fit a normal distribution at the 5% α level.

4.4.5 Calculation of Transport Parameters Using Manual Soil Sampling

Manual soil sampling in May and September 2001 provided a "snapshot" in time of the Br⁻ concentration in soil water throughout the upper 1.2 meters of the soil profile. The temporal transport parameters calculated by the PCAPS at a fixed depth through time were applied to the ADE to predict the Br⁻ profiles measured through soil sampling. Prediction of true soil Br⁻ concentration using temporal parameters could only be made on fallow plots due to Br⁻ redistribution on cover cropped plots.

Temporal parameters were not successful at predicting Br⁻ profiles at 143 days (May sampling) or 280 days (September sampling). Temporal parameters predicted a profile that was much more dispersed than the observed spatial profile (Figure 4.8). To satisfy the conservation of mass, peak Br⁻ concentrations were underestimated. The inability of the temporal parameters to predict tracer concentration at an adjacent location might either result from transport parameters being highly variable over space or hydrodynamic dispersion of the tracer plume is not constant in time. Due to temporal parameters consistently overestimating dispersion, the second explanation seems more appropriate.
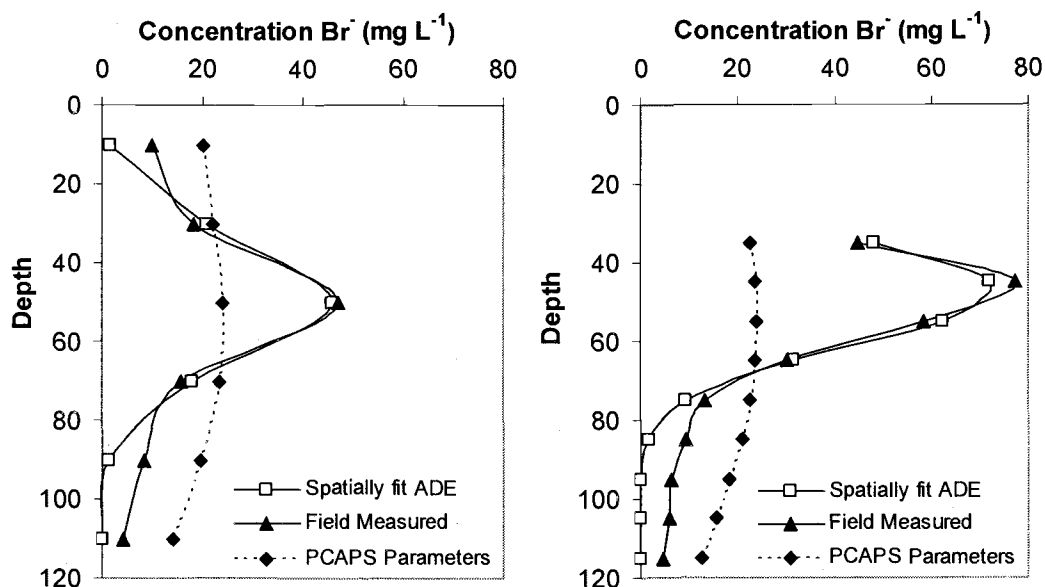
Figure 4.8 Observed Br⁻ concentration profiles for May 2001 soil sampling (a) and September 2001 sampling (b) and the predicted concentration using transport parameters calculated from the temporal breakthrough measured by the PCAPS.

Dry conditions with no intense rain events dominated the site preceding the soil sampling in summer 2001. This type of weather favors matrix flow and D_0 spreading processes. Temporal parameters calculated after May 2002 show the hydrodynamic dispersion of the dry 2001 water year in addition to the above average 2002 water year. Intense rainfalls favored high pore velocities and some preferential flow during the second year, causing the plume to spread and preventing transport parameters to be applicable to both halves (in time) of the tracer breakthrough. Effectively, it appeared that the ADE successfully fitted the concentration data measured from PCAPS, but was incapable of accounting for the transport mechanism dominating tracer transport throughout the breakthrough.

The ADE was fit to the spatial tracer profiles to generate transport parameters (spatial parameters). As suggested by Figure 4.8, spatially fitted dispersivities were much lower than the temporal parameters (Figure 4.9) in all but one location. Spatial dispersivities calculated in September were generally smaller than those calculated in May. This was due to the smaller pore water velocities calculated for September as very little additional water was collected from the PCAPS between May and September, mathematically lowering the average pore water velocity since tracer application. Four of the nine spatial profiles provided negative dispersivities in September. Such non-physical values were generated because of the method of calculating α that accounted for D_0 (Equation 4.11). The negative dispersivities reflect the fact that all spreading could be explained by D_0 during dry summer months in the Willamette Valley.
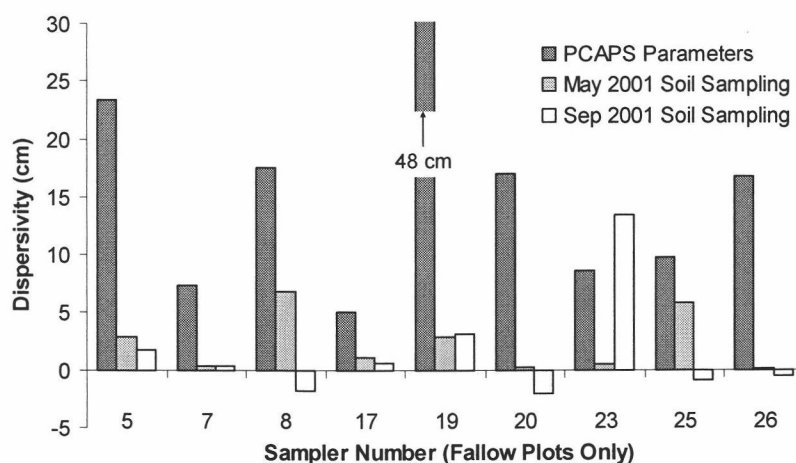
Figure 4.9 Comparison of α values calculated temporally from PCAPS and spatially from manual soil sampling.

4.5 Conclusions

Three applications of conservative tracers were made to vegetable plots over an eleven year period. Breakthroughs of the tracers at the 1.2 meter depth took place over several seasons and were predominately driven by winter rainfall. Preferential flow was observed at the site, but was not the dominant factor in tracer transport and spreading. Long periods of flow through the soil matrix resulted in unimodal tracer breakthroughs with steady rising and falling limbs. The ADE fit the data well.

Tracer breakthrough and transport parameters varied following each application. Dispersivity values calculated from different tracer events were found to have no correlation in time and a log-normal distribution in space. Variability of α through time could not be explained by pore water velocity. Dispersivity values calculated using PCAPS had no capacity to predict spatial Br⁻ concentrations for earlier times during the same breakthrough. Spatially calculated α values during May and September 2001 could not capture the complex solute spreading processes occurring the following winter and expose the inability for one model parameter to describe solute spreading in natural conditions.

These results call into question the concept of a characteristic and static α within soil systems; instead suggesting that differences in cycles between molecular diffusion and

mechanical dispersion dominated periods were foremost in determining solute spreading. Dominant spreading processes were related to varying periods of preferential and matrix flow and the balance between more and less water mobile portions of the soil column. Transient flow along structural features between long periods of molecular diffusion dominated transport affected tracer spread in the system. These observations agree with those of Persson and Berndtsson (1999) as they correlated smaller dispersivities with more consistent water application and less variation in water velocity.

These findings imply that the ADE has limited capability to predict solute spreading for structured soils under variable rainfall conditions. In view of the inability of the current conceptual model describing solute spread as dependent on a static α, site specific parameters calculated from previous experiments should not be used until variations in water inputs, risk and room for error in the engineering application are considered.

Acknowledgements

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5. Summary and Conclusions

Contamination of groundwater aquifers with agrochemicals is a problem impossible for growers in Oregon and around the country to ignore. As our increasing population becomes ever more dependent on groundwater for domestic use, protecting this drinking water resource is imperative.

PCAPS wick lysimeters were used to sample shallow groundwater under vegetable plots at the North Willamette Research and Extension Center (NWREC) in Oregon's Willamette Valley. Water collected over an eleven year period was analyzed to quantify the concentration and mass of NO_3^- leaching below the root zone of plots managed with and without winter cover crops. Both cereal and common vetch / triticale cover crops were tested for their ability to reduce NO_3^- additions to groundwater through the scavenging residual N from the root zone. Three conservative tracers were applied to the vegetable plots, the first a Br^- tracer in 1992, second a Cl^- tracer in 1995, and the last another Br^- tracer in 2000. To determine the process by which cover crops reduce NO_3^- contamination, NO_3^- and Br^- concentrations were determined through soil sampling and extraction of ions within the soil water. The breakthrough of each applied tracer was measured with PCAPS at 1.2 meters depth to determine the ability of the Advection Dispersion Equation (ADE) to describe and predict the longitudinal spreading of soluble chemicals within highly structured soils under natural rainfall conditions.

Cereal and common vetch / triticale cover crops demonstrated their utility as a best management practice (BMP) for reducing $NO_3^- - N$ concentrations in soil water. A 29% average reduction in $NO_3^- - N$ concentration was achieved on fully fertilized plots due to cover crops. Reductions in mass of NO_3^- leached (40%) were achieved using cover crops on fully fertilized plots. Considering the success of cover crops observed in these and many other field experiments, it is time for the industry to utilize this nitrogen BMP when conditions permit. With the recent focus on the quality of Southern Willamette Valley groundwater as a drinking water source for the future, the findings of this research are very applicable.

Soil sampling and extraction indicate that common vetch assimilated root zone NO_3^- and Br^- during the winter of 2000, retained the ions in plant tissue during the winter and apparently redistributed them on the soil surface during spring through mineralization. Cover cropping not only has the potential to reduce contamination to the groundwater, but may also provide a N credit to the soil once organic matter levels reach the critical level where mineralization rates exceed assimilation rates during the growing season.

Analysis of chemical transport using three tracer applications showed that spreading characteristics varied for each of the events. The Advection Dispersion Equation used to model observed tracer breakthrough at PCAPS depth fit the data well was able to generate transport parameters that described the solute spreading above individual samplers for each tracer breakthrough. Calculated transport dispersivity values were not constant over time and space. Variability of dispersivity through time could not be explained by pore water velocity and question the concept of a characteristic soil dispersivity, instead suggesting that differences in cycles between diffusion dominated and dispersion dominated periods were the determinants in solute dispersion. The results of this study question the utility of the ADE for the prediction of solute spreading in structured soils under variable rainfall conditions.

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Appendices

Appendix A: Field Measurements of Nitrate Leaching Below Willamette Valley Row and Mint Crops

Jeff Feaga and John Selker

Problem

Western Oregon's Willamette Valley has fertile soils and a climate ideal for growing a wide variety of crops. Long rainy winters make the Willamette Valley vulnerable to post-harvest nitrate (NO_3^-) leaching and shallow groundwater contamination. According to the United States Geological Survey (1998), 9% of wells in the Willamette Valley exceed the EPA's 10 ppm drinking water standard for nitrogen (N) in the NO_3^- compound, or NO_3^--N. A study of 281 domestic drinking water wells in Lane County showed that 22% of the wells exceeded the same standard (Penhallegon, 1994). In a 2000-2001 study of 476 wells in the Southern Willamette Valley, 35 wells exceeded the drinking water standard, with 21% of the total wells exceeding 7 ppm NO_3^--N (DEQ, 2002). Other studies have shown high concentrations of NO_3^--N within water draining from experimental fields planted in corn, broccoli, and snap beans (Sattell et al., 1999). Exposure to high concentrations of NO_3^- is known to cause breathing complications in infants (Weisenburger, 1993) and has been linked to cancerous tumors in laboratory animals (Pierzynski et al., 1994).

The objective of this field study was to measure NO_3^--N concentrations in water percolating below root depth on privately owned and operated vegetable and mint fields within Lane County. Two groundwater samplers were installed on each of six fields in conventional row vegetables, five fields in mint, and two fields in organic vegetables. All crop and nutrient management decisions on the fields were left entirely up to the individual growers. The goal of this research is to provide growers and extension agents with accurate information about the occurrence of NO_3^- leaching to shallow groundwater from typical commercial farms in the Willamette Valley.

The NO_3^- Leaching Process

When fertilized at rates recommended for maximum yield, most cultivated crops are not very efficient at using applied N fertilizers. Vegetables are highly inefficient at using applied N because of their shallow rooting depths and relatively large distances between plants due to cultivation practices. Due to the relatively high economic value, low water-stress tolerance, and large N requirements of mint, it is a high risk crop for NO_3^- leaching.

The quantity of N removed during harvest is generally 30 – 70% of the total quantity of N applied as fertilizer (Hermanson et al., 2000). Nitrogen not removed with the harvest either remains as NO_3^- in the soil, in the remaining plant residue, or is lost to the atmosphere as either ammonia (NH_3) or elemental nitrogen (N_2) and nitrous oxide (N_2O)

after denitrification. Excessive residual soil N after harvest may indicate that some of the following processes may be involved:

- Fertilizer is applied at **rates** exceeding recommended amounts to achieve maximum yield.
- Fertilizer was added at the wrong **time** of the growth cycle.
- Method of **application** is not efficiently supplying the growing plants with N.
- Excessive organic N is being **mineralized** into inorganic NO_3^-.
- A **poor yield** at harvest will ensure that significant quantities of NO_3^- remain in the soil to leach the following winter.

Several other factors also contribute to the potential for NO_3^- leaching (Table A.1).

Nitrate (NO_3^-) and ammonium (NH_4^+) are the only plant-available forms of N. Because the top few feet of most agricultural soils are well-aerated during the growing season, the majority of NH_4^+ in applied fertilizers is quickly converted to NO_3^- by oxidation. Decaying organic N in plants undergoes a biological conversion called mineralization, which is the transformation of organic N to inorganic plant-available forms.

In most natural systems, N is limited and is cycled through its various chemical forms without significant loss to aquifers. In agricultural systems, the need to remove the N limitation on production results in accumulated free NO_3^-. Nitrate is easily leached with moving water because it has a negative charge and therefore does not bond to soil particles. The most critical time for NO_3^- leaching is the beginning of the winter rains when soluble NO_3^- remaining after harvest are transported from the upper soil with percolating rainwater.

Table A.1 Key issues to consider when assessing the potential for NO_3^--N leaching.

Nitrate Leaching Potential

Amount of Residual Nitrogen Left After Harvest -
- ▲ Amount: nitrogen application in excess of crop recommendations
- ▲ Timing: nitrogen applications after crop ceases to assimilate nitrogen for growth
- ▲ Uniformity: inefficient nitrogen application methods

Irrigation Practices -
- ▲ Non-uniform irrigation due to incorrect operating pressures and/or worn or mismatched nozzles
- ▲ Over-irrigation in response to excessively dry or poor production areas
- ▲ High background nitrates in irrigation water

Mineralization of Organic Materials -
- ▲ Untimely decay of nitrogen-rich dry matter or manure

Landscape and Climate -
- ▲ Rainfall exceeding potential evapotranspiration for extended periods
- ▲ Rainfall while fields are not covered in standing crop
- ▲ Additional water saturating field from other areas (overland flow)

How Did We Make Field Measurements of NO$_3^-$ Leaching?

A cooperative study was operated between 1993 and 1998 between several Lane County vegetable and mint growers and the Oregon State University Extension Service through the Department of Bioresource engineering. Two large groundwater sampling units and three suction cup samplers were installed on each of six conventional vegetable, five mint, and two organic vegetable fields. The passive capillary samplers, or PCAPS, were installed at around 3.5 ft (1.1 m) below the field surface (Figure A.1). The method of installation ensured that the soil column above the PCAPS was not disturbed, and that the overlying soil's original structure was retained. Soil water and dissolved NO$_3^-$ percolating below the plant roots were collected as this water is likely to move into groundwater aquifers with sufficient time.
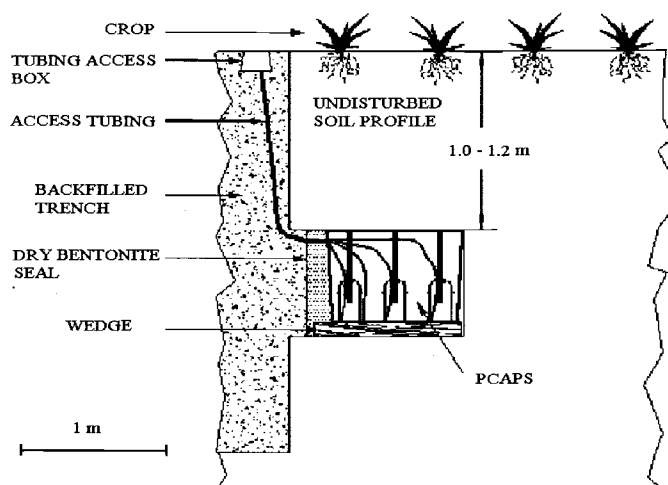
Figure A.1 PCAPS unit. Samplers were installed laterally from the trench to minimize soil disturbance above the lysimeters (Picture modified from Brandi-Dohrn (1993)).

PCAPS use fiberglass wicks that equilibrate to the surrounding soil's water pressure. As a result of this design, PCAPS can collect water moving through the soil over a range of conditions from saturated to draining and approaching field capacity. The PCAPS wicked surface collects water from an area of 3 ft^2 (0.28 m^2). The samplers were drained by pumping from the surface. The PCAPS had a 15.9 gallon (60 L) capacity and were large enough to require sampling only once a month. More details on the installation and operation of the PCAPS are given in Brandi-Dohrn et al. (1997) and Louie et al. (2000). In addition to PCAPS sampling, suction cup samplers were used to make point measurements of NO$_3^-$-N concentrations at the estimated winter water table depth ranging from 4.9 – 9.8 ft (1.5 – 3 meters).

How Were NO$_3^-$-N Concentrations Measured and Averaged?

Water collected by the PCAPS was frozen until it could be analyzed for NO$_3^-$-N concentration in the laboratory. Concentration data collected over the five-year study were averaged with respect to flow. Flow-weighted averaging is an appropriate method to represent the average concentration over multiple sampling events or when more than one sampler is measuring an event. To find a flow-weighted average for a five-year period, for example, the total mass of NO$_3^-$-N collected divided by the total volume of water collected would be the five year flow-weighted average. Flow-weighted averages are better than simply averaging monthly NO$_3^-$-N concentrations because the method can prevent misleading data caused when sampling events that collect small volumes have very high concentrations.

What We Found

NO$_3^-$-N Concentrations

Concentrations of NO$_3^-$-N observed below the Lane County conventional vegetable and mint crops were flow-weighted and averaged for the entire study period and grouped according to field number (Figures A.2 and A.3). Average NO$_3^-$-N concentrations below the root zone of five out of the six conventional vegetable fields exceeded the 10 ppm maximum NO$_3^-$-N concentration allowed by the EPA for water *intended for drinking*. Three out of the five mint fields had average NO$_3^-$-N concentrations exceeding the 10 ppm maximum. Although the water measured during these experiments was not intended for drinking and therefore not regulated, the data indicate NO$_3^-$-N concentrations in underlying aquifers will likely rise in the future. In regions of the Willamette Valley where high N input crops are being intensively grown over large contiguous areas, aquifers used for drinking water supply can be expected to approach or exceed the 10 ppm drinking water standard with time unless nutrient management practices are modified.
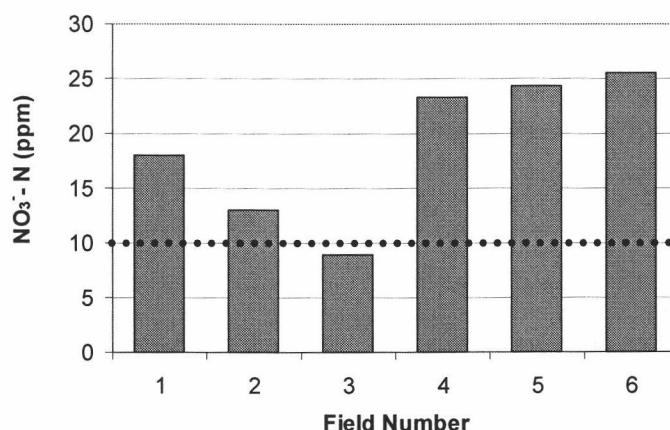
Figure A.2 Flow-weighted NO_3^--N concentrations for six fields planted to row crops during the entire study period. The dotted line indicates the EPA 10 ppm NO_3^--N drinking water standard.
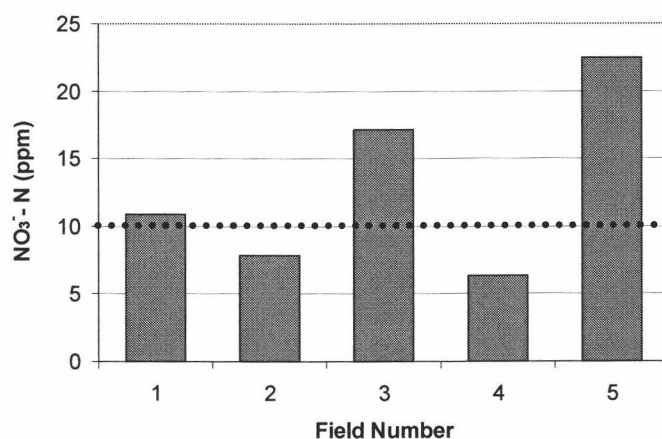
Figure A.3 Flow-weighted NO_3^--N concentrations for five fields planted to mint during the entire study period. The dotted line indicates the EPA 10 ppm NO_3^--N drinking water standard.

Organic growing operations represented in the study also showed considerable NO_3^--N concentration below the root zone (Figure A.4). After the first two seasons, it was clear that NO_3^--N concentrations were very high. It was determined that poultry manure was applied in amounts well above recommended agronomic rates. After an estimation of N content and mineralization rates of the manure, applications were reduced. The management changes enacted in February 1996 resulted in apparent reductions in NO_3^--N concentration during the latter half of the study. Mineralization rates and thus N availability from manure is hard to predict without a periodic analysis of the nutrient content in the manure. In addition, manure is often diluted with water to different degrees, making applications at recommended agronomic is rates highly dependent on water content.
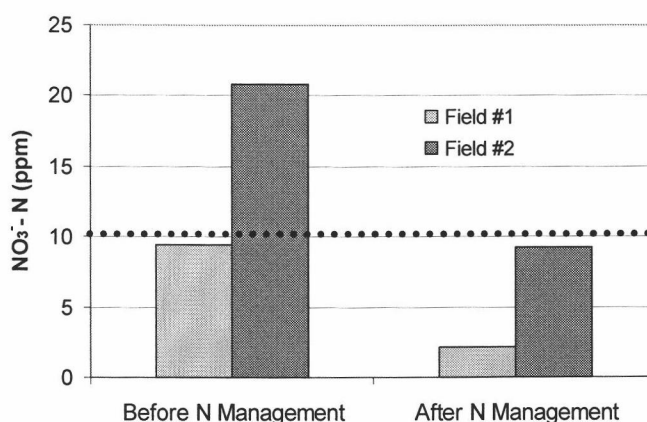
Figure A.4 Flow-weighted NO_3^--N concentrations for two fields planted to organically grown vegetables during the entire study period. After revising manure application rates according to plant requirements, a considerable decrease in NO_3^--N concentration was observed. The dotted line indicates the EPA 10 ppm NO_3^--N drinking water standard.

How Do NO_3^--N Concentrations Change Throughout the Season?

Concentrations from all samplers on all conventional vegetable fields were flow-weighted and grouped by month (Figure A.5). Concentrations were the highest in the summer when crops were fertilized and less water was available for dilution. At the onset of winter rains, the soil profile was flushed and most of the NO_3^- left after harvest is moved past sampler depth. Concentrations also decreased during this time due to the high volumes of water diluting the NO_3^--N. Even during late winter, NO_3^--N concentrations on average were near to or higher than the EPA 10 ppm standard for drinking water.
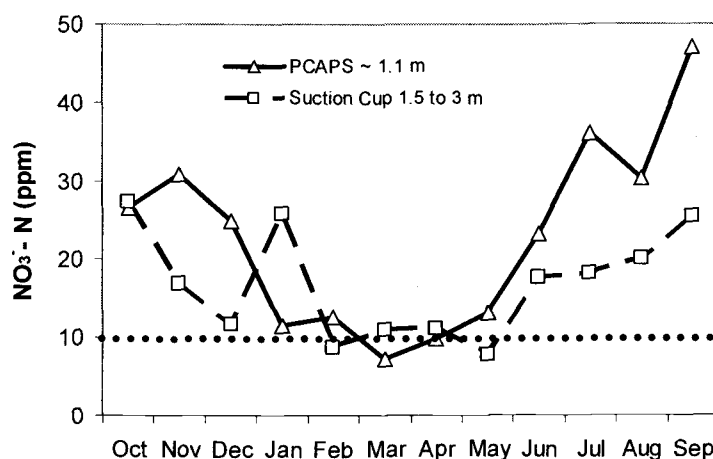
Figure A.5 Measured monthly flow-weighted average NO$_3^-$-N concentrations below six Lane County conventional vegetable fields throughout the five-year study period. Shown are the concentrations measured by PCAPS as well as suction cup samplers. The dotted line indicates the EPA 10 ppm NO$_3^-$-N drinking water standard.

How Much Applied N Fertilizer Are We Losing Through Leaching?

The ability of Lane County PCAPS to make an accurate estimate of the real volume of water percolating to the groundwater was determined by Louie et al. (2000). The analysis was made by comparing collected water volumes from PCAPS to expected volumes calculated from a water balance. Results indicated that Lane County PCAPS overestimated the volume of water entering the ground by 25%. If we take this into account, we are able to make a good estimate of total mass of NO$_3^-$-N lost to the groundwater (total mass NO$_3^-$-N is the estimated water volume multiplied by the concentration NO$_3^-$-N).

Average N fertilizer application rates were 200 lbs / acre for row crops and 250 lbs / acre for mint. Average losses of NO$_3^-$-N mass per acre were calculated for each of the study years and are shown in Figure A.6 and A.7. Average annual mass losses for each crop type were 93 and 82 lbs / acre from conventional vegetable and mint fields, respectively. In monetary terms, such losses of N would equate to $3300 - $3700 for each 100 acre field if N is estimated to cost $0.40 / Lb. It is important to note that a fraction of the NO$_3^-$-N leached from this soil system may have originated as mineralization of organic matter. Soils are like a bank with respect to N fertilizers; sometimes N is applied and "saved" in the soil, while at other times N is "withdrawn" through mineralization. Whether or not the soil is gaining or losing N is dependent on many factors, including cultivation history, temperature, water content and the carbon to N ratio within the soil.

Figure A.6 Average NO_3^--N mass leaching rate from vegetable fields during each of the study years. Water years are used (October of previous year to the September of posted year) as they correspond to breaks in the growing season and encompass an entire winter's precipitation. Note - October 1993 is not represented.
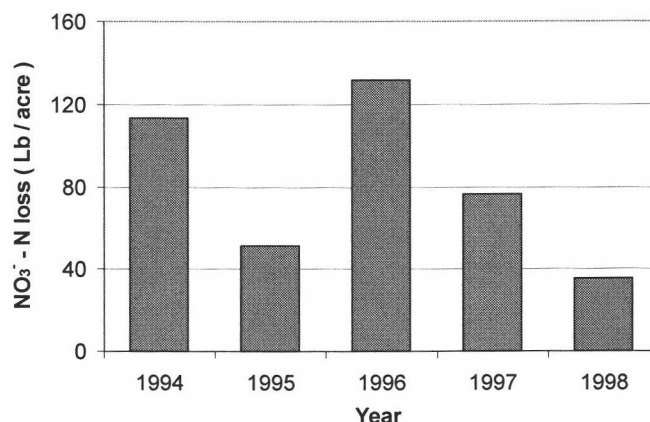
Figure A.7 Average NO_3^--N mass leaching rate from mint fields during each of the study years. Water years are used (Oct. – Sept.) as they correspond roughly to breaks in the growing season and encompass an entire winter's precipitation. Note - October 1993 and August and September 1998 are not represented.

What Do Our Findings Mean For Management?

A five-year field study of water collected from below the root zone of commercially grown vegetables and mint showed that NO_3^--N concentrations measured below a

majority of the fields were higher than the EPA 10 ppm NO_3^--N standard set for drinking water.

The good news is that both excellent yields and environmental stewardship can be achieved by managing N and water inputs, as illustrated by sites with low NO_3^- losses under each cropping system. There are many options available to the grower to manage N. However, at this time it is still uncertain which of these best management practices (BMP's) are the most effective in the Willamette Valley. Currently, growers and research extension are collaborating to find:

- Which BMP's are most effective for the various cropping systems and soil types in the Willamette Valley?
- What barriers do growers face in adopting alternative management practices and BMP's to reduce NO_3^- loss?
- Which are the most cost effective management practices to reduce NO_3^- inputs to groundwater?

Though it is unclear at this time which of the BMP's will be found most effective and practical in the Willamette Valley, it is likely that future BMP's will concentrate on the following opportunities:

1) Reduction of N Inputs

- **Determining Rate and Timing of Fertilizer Applications** - Fertilizer applications are specific for each type of crop and often vary depending on soil type. Applications that are coordinated to times of rapid plant uptake are most effective to achieve high yields and minimize leaching losses.

- **Soil Sampling** - Method focuses on periodic soil sampling to reduce N inputs. Chemical analysis of soils provides information on residual N content, as well as an estimate of organic matter content.

- **Accounting for Mineralization** - Mineralization of organic matter is a large source of plant-available N and can provide a substantial portion of N for the growing crop. If not accounted for, it may produce surplus plant-available NO_3^- that can be leached.

- **Stem NO_3^- Tests For Peppermint** - The changing N requirements of peppermint throughout the growing season give an opportunity to design an application schedule according to crop N demand. To determine the current demand of N for the crop, a weekly stem tissue NO_3^- test is an option. The stem NO_3^- test uses a correlation between oil yields and NO_3^- content within the mint stem. Stem samples are taken throughout the period of fastest growth to ensure that NO_3^- content within the mint stem does not fall below critical values. More information can be found in Brown (1982) and Smesrud and Selker (1998).

2) **Use of Cover Crops to Scavenge N** - Research in Oregon and around the country has shown promising results of winter cover crops to reduce the mass of NO_3^- leached below the root zone during the winter. Cereal crops are the most effective winter covers (Brandi-Dohrn, 1997; Sattell et al., 1999).

3) **Irrigation Management** - To effectively minimize NO_3^- leaching from growing operations, it helps to make sure irrigation systems are properly maintained and that fields are not over-irrigated. Over-irrigation can easily leach plant-available N past the root-zone making it unavailable for plant growth. Irrigation systems applying non-uniform coverage will cause uneven NO_3^- leaching rates from the root zone. As a result, parts of the field will be N deficient, requiring additional applications or risking a poor harvest. Other areas of the field can accumulate N in excess of crop requirements.

   Soil properties can vary throughout fields. For example, areas of the field may consist of more coarsely textured soils. The infiltration of water into course soils is typically more rapid and can cause high leaching rates. Irrigation schemes should consider the variability of soils in the field.

   An effectively managed irrigation scheme can consider the following:

   - **Operating pressure** - Each system has an acceptable range of operating pressures. Deviation from this leads to poor uniformity of application.

   - **Nozzles** - Ensure that nozzles are of a consistent make and diameter and are not worn. Periodically (at least every four years) replacing all the nozzles at once is recommended to maintain uniform application.

   - **Soil water monitoring** - With the aid of soil water monitoring, irrigation schedules can be adjusted to ensure that soil water content in the root zone is maintained within an appropriate range.

   - **Schedule irrigation based on expected ET** - Irrigation requirements vary throughout the year as potential evapotranspiration (ET) rates change with the weather. Set times of irrigation events can be adjusted as ET rates change to irrigate to depths that will recharge the water holding capacity of the soil

Where to Find More Information

Several extension papers discuss methods to streamline irrigation operations. The *Western Oregon Irrigation Guides* EM 8713 (Smesrud et al., 2000) provide worksheets to aid in calculating irrigation set times based on potential evapotranspiration for 16 different commonly grown Willamette Valley crops. Sprinkler head maintenance and its relationship to application uniformity is discussed in Louie and Selker (2000). The post-harvest water requirements of mint are presented in Smesrud and Selker (1999).

Other extension publications describing N requirements and application schedules for various vegetables, mint and other crops include *Nitrogen uptake and utilization by Pacific Northwest crops* (Sullivan et al., 1999) and *Peppermint growth and nutrient uptake* (Christensen et al., 1998). A useful on-line source for papers and information concerning many aspects of agriculture and the environment is the Oregon State University extension and experiment station communications, located on the World Wide Web at http://eesc.orst.edu/default.html.

The first step to address the NO_3^- leaching problem is to become aware of the risk. With cooperation, groundwater and soil resources will be conserved for years to come. Using county extension is a helpful resource to keep up on current information that can assist growers in managing a sustainable, productive and profitable business.

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Appendix B: Nitrates and Groundwater: Why Should We Be Concerned With Our Current Fertilizer Practices?

J. Feaga, R. Dick, M. Louie, J. Selker

Introduction

The general understanding of our water resources and their vulnerability to contamination has improved in recent decades. The origins of much of this contamination are termed non-point sources as they originate from activities carried out over large areas rather than a pipe outfall. Chemicals originating from non-point sources enter our streams, lakes and groundwater through processes such as runoff and percolation, and have proved to be a large contributor of contamination. Non-point source pollution is typically associated with agricultural activities and is difficult to regulate and control. Supported by many years of research, cooperation with the farm community and the technology to manage and present data, agricultural scientists are now able to measure some of the environmental impacts of chemical loss from agricultural operations. Understanding the scope and depth of the problem is the first step to address agricultural contamination.

Currently, the status of agriculture in Oregon and across the United States is such that production, profit and competitiveness are of utmost importance. The methods of farming currently in use are proven to be very effective at maximizing these economic measures, but do not generally account for degradation of soil and water resources. Frequently, the most serious agricultural contamination problems with respect to water resources stem from loss of excess nutrients. Nutrients and their correct use and application have become a widespread issue following the separation of crop and animal production, a trend which began 50 years ago when commercially made fertilizers were made available (Dinnes et al., 2001). Nitrogen (N) fertilizer is necessary for the competitive production of crops, but in excess, nitrates (NO_3^-) can be toxic to infants (Weisenburger, 1993) and infirm people. Nitrate can adversely affect aquatic habitats by overly stimulating primary production in surface water, causing anoxic conditions and eutrophication. Voters and lawmakers concerned about the environment understand this and have put ever-increasing pressure on growers and ranchers to control their environmental impacts while continuing to supply ample and low-cost food.

This paper describes the challenges faced by the western Oregon grower with concern to NO_3^- leaching losses to shallow groundwater aquifers. This paper describes: 1) The conditions that make western Oregon a particularly vulnerable environment to NO_3^- leaching and contamination; 2) The incentives for western Oregon growers to undertake a careful nutrient management plan; and 3) The options for best management practices (BMP's) available to growers including:

- Winter Cover Crops
- Soil Testing
- Fertilizer and Manure Chemical Analysis
- Maintenance of Irrigation Systems

Nutrient Contamination

A Mediterranean climate with seasonally high water tables combined with intensive agriculture makes the Willamette Valley a particularly vulnerable location for nitrate (NO_3^-) contamination (Owens, 1990). According to the United States Geological Survey (1998), 9% of wells in the Willamette Valley exceed the EPA 10 ppm drinking water standard for N in the nitrate compound (NO_3^--N). A study of 281 domestic drinking water wells in Lane County showed that 22% of the wells exceeded the same standard (Penhallegon, 1994). In a 2000-2001 study of 476 wells in the Southern Willamette Valley, 35 wells exceeded the drinking water standard, with 21% of the total wells exceeding 7 ppm NO_3^--N (DEQ, 2002). Other studies have shown high concentrations of NO_3^--N within water draining from experimental fields planted in row crops such as corn, broccoli, and snap beans (Sattell et al., 1999).

Nitrate and ammonium (NH_4^+) are the only plant-available forms of N. Other forms of N in the environment can be transformed to NO_3^- through several natural processes; NH_4^+, for example, is oxidized by soil bacteria into NO_3^- in aerated soils. Because the top few feet of most agricultural soils are well-aerated during the growing season, the majority of NH_4^+ in applied fertilizers is quickly converted to NO_3^- by oxidation. Decaying organic N in plants undergoes a biological conversion called mineralization; the transformation of organic N to inorganic plant-available forms. In most natural systems, N is limited and is cycled through its various chemical forms without significant loss to aquifers. In agricultural systems, the need to remove the N limitation on production results in accumulated free NO_3^-. Nitrate is easily leached with moving water because it has a negative charge and therefore does not bond to soil particles. The most critical time for NO_3^- leaching is the beginning of the winter rains when soluble NO_3^- remaining after harvest is transported from the upper soil with percolating rainwater.

During Oregon's dry summers, careful irrigation practices ensure that NO_3^-, NH_4^+ or ammonia (NH_3) applied during the spring and early summer remain in the soil root zone throughout the growing season. The quantity of N removed in harvest is almost always less than the quantity of N fertilizer applied. Typically, the quantity of N removed during harvest is around 30 – 70% of the total quantity of N made available to the plant by fertilizer applications, mineralization of organic matter in the soil and dissolved in irrigation or rainwater (Hermanson et al., 2000).

If N efficiencies can be estimated, why can't a cookbook approach be used to ensure that crops are being fertilized at a rate to maximize efficiency and minimize N available to leaching? In reality, natural processes and seasonal variation make exact N balances difficult (Table B.1). Nitrogen not removed with the harvest either remains as NO_3^- in the soil, in the remaining plant residue, immobilized in soil organic matter, is lost to the atmosphere ammonia (NH_3), or is lost as elemental nitrogen (N_2) and nitrous oxide (N_2O)

after denitrification. With the onset of the rainy season, soil and fertilizer N unused by the summer crop is susceptible to leaching to the shallow water table. Beyond the depths of cover crop roots, biological activity slows down, though it is possible that denitrification can take place in anaerobic zones and may be an important process in some soil systems in Western Oregon (R.P. Dick, 2003, personal communication). Deep NO_3^- not included in denitrification processes will be lost to the groundwater.

Nitrate -N concentration in groundwater is a function of the quantity of water recharging the aquifer and the rate of N loss from the surface. Climates receiving much rainfall would be expected to dilute N inputs and therefore have lower NO_3^--N concentrations in groundwater. Table B.2 gives the steady-state groundwater NO_3^--N concentration for aquifers with an array of N and water input rates. To use Table B.2, make an estimate of the total annual aquifer recharge of water in equivalent depth (height of water). The average rainfall in the Willamette Valley is around 45 inches, but only about 30" of water truly infiltrates into the ground and recharges the aquifer because of other processes such as evaporation, surface runoff, and storage in the soil. A typical rate of N loss from an operation growing mint or vegetables is 80 lb/acre. Using Table B.2, these values would result in a long-term groundwater NO_3^--N concentration nearing 12 ppm. Even in the wet Willamette Valley we can expect long-term groundwater concentrations to exceed the EPA drinking water standard. Drier climates east of the Cascades or in California's Central Valley would expect much higher long-term concentrations. Existing data support these simple calculations.

Table B.1 Average N removal efficiencies for some common crop types in the Willamette Valley. The amount of N removed from the field is usually small in comparison to the amount added.

Calculating the N efficiency of a crop is not an easy task. Why?

When accounting for N in a balance, the easiest inputs to consider are the fertilizer additions.

With a crop analysis we could easily make the calculation:

$$\frac{\textbf{N in the mature crop}}{\textbf{N applied as fertilizer}} = \textbf{Fertilizer recovery, but NOT overall N efficiency}$$

This calculation is not really the N uptake efficiency for the crop because it does not account for all N inputs, which are very difficult to measure.

To calculate the **overall N efficiency**, it is necessary account for other N inputs in the equation.

$$\frac{\textbf{N in the mature crop}}{\textbf{fertilizer N + residual inorganic N + mineralization of organic N + N from irrigation or rainfall}}$$

Other N inputs can be significant. Often, ½ of plant available N comes from sources other than applied fertilizer. N provided from mineralization of last years' crop residue can be substantial. In fact, crops such as broccoli, cauliflower, grass for seed, and hops leave more N in the crop residue than is removed from the field during harvest.

An oversimplified, theoretical N efficiency for corn, a crop with large N requirements and low N efficiency would look more like this (Example extracted from Hermanson et al., 2000)

$$\frac{\textbf{Max Corn N Accumulation 225 lb/acre}}{\textbf{225 lb/acre N from fertilizer + ~ 225 lb/acre N supplied by other sources}} = \textbf{50\% N efficiency}$$

The 225 lb/acre not used by the crop is stored in the soil, denitrified, or is leached into groundwater. As the balance describes more N processes take place than meets the eye!

Nitrate leaching potential is most directly controlled by the amount of NO_3^- fertilizer remaining in the soil after the growing season. Excess residual soil N after harvest indicates that either: 1) Fertilizer is applied at rates exceeding recommended amounts to reach maximum yield; 2) Fertilizer was added at the wrong time of the growth cycle; 3) The method of application is not efficiently supplying the growing plants with N; 4) Excessive organic N is being mineralized into inorganic NO_3^-. Of course, a poor yield at harvest will ensure that significant quantities of NO_3^- remain in the soil.

Table B.2 Theoretical steady-state NO_3^--N concentrations calculated for an aquifer with variable quantities of recharge and N loading. Nitrate -N concentrations are highest in areas with little recharge and large leaching losses. Values assume entire land area collecting recharge water receive the same management. Example is for NO_3^--N concentration in the Willamette Valley and is explained in the text.

Expected Long-Term Shallow Aquifer NO_3^-- N Concentrations (ppm) Under Variable Rainfall Conditions and N Inputs						
N loss	Percolation to Groundwater (inches / year)					
Lbs/Acre	5	10	15	20	30	40
20	18	9	6	4	3	2
40	35	18	12	9	6	4
60	53	27	18	13	9	7
80	71	35	24	18	12	9
100	88	44	29	22	15	11
150	133	66	44	33	22	17
200	177	88	59	44	29	22
250	221	111	74	55	37	28
300	265	133	88	66	44	33
400	354	177	118	88	59	44

Quantifying Leaching Under Willamette Valley Crops

A cooperative effort has been undertaken by Oregon growers, the Oregon State University Extension Service, the Oregon Dept. of Environmental Quality and the Oregon Dept. of Agriculture to understand the process of NO_3^- leaching. Nitrate leaching studies have been completed throughout Lane County and at Oregon State's North Willamette Research and Extension Center (NWREC). The goal of these studies was to assess the characteristics of NO_3^- leaching within the Willamette Valley and research practical and economically feasible options for management. These long term studies show that Oregon agriculture contributes large amounts of nutrients to groundwater, but very effective methods exist to treat the problem.

Passive Capillary Wick Samplers

A unique sampling device was used to quantify NO_3^- leaching throughout the Willamette Valley. Termed PCAPS for Passive Capillary Samplers, the lysimeters are capable of quantifying the volume and chemical content of water percolating to groundwater. PCAPS use fiberglass wicks that equilibrate to the surrounding soil's water pressure. The samplers are capable of intercepting draining soil water solutions from a

known area below soils with pressures ranging from 0 m - 0.8 m. As a result of this design, PCAPS can collect water moving through the soil over a range of conditions from saturated to draining and approaching field capacity. The PCAPS surface panel collects water from a horizontal plane with an area of 3 ft^2 (0.28 m^2). The method of installation ensured that the soil column above the PCAPS was not disturbed, thus retaining original hydraulic characteristics above the lysimeters (Figure B.1). The samplers were drained by applying a vacuum through the sample tubing from the surface.
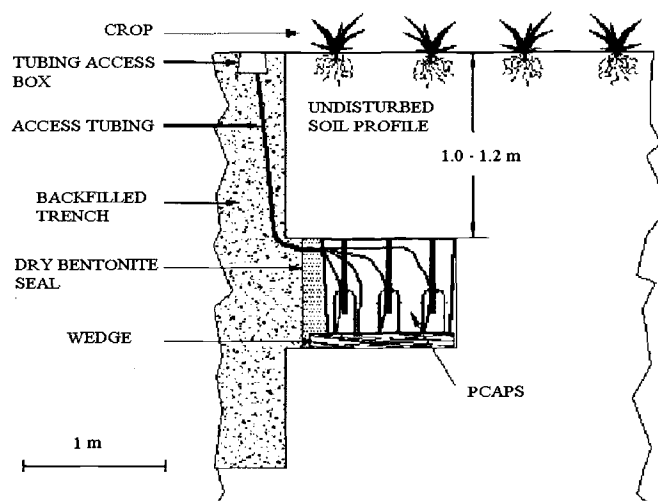
Figure B.1 PCAPS sampling unit. Samplers were installed laterally from the trench to minimize soil disturbance above the lysimeters (Picture modified from Brandi-Dohrn (1993)).

Two different PCAPS designs were used throughout the study. The PCAPS used in the controlled experiments at the North Willamette Research and Extension Center (NWREC) were built with fiberglass and a stainless-steel top panel. Three 1 gallon (3.78 liter) bottles collected the water. This relatively small capacity required frequent pumping. The PCAPS models used in Lane County were either similar to those at the NWREC or were built out of high density polyurethane (HDPE). The Lane County PCAPS had a 15.9 gallon (60 L) capacity. The PCAPS built of HDPE eliminated the use of a separate collection vessel altogether. The large capacity of the Lane County samplers enabled the units to be sampled monthly, even when precipitation events during the period were frequent or large. More details on the installation and operation of the PCAPS are given in Brandi-Dohrn et al. (1997) and Louie et al. (2000).

Concentration data collected over the five-year study were averaged with respect to flow. Flow-weighted averaging is an appropriate method to represent the average concentration over multiple sampling events, or when more than one sampler is measuring an event. To find a flow-weighted average for a five-year period, for example, the total mass of NO$_3^-$ collected divided by the total volume of water collected would be the five year flow-weighted average. Flow-weighted averages are better than simply averaging monthly NO$_3^-$-N concentrations because the method can prevent misleading

data caused when sampling events that collect small volumes have very high concentrations.

Leaching Under Field Conditions

In Lane County, 21 privately owned and operated commercial agricultural fields were assessed for their contribution of NO_3^- to shallow groundwater. Growers volunteered to take part in the study and made all crop and nutrient management decisions. Two PCAPS lysimeters were installed on each of the fields planted with row vegetables, mint, certified organic vegetable crops, orchards or blueberries. The four years of water quality data collected under the vegetable crops and five years of data under mint crops are the focus of this analysis. The data collections began in November 1993 and continued until November 1997 and July 1998 for the vegetable fields and mint fields, respectively. PCAPS were installed at depths ranging from 3.3 – 3.9 ft (1.0 – 1.2 m) below the field surface. In addition to PCAPS, an alternative smaller groundwater sampling tool called suction cup samplers were used to make a point measurement at the estimated winter water table depth 4.9 – 9.8 ft (1.5 – 3 m). Suction cup samplers operate differently than PCAPS samplers because the user must apply suction to the devices that exceed the water-holding pressure of the soil and are best used near the water table, where PCAPS cannot be employed (Brandi-Dohrn et al., 1996).

Monthly flow-weighted NO_3^--N concentrations under fields planted in row crops (vegetable annuals) are shown in Figure B.2. Nitrate -N concentrations below mint show similar trends (Figure B.3). From monthly NO_3^--N averages, we can determine NO_3^- leaching potential throughout the year. Concentrations were the highest in the summer when crops were fertilized and less water was available for dilution. At the onset of winter rains, the soil profile was flushed and most of the NO_3^- left after harvest is moved past sampler depth. Concentrations also decreased during this time due to the high volumes of water diluting the NO_3^--N. Even during late winter, NO_3^--N concentrations on average were near to or higher than the EPA 10 ppm standard for drinking water.
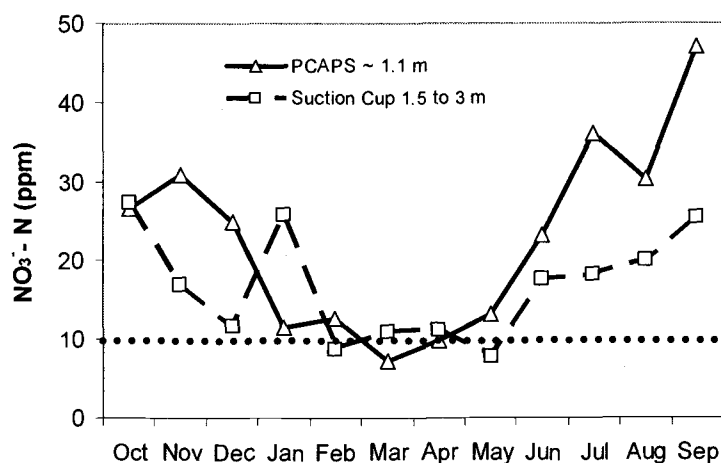
Figure B.2 Measured monthly flow-weighted average NO_3^--N concentrations below six Lane County vegetable fields during the four-year study period.   Shown are the concentrations measured by PCAPS as well as suction cup samplers. The dotted line indicates the EPA 10 ppm NO_3^--N drinking water standard.
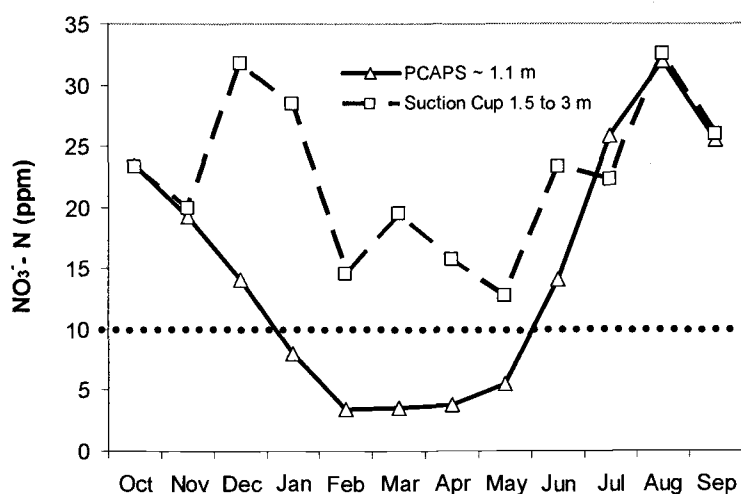
Figure B.3 Measured monthly average NO_3^--N concentrations below five Lane County mint fields during the five-year study period.   Shown are the concentrations measured by PCAPS as well as suction cup samplers. The dotted line indicates the EPA 10 ppm NO_3^--N drinking water standard.

Flow-weighted NO_3^--N concentrations over the entire study period showed that the average recharge concentration exceeded the EPA NO_3^--N drinking water standard for a majority of the fields (Figures B.4 and B.5). The data indicated NO_3^--N concentrations in underlying aquifers may rise in the future. In regions of the Willamette Valley where high N input crops are being intensively grown over large contiguous areas, aquifers used for drinking water supply can be expected to approach or exceed the 10 ppm drinking water standard with time unless nutrient management practices are modified.
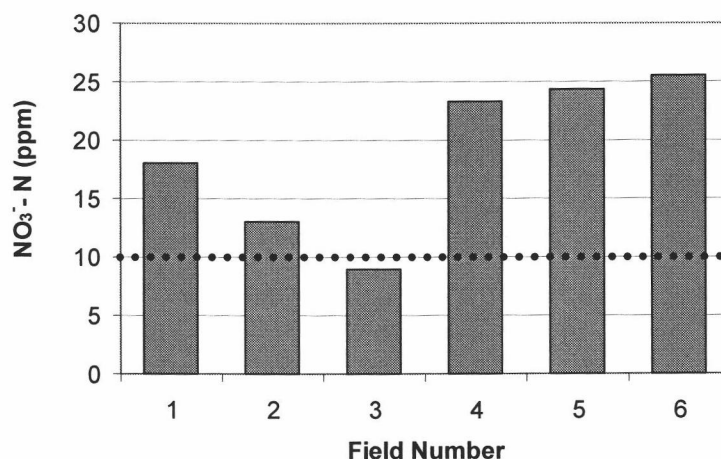
Figure B.4 Flow-weighted NO_3^--N concentrations for six fields planted to row crops during the entire four-year study period. The dotted line indicates the EPA 10 ppm NO_3^--N drinking water standard.
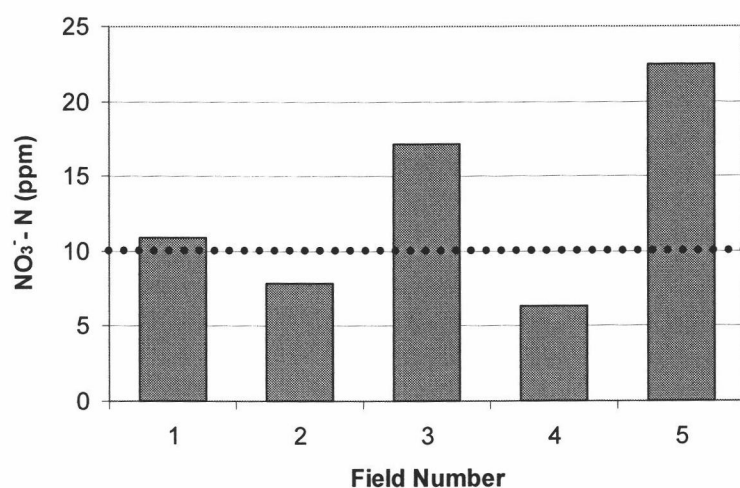
Figure B.5 Flow-weighted NO_3^--N concentrations for five fields planted to mint during the entire five-year study period. The dotted line indicates the EPA 10 ppm NO_3^--N drinking water standard.

The ability of Lane County PCAPS to make an accurate estimate of the real volume of water percolating to groundwater was determined by Louie et al. (2000). The analysis was made by comparing collected water volumes from PCAPS to expected volumes calculated from a water balance. Results indicated that Lane County PCAPS overestimated the volume of water entering the ground by 25%. If we take this into account, we are able to make a good estimate of total mass of NO_3^--N lost to groundwater (total mass NO_3^--N is the estimated water volume multiplied by the concentration NO_3^--N).

Average N fertilizer application rates were 200 lbs / acre for row crops and 250 lbs / acre for mint. Average losses of NO_3^--N mass per acre were calculated for each of the study years and are shown in Figure A.6 and A.7. Average annual mass losses for each crop type were 93 and 82 lbs / acre from conventional vegetable and mint fields, respectively. In monetary terms, such losses of N would equate to $3300 - $3700 for each 100 acre field if N is estimated to cost $0.40 / Lb. It is important to note that a fraction of the NO_3^--N leached from this soil system may have originated as mineralization of organic matter. Soils are like a bank with respect to N fertilizers; sometimes N is applied and "saved" in the soil, while at other times N is "withdrawn" through mineralization. Whether or not the soil is gaining or losing N is dependent on many factors, including cultivation history, temperature, water content and the carbon to N ratio within the soil.

Figure B.6 Average NO_3^--N mass leaching rate from vegetable fields during each of the study years. Water years are used (October of previous year to the September of posted year) as they correspond to breaks in the growing season and encompass an entire winter's precipitation. Note - October 1993 is not represented.
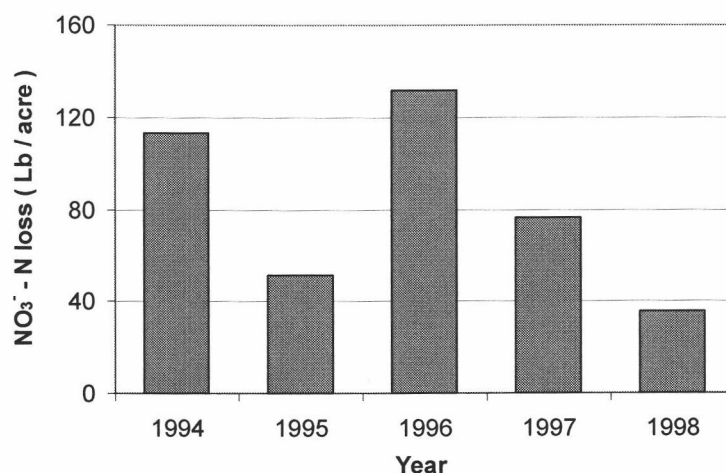
Figure B.7 Average NO_3^--N mass leaching rate from mint fields during each of the study years. Water years are used (Oct. – Sept.) as they correspond roughly to breaks in the growing season and encompass an entire winter's precipitation. Note - October 1993 and August and September 1998 are not represented.

Incentives for Growers

At this point, we have described the NO_3^- leaching problem and have seen some convincing field evidence of high NO_3^--N concentrations in western Oregon. We are fortunate in western Oregon because practical and effective solutions to reduce the problem do exist! Still, one of the most important hurdles of the NO_3^- contamination problem remains; what are the incentives for growers to change management practices? The most important and widespread incentives is the growers' own desire to be conscientious stewards of their soil and water resources. At the same time, the general consumer in western Oregon would prefer growers to deliver their product with minimal environmental impacts, and are often willing to pay a premium for responsibly grown products (Kurki and Matheson, 2001). Some may believe that simply reducing fertilizer use will result in a larger profit margin. However, the risk of using too little fertilizer and having a poor yield does exist, so reducing fertilizer inputs requires careful planning. Nonetheless, reducing N fertilizer inputs can amount to significant savings for the grower (Table B.3).

Table B.3 Estimated reductions in operating costs as a result of 25% reductions in N use.

Savings Realized Using 25% Less N On a 100 Acre Plot				
Crop	**N Rate**	**Costs (N at $0.40 / Lb)**		
	Lb / Acre	Full N Rate	25% Less N	Savings
Corn / Mint	225	$9,000	$6,750	$2,250
Snap Beans	135	$5,400	$4,050	$1,350
Ryegrass or Tall Fescue	135	$5,400	$4,050	$1,350
Broccoli	280	$11,200	$8,400	$2,800

There are reasons to undertake a nutrient management plan besides fertilizer cost savings and reduced environmental impacts. Often the methods of instituting a good management plan for nutrients will improve other aspects of growing high quality crops at lower costs. For example, it has long been known that winter cover crops that are disked into the ground before spring planting protects the soil surface, add organic matter, smother weeds and improve soil tilth (Sattell et al, 1998). Cover crops maintain organic matter content, reduce erosion on slopes and impede the formation of hard pan layers due to raindrop effects on bare earth. These effects will improve infiltration and water-holding capacity during the dry months of summer requiring irrigation. Besides improving soil quality, cover crops scavenge excess N left in the soil after fall harvest, reducing NO_3^- leaching to groundwater (Brandi Dohrn, 1997).

Periodically conducting a comprehensive soil sampling survey has many benefits. Estimates can be made of N content, other nutrients such as phosphorus (P) and potassium (K), and other important soil traits such as pH. These tests will be helpful in determining inputs of lime and possible changes in the chemical ratios of your typical fertilizer mix that may be needed to increase productivity. Having a complete understanding of nutrient content and availability within commercial fertilizers and organic fertilizers (animal manures and compost) is imperative to maximize crop production and minimize leaching losses.

Controlled Leaching Experiments – Cover Cropping

A controlled experiment studying the transport of agrochemicals has been in operation for a decade at Oregon State University's North Willamette Research and Extension Center (NWREC). The North Willamette lysimeter experiments began in 1992 and are still operating. These experiments predate the Lane County experiments previously described. Using controlled experiments enables researchers to assess the effect field management and weather conditions have on NO_3^- leaching. Similar to the Lane County measurements, the study revealed the vulnerability of Oregon's

groundwater to NO_3^- contamination, especially under fields left fallow during the winter. The study made a clear case for the effectiveness of cover crops to reduce groundwater contamination.

Twenty-six PCAPS lysimeters were installed carefully at a depth of 3.9 ft (1.2 meters) in seven separate 30 x 60 ft (9 x 18 m) plots. The cropping systems analyzed for leaching potential included summer row vegetables left fallow during winter (C plots) or summer row vegetables planted with a cover crop during the winter (H plots). Subplots within each cropping system were termed N0, N1 and N2 for no additions of fertilizer, half the recommended amount to maximize yields for particular crop, and the recommended fertilizer amount, respectively.

Sweet corn, snap beans, or broccoli were planted during the summer growing season. For the first eight years, cereal rye or Celia triticale, a hybrid of wheat and rye were planted as a cover crop on half the plots during the winter months. A mixture of Common vetch and triticale was used as a cover crop starting in winter 2000. Common vetch is a legume and commonly contains as much as 50 to 120 lbs or N per acre (56-135 kg of N per hectare) (Sattell et al., 1998). Nitrogen application rates were 225 Lbs / acre for corn, 135 Lbs / acre for beans, and 280 Lbs / acre for broccoli on the fully fertilized N2 plots. Figure B.8 shows the eight year flow-weighted average concentration of NO_3^--N in water collected by PCAPS below the various NWREC treatments during years using cereal cover crops. At the recommended rate of fertilization NO_3^--N concentrations exceeded the drinking water standard for both fallow and cover cropped systems (Figure B.8).

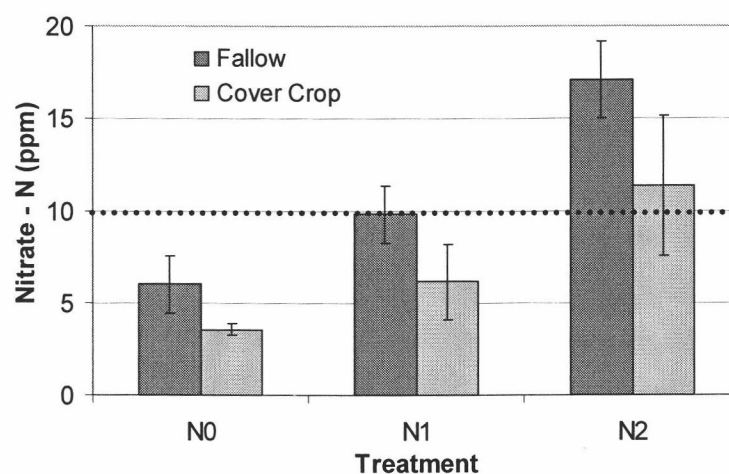Figure B.8 Flow-weighted NO_3^--N concentration in shallow groundwater at the North Willamette Research and Extension Center for the first eight seasons using cereal cover crops. Crops were fertilized at three different rates, N0 received none, N1 half, and N2 the recommended agronomic rate to maximize crop yield. The dotted line indicates the EPA 10 ppm drinking water standard. Error bars show the standard deviation.
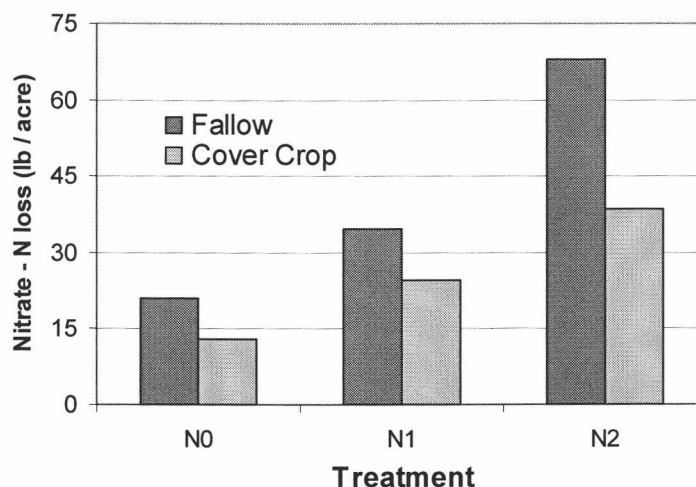
Figure B.9 Average NO_3^--N mass leaching rate for plots at the North Willamette Research and Extension Center during the first eight seasons using cereal rye cover crops. Crops were fertilized at three different rates, N0 received none, N1 half, and N2 the recommended agronomic rate to maximize crop yield.

Using measured concentrations and amount of water collected during PCAPS sampling, the mass of N lost to groundwater can be quantified. These N losses, expressed in lbs/acre/year (Figure B.9) can be compared directly to fertilizer applications. For fields receiving the recommended fertilizer rate (N2), 25% of all applied N was leached to groundwater and unavailable for plant use. For the plots at the NWREC, cover cropped plots reduced the NO_3^- contribution to groundwater by 40% over the fallow fields during the eight year period using cereal cover crops.

Sampling using the PCAPS gave NO_3^--N concentrations in soil water at a depth of 3.9 ft (1.2 m). To more precisely describe N content throughout the soil profile a soil sampling plan was designed. The data was collected in May of 2001. At the time of sampling, the common vetch / triticale mix cover crop was decomposing at the soil surface after being tilled back into the ground two weeks before. Two holes were augered at each of the NWREC subplots about 4.9 ft (1.5 m) from the buried PCAPS samplers, and each of the 3.9 ft (1.2 m) holes were divided into six increments composed of 20 cm of soil depth. The soil from each of the units was extracted in the lab to determine the NO_3^--N concentration.

As expected, the magnitude of NO_3^--N concentrations correlated with fertilizer inputs. Regardless of magnitude, however, the shape of the NO_3^--N distribution within the soil profile was dependent on winter cover crop treatment (Figure 10). The cover cropped fields had much higher concentrations near the ground surface, an indication that the cover crop had successfully scavenged nutrients from the soil during the preceding winter months. During the time of sampling, much of this organic material seemed to cause N mineralization, as there were elevated levels of NO_3^- in the upper soil column. The fallow fields showed a NO_3^- plume moving through the soil profile at an average depth of

2.6 ft (80 cm). From tracer tests completed and analyzed at the same time as the NO_3^- analysis, we know that this NO_3^- plume originated as post harvest NO_3^- remaining in the soil after the fall 2000 corn harvest. The NO_3^- under the cover cropped plots was available to the following summer crop, but the NO_3^- at deeper depths below winter fallow plots would largely be below the vegetable rooting zone.
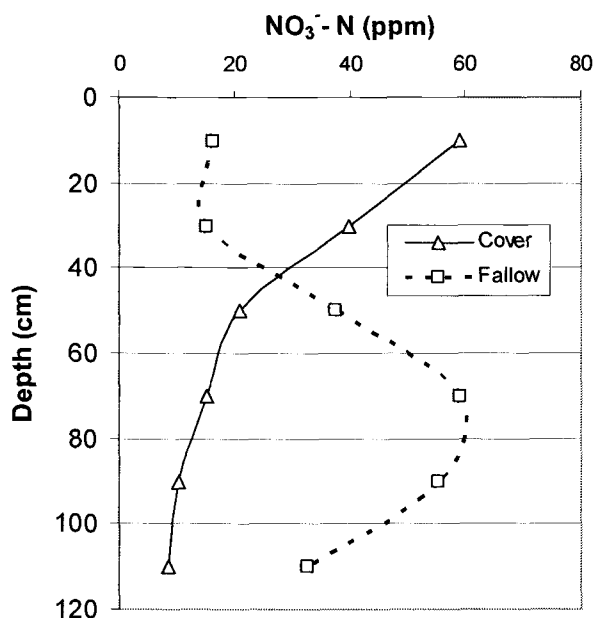
Figure B.10 Nitrate -N concentration profile in soil water at the North Willamette Research and Extension Center. Values are the average concentrations from the entire pool of N0, N1, and N2 fallow and cover cropped plots. Crops were fertilized at the OSU recommended level.

Other Management Practices for Reducing NO_3^- Leaching

There are still other ways to minimize the leaching of NO_3^- to groundwater. When considering such efforts, it is best to be assured that those things that are done to reduce NO_3^- leaching are also good for business: the grower wants to understand his/her soil and remedy any fertility problems to streamline their farming operation.
- Soil Testing
- Fertilizer and Manure Chemical Analysis
- Stem Nitrate Tests for Peppermint
- Maintenance of Irrigation Systems

Soil Testing

It is in the best interest for growers to periodically sample the upper layers of their soil and send the sample off to a laboratory for chemical analysis. Fertilizer applications can be fine-tuned to crop needs, reducing costs and groundwater contamination. The OSU central analytical laboratories suggest completing this process about every three years for perennial crops and yearly before planting annuals (OSU Extension, 2000). Included in a standard analysis is the determination of the correct mix and amount of fertilizer, as well as pH and liming requirements. Testing available N, phosphorus and potassium in the soil is an effective way to reduce the possibility of over-fertilizing. With this analysis, an appropriate N-P-K mix can be bought, eliminating unneeded additions. Physical analyses of soil properties can be an important consideration to assess NO_3^- leaching potential.

A physical analysis will provide a particle size distribution, water retention characteristics and the hydraulic conductivity of the soil. Water retention, the relationship between soil pressure and the amount of water in the soil, should be a consideration when preparing an irrigation schedule specific to a particular crop and soil type. With the test results, a grower can estimate the water content of the soil by measuring soil pressure, eliminating the need for guessing whether the crop is receiving sufficient water or has been over-irrigated.

Different soil sampling methods are used to prepare for each laboratory analysis. If you only require a nutrient analysis and/or particle size analysis, you can easily and quickly sample with an auger or a shovel. If you would like to have bulk density and/or water retention determined as well as nutrient analysis / particle size, then a coring device must be used to maintain soil structure. A good soil core can provide a wealth of information, especially if you are irrigating your crops.

To represent an entire field, it is important to take sub samples throughout the field and mix them into one representative sample. Field areas with varying drainage and slope can cause differences in soil quality, so these places should be sampled separately. Management history can greatly affect soil nutrient content. If you are going to change the growing crop or add new acreage to an existing field, these will be times sampling is in order. For more specific sampling procedures and strategies concerning sub-sampling, mixing and handling, see the OSU extension service pamphlet (Gardner and Hart, 2000).

Post-harvest NO_3^- testing is an effective way to ensure that the summer crop is using all or nearly all of the fertilizer applications during growth. The post-harvest NO_3^- test analyzes for NO_3^- left in the soil profile after the crop harvest. It is important that sample collection for this test take place during the season between rapid plant growth (corresponding with rapid N uptake) and the rainy leaching season. In general, large quantities of NO_3^- in the soil after a high-yield harvest indicate that the summer crop did not require the entire N application and that future application rates could likely be reduced.

Interpretation of a post-harvest NO_3^- test is dependent on the type of growing operation and fertilizer type used. Much work regarding post-harvest NO_3^- testing has been completed for dairy operations that require manure disposal and growing operations that use organic fertilizers as the predominant nutrient source. Operations using manure

are most likely to exceed recommended nutrient agronomic rates due to timing of applications and mineralization. Sullivan and Cogger (2002) provide a useful interpretation guide for using the post-harvest NO_3^- test for operations using manure.

Operations using commercial fertilizers may also use the post-harvest NO_3^- test, but results must be interpreted differently because detailed chemical analysis provided with the fertilizer allow much lower amounts of residual NO_3^- to be left in the profile compared to operations using manure. The test is still very effective at gauging nutrient application to plant requirements, especially if the test is done yearly so that application rates can be adjusted using the results.

Nutrient Content of Fertilizer Versus Manure

The post-harvest NO_3^- test has begun to expose us to the differences between commercial fertilizers and organic fertilizers such as manure and compost. When buying commercially prepared fertilizers, growers have the convenience of knowing exactly what they are applying on their field. By following the chemical analysis of the product and with the help of soil sampling, it is possible to fine-tune fertilizer applications in response to plant requirements. Manure, on the other hand, does not come with a chemical analysis and the minority of growers that use organic fertilizers instead of commercial fertilizers must rely on intuition and experience to gauge their fertilizer rates. In the case of manure, mineralization rates and thus N availability are hard to predict without a periodic analysis of the nutrient content in the material. In addition, manure is often diluted with water to different degrees, making applications at recommended agronomic rates is highly dependent on water content. To properly fertilize with manure it is important to consider the things outlined in Table B.4.

Table B.4 Important considerations when using organic fertilizer.

Fertilizing with manure

Manure Analysis -
       Water content
       Nitrogen content – organic, ammonium, NO_3^- portion
Application -
       Percentage of the manure will mineralize during the current season.
       Account for mineralized N from previous applications.
       Application method - Are there going to be ammonia losses?

One of the most important things to consider when using manure is that organic fertilizers mineralize slowly. In fact, some manure will actually supply more plant-available N two or three seasons after application than was supplied during the initial season. Consequently, it is easy for growers to supply N in rates well above the estimated plant requirement.

In the previously discussed Lane County study, two additional fields planted in certified organic vegetables were used to assess NO_3^- leaching contributions from organic growing operations. As with the mint and conventional vegetable fields, two PCAPS were installed per field. After the first two seasons, it was clear that NO_3^--N concentrations were very high. It was determined that poultry manure was applied in amounts well above recommended agronomic rates. After an estimation of N content and mineralization rates of the manure, applications were reduced. The management changes enacted in February 1996 resulted in apparent reductions in NO_3^--N concentration during the latter half of the study (Figure B.11).
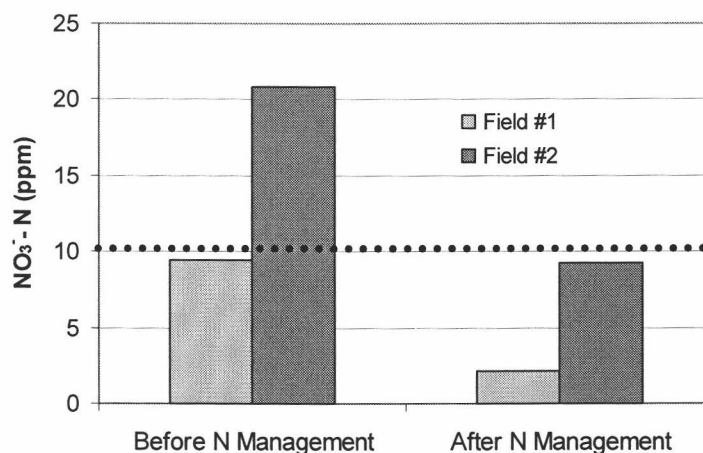
Figure B.11 Flow-weighted NO_3^--N concentrations for two fields planted in organically grown vegetables during the entire four-year study period. After revising manure application rates according to plant requirements, a considerable decrease in NO_3^--N concentration was observed. The dotted line indicates the EPA 10 ppm NO_3^--N drinking water standard.

Stem NO_3^- Tests for Peppermint

Due to the relatively high economic value and large N requirements of mint, it is a high risk crop for NO_3^- leaching. Optimum fertilization rates for peppermint have been determined to range between 180 to 300 lb/A, depending on the region. An N application rate of between 200 and 225 is recommended for western Oregon. Christensen (1996) showed that no increase in mint oil yield was achieved after N rates were increased from 225 to 320 lb/A. Fertilizer applications should be distributed at about 30 lbs/acre after harvest to stimulate roots, 40 lbs/acre after spring flaming, and the remaining 130 lbs/acre

when the post flame crop is growing in June (Jackson et al., 2000). Fertilization in excess of these amounts will not result in more plant growth because it has been shown that the crop has variable N uptake rates according to its stage of growth (Sullivan et al., 1999).

When using fertigation, a weekly stem tissue NO_3^- test is a management option. Developed about 30 years ago, the stem NO_3^- test uses a correlation between oil yields and NO_3^- concentration within the mint stem. Stem samples are taken throughout the period of fastest growth to ensure that NO_3^- concentrations within the mint stem do not fall below the critical values determined by the measured correlation. Using results from the stem NO_3^- test, N application rates can be matched to plant requirements. More information on the stem NO_3^- test is given in Brown (1982) and Smesrud and Selker (1998). Other extension publications describing N requirements and application schedules for mint and other crops include, *Nitrogen uptake and utilization by Pacific Northwest crops* (Sullivan et al., 1999) and *Peppermint growth and nutrient uptake* (Christensen et al., 1998).

Irrigation System Maintenance

Often, a good opportunity to reduce N losses from a growing operation is to ensure that irrigation systems are properly maintained and that fields are not over-irrigated. Applying excess water will force the grower to use more fertilizer than necessary. An appropriate irrigation schedule assures that water does not percolate past the root zone of the growing plants. Application rates are controlled by the potential evapotranspiration of the crop and growing stage of the crop. In the case that too much water is added, plant-available NO_3^- will be washed out of reach of the plant roots. Should this occur the grower will either need to make an additional application of fertilizer or risk a poor harvest.

The effects of an improperly maintained irrigation system on NO_3^- leaching potential are similar to over-irrigation. An improperly maintained irrigation system will not distribute water evenly throughout the field. Uneven irrigation can set of a chain of events that effectively cause the grower to require additional fertilizer applications. Consider an irrigation system that applies water in excess to certain parts of the field and deficiently in others. Portions of the field receiving the bulk of this water will have higher NO_3^- leaching rates. Other parts of the field will be under-watered. In response, the grower will increase the irrigation amount to ensure that the drier areas are receiving enough water. Now the leaching potential in the wetter areas will further increase. Nitrogen deficient plants in the over-irrigated areas will require additional fertilizer, assuming that the grower is able to realize the problem in time to counteract the deficiency. Crops in the drier areas of the field will have a N surplus.

An effectively managed irrigation scheme considers the following:

- **Operating pressure** - Sprinkler discharge and wetted diameters are a function of water pressure and nozzle diameter, therefore application rates and sprinkler uniformity will be dependent on the operating pressure of the system. Ensure that gauges throughout the system are operational so that ideal operating pressures for the system can be maintained.

- **Nozzles** - Ensure that nozzles are of a consistent make and diameter and are not worn. Periodically (at least every four years) replacing all the nozzles at once is recommended to maintain uniform application.

- **Dual nozzle sprinklers** -If using a standard 40' x 60' sprinkler spacing design, dual nozzle emitters should be used as they are known to give more uniform coverage.

- **Soil water monitoring** - The goal of a fine-tuned irrigation system is to maintain soil water levels so that the growing crop can effectively utilize water without waste and minimize the leaching of nutrients. With the aid of soil water monitoring, irrigation schedules can be adjusted to ensure that soil water content in the root zone is maintained in an appropriate range. Technologies becoming less expensive can use changes in soil moisture levels to relay information to turn on and operate irrigation equipment.

- **Schedule irrigation based on expected ET** - Irrigation requirements vary throughout the year as potential evapotranspiration (ET) rates change with the weather. After determining the soil's available water capacity, the rooting depth of the growing crop and the degree to which the soil can dry between irrigation events, an irrigation schedule can be developed. Assuming a constant discharge throughout the system during the entire year (check nozzles and monitor the pressure!), set times of irrigation events can be adjusted as ET rates change to apply an irrigation depth that will recharge the water holding capacity of the soil. Monthly irrigation schedules should be validated by soil water monitoring to ensure that plant requirements are being met.

Fortunately, replacing the sprinkler nozzles on irrigation equipment is an inexpensive and often the most successful way to improve spray uniformity. Figure B.12 shows the pressure distribution of a system with old or mismatched sprinkler nozzles and the same system after nozzle replacement. Pressure distribution and spray uniformity were greatly improved with this small amount of work.
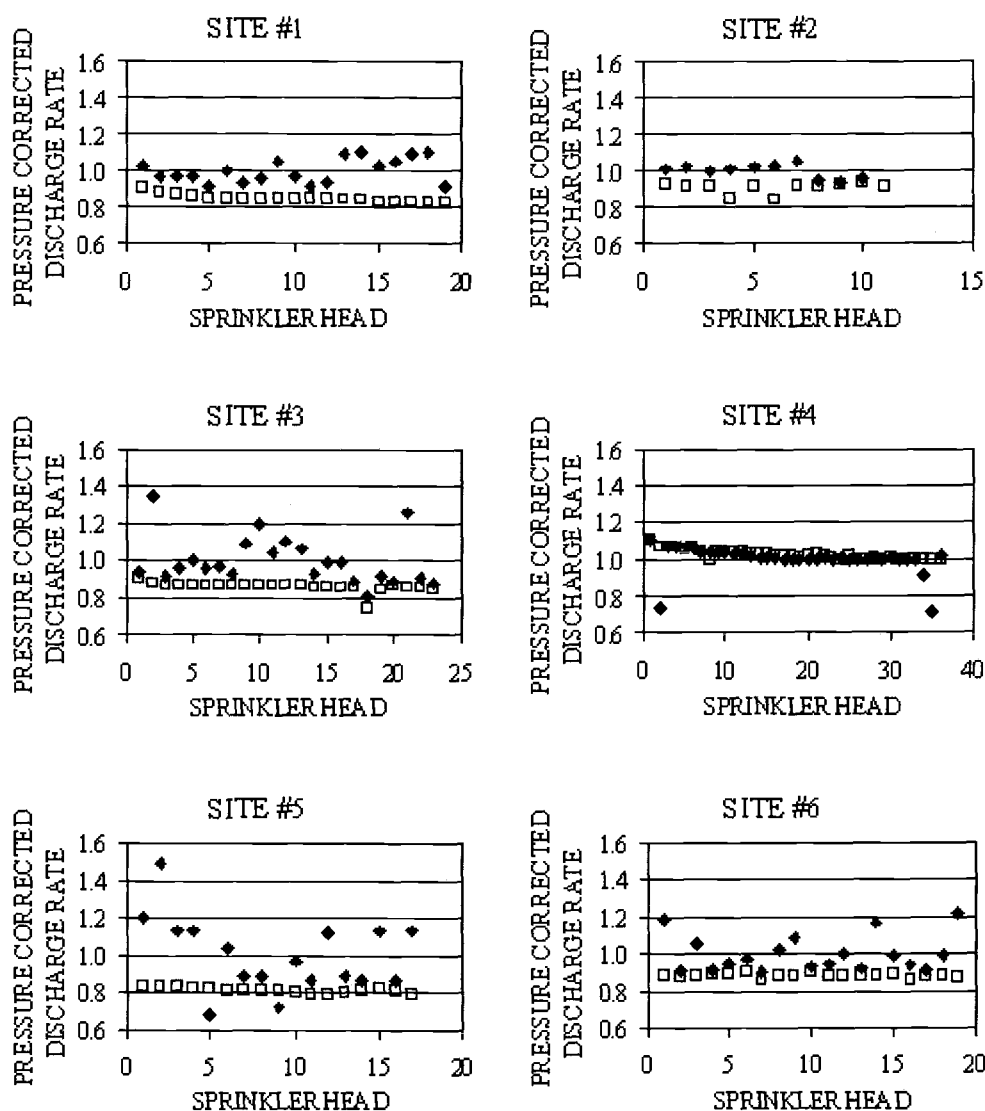
Figure B.12 Sprinkler discharge rates before and after nozzle replacement. Solid diamonds show discharge rates before nozzle replacement while the hollow squares show post-replacement discharge rates. More uniform application was achieved using replaced nozzles, likely reducing field susceptibility to NO$_3^-$ leaching. Figure from Louie and Selker (2000).

Several extension papers discuss methods to streamline irrigation operations. The *Western Oregon Irrigation Guides* (Em 8713 (Smesrud et al., 2000)) provide worksheets to calculate irrigation set times based on potential evapotranspiration for 16 different commonly grown crops. *Soil Water Monitoring and Measurment* (PNW 475 (Ley et al., 1994)) is a helpful guide to understanding soil water potential. Sprinkler head maintenance and its relationship to application uniformity is discussed in Louie and Selker (2000). The post-harvest water requirements of mint are presented in Smesrud

and Selker (1999). A great on-line source for papers and information concerning many aspects of agriculture and the environment is the Oregon State University extension and experiment station communications, located on the World Wide Web at http://eesc.orst.edu/default.html.

Conclusions

Nitrate contamination of groundwater is an issue that is impossible for growers in Oregon and around the country to ignore. As our increasing population becomes ever more dependent on groundwater for domestic use, protecting this drinking water resource is imperative. The first step to address the NO_3^- leaching problem is to become aware of the risk. Table B.5 is a general list to check when assessing the potential for N leaching from the farm. There are many effective nutrient management plans that have reduced N loss from both experimental and privately owned growing operations.

Table B.5 Key issues to consider when assessing the potential for NO_3^--N leaching.

Nitrate Leaching Potential

Amount of Residual Nitrogen Left After Harvest -
- ▲ Amount: nitrogen application in excess of crop recommendations
- ▲ Timing: nitrogen applications after crop ceases to assimilate nitrogen for growth
- ▲ Uniformity: inefficient nitrogen application methods

Irrigation Practices -
- ▲ Non-uniform irrigation due to incorrect operating pressures and/or worn or mismatched nozzles
- ▲ Over-irrigation in response to excessively dry or poor production areas
- ▲ High background nitrates in irrigation water

Mineralization of Organic Materials -
- ▲ Untimely decay of nitrogen-rich dry matter or manure

Landscape and Climate -
- ▲ Rainfall exceeding potential evapotranspiration for extended periods
- ▲ Rainfall while fields are not covered in standing crop
- ▲ Additional water saturating field from other areas (overland flow)

Many management options to reduce susceptibility to NO_3^- leaching can be implemented with little costs, and may even reduce operational costs in the long-term. Maintaining irrigation systems, determining nutrient needs through soil sampling and using cover crops may increase yields, reduce N losses, and decrease water use, enabling more irrigated acreage. Researchers continue to work in the field and cooperate with growers, a relationship that has and will continue to provide improvements for managing non-point source nutrient contamination. Using county extension is a helpful resource to keep up on current information that can assist growers in managing a sustainable, productive and profitable business. With cooperation, groundwater and soil resources will be conserved for years to come.

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Appendix C: CD ROM of all raw data included in this analysis (Attached to back cover)