AN ABSTRACT OF THE THESIS OF

Patrick M. Shelby for the degree of Master of Science in Bioresource Engineering presented on April 20, 1995.
Title: Assessment of Ground Water Recharge and Quality Under Agricultural Production in Lane County, Oregon.

Redacted for Privacy

Abstract approved: ______________________________
John S. Selker

Assessment of the environmental impacts of an agricultural production system requires information on both soil water quality and solute flux. Passive Capillary Samplers (PCAPS), which sample water from the vadose zone using fiber glass wicks, have shown potential to provide both flux and solute concentration in unsaturated zone sampling but have not been tested under long-term, natural, rainfall conditions. The objectives of this study are to (1) evaluate PCAPS operation under non-steady, natural rain and irrigation fed conditions, (2) determine the samplers ability to estimate recharge, and (3) estimate the loss of nutrients resulting from agricultural production.

32 PCAPS and 78 suction cup samplers were installed below the root zone at 16 commercial fields in Lane County, Oregon. PCAPS' were installed in positions using ground penetrating radar such that PCAPS' were placed in homogeneous or concave profile locations. Two PCAPS and six suction cups were installed at each site. Rain gages and TDR probes were installed at eight of the 16 sites. These data were used to develop a mass balance for each of the eight special study sites. Comparison to mass balance data indicates that the PCAPS flux measurements were within 10% of the mass balance estimated recharge. Surface runoff of potential drainage water during periods of high rainfall was a point of concern for estimated recharge discrepancies because runoff was not measured. The saturated hydraulic conductivity was shown to be the most

influential design parameter for matching wick and soil types. On the other hand, the incident flux, rather than conductivity, determined the ultimate ground water recharge. PCAPS collection was found to be significantly correlated (average $R^2 = 0.75$) to the mass balance monthly estimated recharge. To estimate the mean monthly recharge at each site with a 30% bound on the mean and 95% confidence level, 20 PCAPS would be required at each site.

PCAPS were found to be superior to suction cup samplers for estimating ground water recharge concentrations because PCAPS were able to sample both flux and resident concentrations. Mint and row crop, organic and inorganic, production systems contributed to the largest adverse environmental impacts with average recharge concentrations for mint and row crop of 24 mg L^{-1} and 28 mg L^{-1}, respectively. Orchard and blueberry production systems had little impact with their seasonal concentrations averaging below the EPA water quality standard. Amounts of percolation were key in determining which management systems were inefficiently operated. Over-irrigation during the summer lead to increased losses of nitrogen for the mint production systems in the summer as well as the winter. Over-fertilization was important for creating significant differences in seasonal mass losses of nitrogen from row crop production systems. Overall, the PCAPS estimated nitrogen loss was 12% lower than that calculated using a simplified nitrogen mass balance approach. Best management practice suggestions concerning irrigation, fertilization and cover cropping were provided as a direct result of the findings of the project. With technical support and increase in concern over nitrate contamination, farmers should be able to control leaching losses without the use of quotas or allotments.

Assessment of Ground Water Recharge and Quality Under Agricultural Production in Lane County, Oregon

by

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Major Professor, representing Bioresource Engineering

Redacted for Privacy

Chair of Department of Bioresource Engineering

Redacted for Privacy

Dean of Graduate School

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Patrick M. Shelby, Author

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Chapter I. Introduction

Intensified agricultural management practices in the past few decades have caused concern over the possibility of increased losses of nitrate-nitrogen (NO_3^--N) below the root zone resulting in a potential health risk. The use of increased supplemental irrigation during the growing season (Irrigation Journal, 1979) combined with the heavy rainfall during the winter in the Northwest United States creates a greater opportunity for the loss of excess nitrogen throughout the year. The USEPA's 1990 national survey of drinking water wells indicates that nitrate (NO_3^-) was the most commonly found contaminant with 57% of the rural wells and 52% of the community water supplies, respectively, containing detectable concentrations. In 1988, 21% of 136 wells tested in the Willamette Valley of Oregon showed concentrations of NO_3^--N above 10 mg L^{-1} which is the EPA water quality standard (Pettit, 1988). In 1994, a voluntary well water study carried out in Lane County, Oregon in parallel with this study found 21% of 281 wells in the agriculturally active portion of the county to have values greater than 10 mg L^{-1} NO_3^--N.

To assess agriculture's contribution to NO_3^- pollution of groundwater, leaching losses must be measured accurately. If leaching losses are accurate, the sampling procedure must also be able to monitor the recharge rate. NO_3^- is completely soluble in water and is transported at the rate of the soil solution flux. To estimate the quantities of NO_3^- lost to the ground water, recharge volumes and concentrations must therefore be monitored directly under agricultural production. A relatively new form of soil solution sampler, Passive Capillary Samplers (PCAPS), which sample water from the vadose zone using fiber glass wicks, have shown potential to provide both flux and solute concentration in unsaturated zone sampling but have not been tested under long-term, natural, rainfall conditions. For this study, 32 PCAPS and 96 suction cup samplers were installed below the root zone at 16 commercial fields in Lane County, Oregon. By studying 16 different sites which incorporated five different management systems and eight different soil types, the PCAPS ability to estimate agricultural recharge and leaching losses was evaluated. The development of the proper sampling method for monitoring the groundwater recharge and quality will allow Lane County, and other areas, to determine the influence of agricultural production systems on the quality of their water.

Chapter II. Materials and Methods

Characterization of Sites

The experiments were carried out at each of 16 separate sites located throughout Lane County, Oregon. Results of the experiments are specific to the sites chosen, but the spatial distribution of the sites throughout the county allow for a wide comparison of results. The experiments evaluate the major cropping systems employed in the region. The cropping systems are listed in Table 1 and sites will be referred to according to crop type and number. Sites were chosen with the cooperation of local farmers and based on 1992 agricultural commodity sales in Lane County. The highest earning crops per acre and most economically significant crops were chosen for the experiments. The spatial distribution of farms in the county are illustrated in Figure 1. The wide variety of soil and crop types will allow other researchers to compare our results to similar experiments. This information will also serve as a reference for future research at the same sites.

Table 1. Cropping systems chosen for experiments with acerage and harvest value for each in Lane County.

Crop System:	Number of Sites:	Total Acreage:	Value/Acre:
Peppermint	4	6,450	$1020
Vegetable row crop	4	7,095	$1340
Organic vegetables	2	≈ 1,000	$2000
Perennial Rye Grass seed	2	8,000	$440
Blueberries	2	150	$5600
Tree fruits	2	3,900	$860

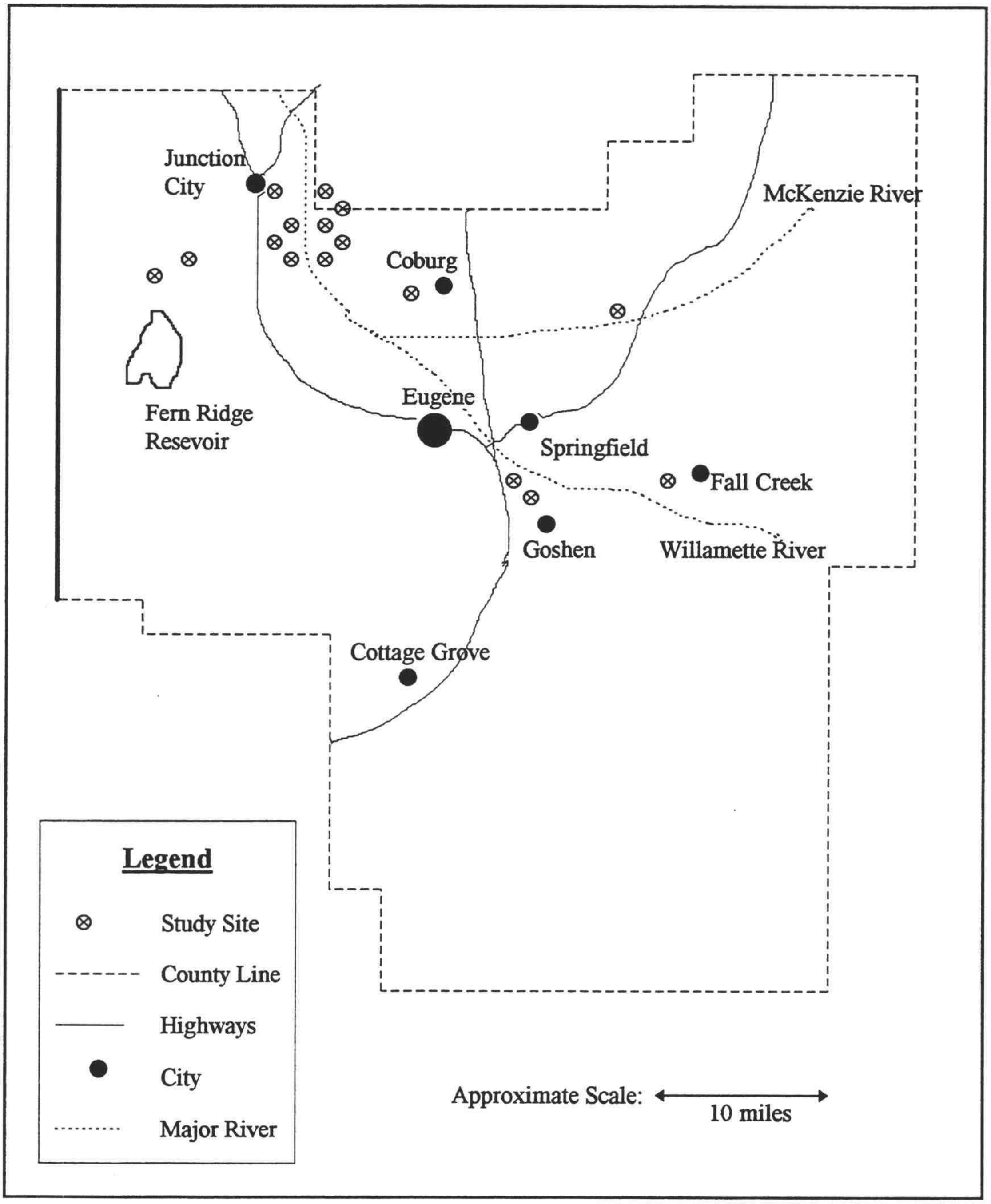

Figure 1. Map of eastern half of Lane County depicting major cities, highways, rivers, and distribution of study sites.

Soil Description

There are a total of eight different soil types for all 16 sites. The classification of soils are based upon analysis of soil profiles during sampler installation at each of 16 sites, Lane County soil survey information, and laboratory analysis. A list of the soil series, taxonomy and geologic parent materials are given in Table 2. Soil samples were taken from all sites and analyzed for some basic properties (Appendix A).

Table 2. Experimental sites, soil series, soil taxonomy and geologic parent materials.

Site	Soil Series	Taxonomic Class	Parent Material
Grass Seed #1	Coburg silty clay loam	Pachic Ultic Argixerolls	silty and clayey alluvium
Grass Seed #2	Awbrig silty clay loam	Vertic Albaqualfs	silty and clayey alluvium
Organic #1	Newberg loam	Fluventic Haploxerolls	recent silty alluvium
Organic #2	Malabon silty clay loam	Pachic Ultic Argixerolls	silty and clayey alluvium
Blueberry #1	Cloquato silt loam	Cumulic Ultic Haploxerolls	recent alluvium
Blueberry #2	Newberg fine sandy loam	Typic Haploxerolls	recent alluvium
Orchard #1	Newberg fine sandy loam	Typic Haploxerolls	recent alluvium
Orchard #2	Fluvents, nearly level		sediment deposits
Mint #1	Newberg loam	Fluventic Haploxerolls	recent silty alluvium
Mint #2	Chehalis silty clay loam	Cumulic Ultic Haploxerolls	recent alluvium
Mint #3	Newberg fine sandy loam	Typic Haploxerolls	recent alluvium
Mint #4	Malabon silty clay loam	Pachic Ultic Argixerolls	silty and clayey alluvium
Row Crop #1	Newberg fine sandy loam	Typic Haploxerolls	recent alluvium
Row Crop #2	Newberg loam	Fluventic Haploxerolls	recent silty alluvium
Row Crop #3	Malabon silty clay loam	Pachic Ultic Argixerolls	silty and clayey alluvium
Row Crop #4	Malabon silty clay loam	Pachic Ultic Argixerolls	silty and clayey alluvium

Soil cores were taken from each site and analyzed for bulk density, particle size as outlined by Gee and Bander (1986) and K_{sat} using the constant head tempe cell as outlined by Klute and Dirksen (1986).

Field saturated hydraulic conductivity was measured using a constant head well permeameter as described by Amoozegar (1989). The steady-state flow rate, Q, was determined by fitting a linear regression to the volume of infiltration versus time plot once steady infiltration was achieved. Q is the slope of the linear regression equation. Using the estimate of Q, K_{sat} was then determined using the Glover solution as suggested by Amoozegar and Warrick (1986)

$$K_{sat} = \frac{CQ}{\left(2\pi H^2 + \pi r^2 C + 2\pi H/\alpha\right)} \tag{1}$$

where H is the constant ponding depth in [L], r is the radius of the well in [L], α is the ratio of field saturated conductivity to matrix flux potential based on soil structural/textural considerations in [L^{-1}], and C is the dimensionless shape factor. The value for α was chosen as 12 m^{-1} which was suggested for most structured soils and medium and fine sands by Elrick et al. (1989). The shape coefficient, C, is given by Zangar (1953)

$$C = \sinh^{-1}(H/r) - \left(r^2/H^2 + 1\right)^{1/2} + r/H \tag{2}$$

Water retention (Appendix A) was determined according to Klute (1986), van Genuchten (1980), and Arya and Paris (1989). Volumetric moisture content was determined gravimetrically using soil cores as well as measured using TDR probes.

Climate

The climate of Lane County is dominated by winds from the Pacific Ocean. Winters and summers are mild with hot days and snow and freezing temperatures rare. The climate is classified as temperate oceanic. Climatic data for the region has been recorded for the last 30 years at the Eugene Airport (Figure 2). Unfortunately, no research and experiment stations are located within Lane County. This creates problems when long-term evaporation and solar radiation data are needed. Yet, Corvallis is located within 30 miles of the nucleus of the test sites. Little differences in the climate of the Willamette valley can be documented, thus climatic data from the Hyslop research station in Corvallis were used for the experiments.

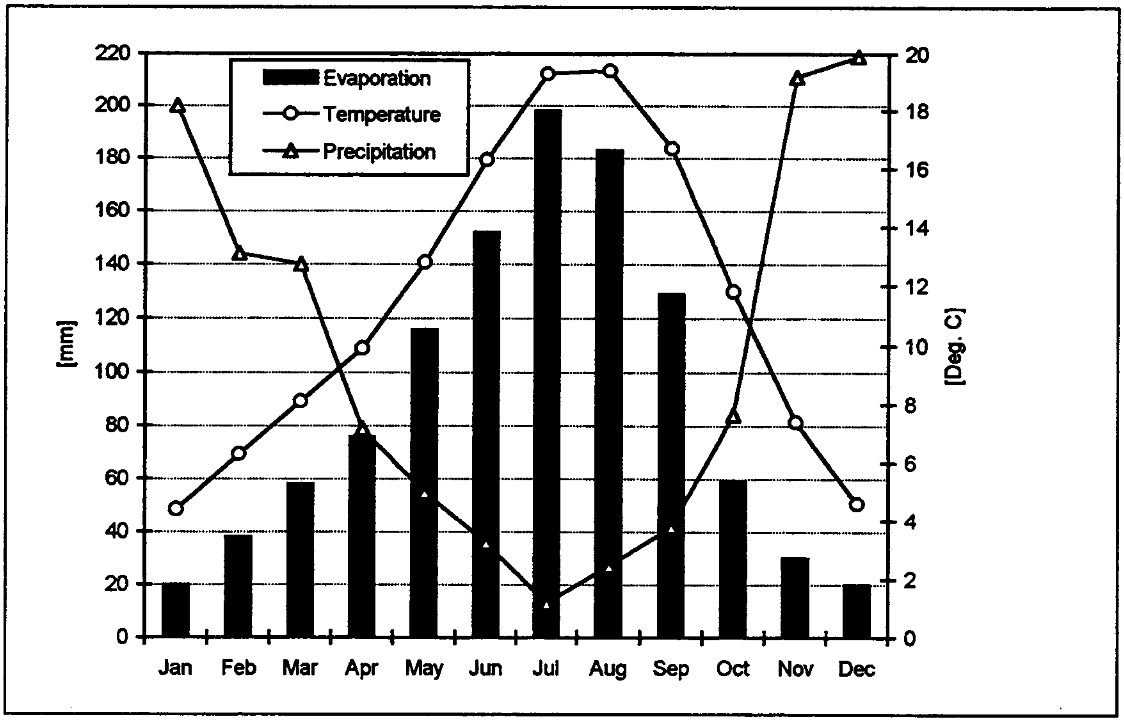

Figure 2. Climatic data at both the Eugene Weather Center and Hyslop Experiment Station 1961-1991 (courtesy of the Oregon Climate Service).

Monthly temperatures during the winter months (Nov. - Apr.) average 6.8°C with monthly average rainfall at 166 mm. In stark contrast, monthly temperatures during the summer months average 16.1°C and monthly average rainfall of 43 mm, which typically results in higher evapotranspiration (140 mm per month) than precipitation. For these reasons, the climate is also classified as a mesic moisture regime. Average total annual precipitation measures 1250 mm, with an average annual temperature of 11.4°C.

Precipitation was measured with a nonrecording gauge at the Eugene Weather Center. For the first year of the project, eight of the 16 sites were chosen for instrumentation with six nonrecording rain gauges (250 mm capacity). After the first year of the project, all sites were instrumented with at least 2 nonrecording gauges. Measurements were corrected by +2% to account for expected error introduced by the average wind speed (Larson and Peck, 1974).

Due to the lack of an experiment station in Lane County, evapotranspiration, required for a water balance, was calculated using the modified Penman-Monteith equation (Penman, 1948; Monteith, 1965). The Penman method is a widely used theoretically based method for estimating evapotranspiration. The Penman equation is based on a combination of aerodynamic and energy budget relationships. The combination of these meteorological relationships essentially eliminates any surface conditions which may effect evapotranspiration. The result is the estimation of evaporation only in terms of the atmosphere or measurable meteorological parameters. The energy budget for evaporation was first discussed by Bowen (1926)

$$R_n = LE + H + G \tag{3}$$

where R_n is the net radiation in [W/m^2], LE is the latent heat flux in [W/m^2], H is the sensible heat flux in [W/m^2], and G is the soil heat flux in [W/m^2].

Penman combined Bowen's equation with an equation by Dalton (1801) which related evaporation to the water vapor deficit and a wind function. Monteith (1965) expanded on the idea of Penman by employing resistance terms, introduced by Penman (1963), into the Bowen energy balance. The two resistance terms are the surface or stomatal resistance and the aerodynamic resistance. The surface resistance is the resistance of evaporation from the plant stomatal surface. When the plant is fully transpiring and the leaf surface is wet, the surface resistance is low. The resistance increases as the plant surface begins to dry out. The surface resistance depends on the vegetation type and a number of atmospheric and hydrological variables which affect the supply of and demand for water. Typically, for growing crops, there seems to be an agreement on 40 - 60 s m^{-1} (Thompson et al., 1981). The aerodynamic resistance is the resistance of water vapor flow from any evaporative surface

$$r_a = \frac{1}{ku_*} ln\left(\frac{10}{z_m}\right) \tag{4}$$

where k is the von Karman constant, usually taken as 0.4, u_* is the friction velocity in [L/T], and z_m is the roughness length in [L]. The aerodynamic resistance governs the transport of heat and vapor within and out of the plant canopy. These two resistance terms were essential to the modification of the Penman equation by Monteith.

The PENMET4 (Martinez-Cob and Carrijo, 1988) computer model is a program to compute reference evapotranspiration using the modified Penman-Monteith equation. The Meteorological Office Rainfall and Evaporation Calculation System (MORECS) Penman-Monteith model (Thompson et al., 1981) is the original computer model which was altered by Martinez-Cob and Carrijo (1988). MORECS is a subroutine of the model developed by Carrijo et al (1988). The PENMET4 model was used to calculate reference evapotranspiration for the data calculation period. The MORECS model used in the PENMET4 program uses the form of the Penman-Monteith equation modified by Thompson et al. (1981) to account for errors caused by differences in temperature at the surface and a reference level above the canopy

$$\lambda \cdot E = \frac{\Delta(R_N - G) + \rho \cdot c_p(e_s - e) / r_a}{\Delta + \gamma \cdot (1 + r_s / r_a)} \tag{5}$$

where E is the rate of water loss in [M L^{-2} T^{-1}], Δ is the rate of change of saturated vapor pressure in [P T^{-1}], R_N is the net radiation in [E L^{-2}], G is the soil heat flux in [E L^{-2}], ρ is the air density in [M V^{-1}], c_p is the specific heat of air at constant pressure (1005 J kg^{-1}), e_s is the saturation vapor pressure at reference height in [P], e is the vapor pressure at reference height in [P], b′ is a correction value, λ is water's latent heat of vaporization (2,465,000 J kg^{-1}), γ is the psychrometric constant (0.66), r_s is the bulk surface resistance in [T L^{-1}], and r_a is the bulk aerodynamic resistance in [T L^{-1}]. The program requires the input of minimum and maximum daily temperature, the minimum and maximum daily relative humidity, average daily solar radiation, average daily wind speed, time of measurements, and elevation of measurements.

Management

Table 3 displays the history of the conventionally and alternatively managed sites. There is no experimental design for each site because only private farmers were used under existing practices. Seeding and harvest dates for the period of the experiment (1993 - 94) are given in Table 4. Each site received fertilizer amounts at the discretion of the private farmer. Fertilizer type, rate and time of application are displayed in Appendix B.

Table 3. History of experimental sites.

Site	1990	1991		1992		1993		1994
	Fall	Spring	Fall	Spring	Fall	Spring	Fall	Spring
Grass Seed #1	mint	mint	mint	mint	mint	mint	rye grass	rye grass
Grass Seed #2	rye grass	rye grass	rye grass	rye grass	rye grass	rye grass	rye grass	rye grass
Organic #1	cover	mixed veg.	cover	mixed veg.	cover	mixed veg.	cover	mixed veg.
Organic #2	cover	veg.	cover	veg.	cover	lettuce	cover	cover
Blueberry #1	blueb.	blueb.	blueb.	blueb.	blueb.	blueb.	blueb.	blueb.
Blueberry #2	pasture	blueb.	blueb.	blueb.	blueb.	blueb.	blueb.	blueb.
Orchard #1	apple	apple	apple	apple	apple	apple	apple	apple
Orchard #2	peach	peach	peach	peach	peach	peach	peach	peach
Mint #1	mint	mint	mint	mint	mint	mint	mint	mint
Mint #2	mint	mint	mint	mint	mint	mint	mint	mint
Mint #3	wheat	wheat	mint	mint	mint	mint	mint	mint
Mint #4	mint	mint	mint	mint	mint	mint	mint	mint
Row Crop #1	rhubarb	rhubarb	wheat	sweet corn	wheat	sweet corn	fallow	sweet corn
Row Crop #2	cover	sugar beet	cover	beans	cover	sweet corn	cover	sugar beet
Row Crop #3	mint	mint	mint	mint	wheat	wheat	sugar beet	sugar beet
Row Crop #4	fallow	beans	sugar beet	sugar beet	wheat	wheat	fallow	sweet corn

Table 4. Times of seeding and harvest for each experimental site.

Site	Seeding	Harvest
Grass Seed #1	October 15, 1992	July 15, 1993
Grass Seed #2	October 15, 1992	July 8, 1993
Organic #1	May - August, 1993	June - December, 1993
Organic #2	May - August, 1993	June - December, 1993
Blueberry #1	Spring, 1986	June - July, 1994†
Blueberry #2	Spring - Fall, 1991	July, 1994†
Orchard #1	Spring, 1985	October, 1993†
Orchard #2	February, 1990	October, 1993†
Mint #1	October, 1990	August 15, 1993†
Mint #2	Fall, 1990	August 8, 1993†
Mint #3	October, 1991	August 18, 1993†
Mint #4	September, 1990	August 10, 1993†
Row Crop #1	May 6, 1993	August 20, 1993
Row Crop #2	June 15, 1993	October 1, 1993
Row Crop #3	October 15, 1992	August 10, 1993
Row Crop #4	October 15, 1992	August 15, 1993

† Seeding date represents initial seeding or planting of crop with harvest the same time each following year

PCAPS

Construction

A 170 L (33 x 87 x 62 cm) custom molded 15-kg epoxy coated fiberglass box serves as the frame for the sampler (Figure 3). The frame is able to withstand greater than 1000 kg vertical load. A stainless steel panel (1 mm thick, 32 x 86 cm, and edges raised 1.75 cm) was fitted into the constructed step in the wall at the top opening of the fiberglass box. The panel is subdivided into three 31 by 29-cm sections, where three wicks are placed. In the center of each section, a hole was punched and a 31.6 mm I.D. alloy 304 stainless steel pipe was pushed through the hole. A single 60 L custom molded High Density Polyethylene (HDPE) sampling vessel (24 x 78 x 32 cm), selected for its lack of chemical adsorption (Topp and Smith, 1992), was fitted to the bottom interior of the fiberglass box. Three wick access holes and one HDPE sample access tubing hole were made in the vessel.

Two types of wicks, a braided 2.93-cm medium density and 2.48-cm high density Amatex fiber glass wicks (#10-863KR-08 and #10-864KR-08, Amatex Co., Norristown, PA), were used for the experiments (Table 5). The wicks were limited to a maximum fiber length of 80 cm due to the molded fiberglass box. The top 20 cm of the wick were unbraided into single strands and then cleaned according to Knutson et al. (1994). The medium density wicks were combusted in a kiln at 400°C for 12 hours while the high density wicks had to be combusted at 1000°C for 12 hours, to insure all impurities were removed. The combusted wicks were spread out radially on the top stainless steel panel of the sampler. The end of each wick strand was glued down at the edge of the panel with a single drop of silicone sealant.

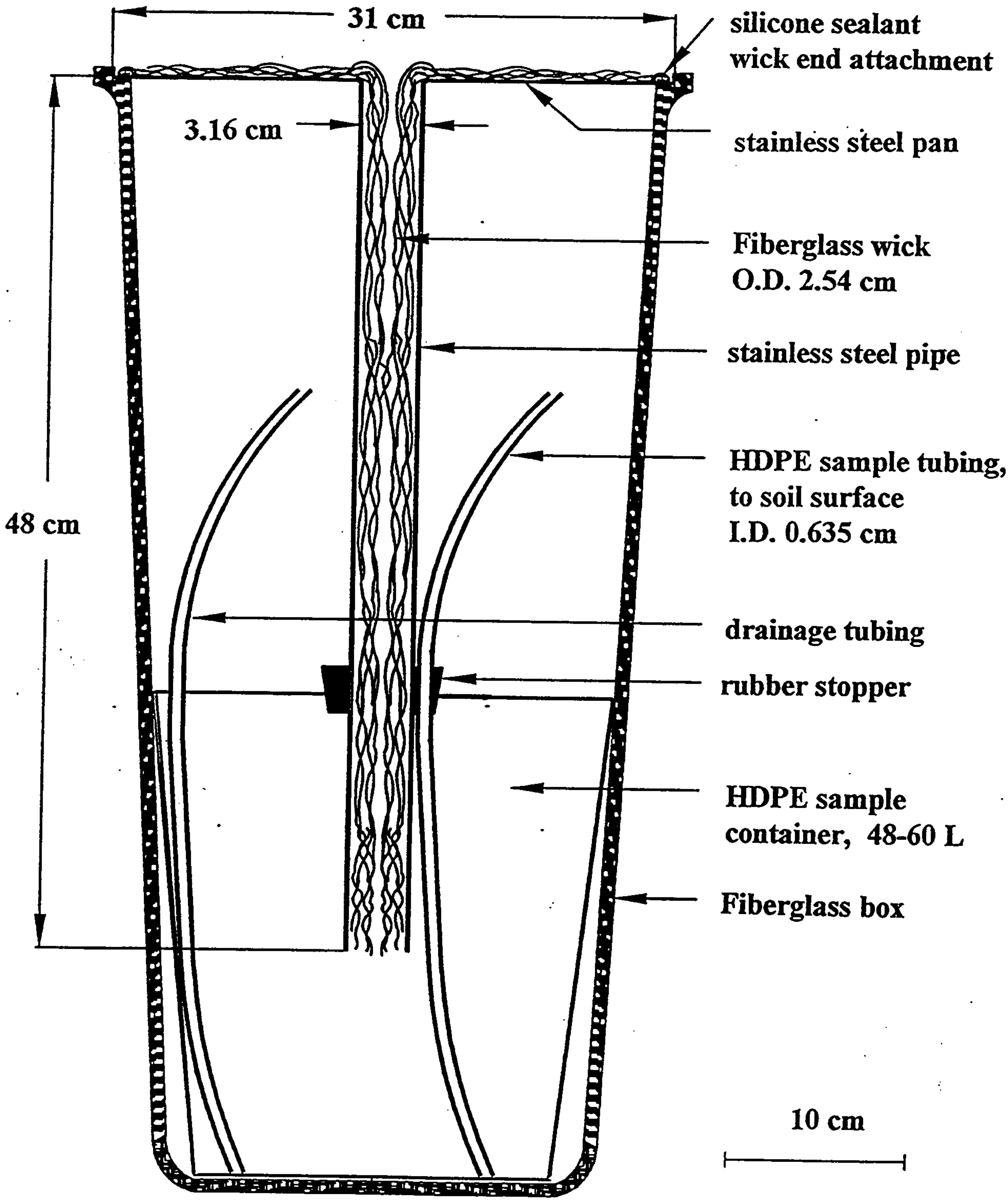

Figure 3. Crossectional view of PCAPS design (drawn to scale, adapted from Brandi-Dohrn, 1993).

After the wicks were in place, silicone sealant and a black rubber stopper were used to fit the pipe and sample tubing into the vessel, to ensure that leachate entered via the wick only. A 5.72-cm hole was drilled in the side of the fiberglass box and the sample access tubing and drainage tubing for the fiberglass box, built in to allow removal of water in case of leaks, were run through a rubber stopper. The sampler access hole and top stainless steel panel were set in place and sealed with silicone sealant to prevent flooding. The sampler was built to be used for an indefinite time period, and since only non-adsorbing materials (fiberglass, HDPE, stainless steel) were used, the sampler is well suited for the monitoring of agrochemical leaching.

Installation

Two PCAPS were installed at each experimental site. Figure 4 depicts the set-up of a typical special study site. Ground penetrating radar (GPR) was used to determine proper sampler locations at the sites. Strong reflections at soil interfaces where adjacent soil layers have sharply differing dielectric constants can be identified (Kung et al., 1991). A Geophysical Survey Systems, Inc. SIR10A GPR with 100 and 500 Mhz antennas was used at each site. Several passes with each antenna were made over each area in the field that was initially selected for sampler placement with the cooperation of the farmers. Soil samples along with GPR transects were compared to get an idea of depth of penetration and soil strata. Areas were identified as ideal and not ideal for PCAP placement. Areas of exclusion included those that did not have homogeneous profiles and those with sloping soil interfaces which may migrate water away from the samplers.

Table 5. Wick types used in the experiment.

Site	Wick Type
Organic #1	Medium Density
Organic #2	High Density
Row Crop #1	Medium Density
Row Crop #2	High Density
Row Crop #3	High Density
Row Crop #4	High Density
Mint #1	Medium Density
Mint #2	High Density
Mint #3	High Density
Mint #4	High Density
Orchard #1	Medium Density
Orchard #2	High Density
Blueberry #1	Medium Density
Blueberry #2	High Density
Grass Seed #1	High Density
Grass Seed #2	High Density

To install the PCAPS, the area in the field chosen for installation was cleared of crops. An area of approximately 10 ft. by 10 ft. was necessary for proper installation. A back hoe was used to dig an 8 ft. by 4 ft. wide by 8 ft. deep trench. The PCAPS were installed off the side wall of the trench so that the tops of each sampler were just below the root zone of the plants under undisturbed soil. The undisturbed soil was critical so as not to disrupt the natural flux present in the field. A tunnel was dug in the side of the trench for the installation of each PCAPS. Typically the top of the tunnel was between 2 and 3 feet of the surface while the bottom of the tunnel was between 5 and 6 feet below ground level (Table 6). The top of the tunnel was undisturbed and flat so as to achieve optimal PCAPS sampling. The top panel of the PCAPS was filled with slightly compacted native soil with an extra layer above the panel to avoid any eventual gaps from forming. The PCAPS were placed in the side wall tunnel and elevated using wooden wedges into the top of the tunnel to insure close contact with the soil (Figure 5). Two PCAPS were installed in each trench. As the trench was backfilled, the samplers were hydraulically sealed in the side tunnel using bentonite. Tubing to drain samples from each PCAPS was run to an irrigation box placed along the side of the field at field level. The trenches were refilled and soil recompacted to avoid any settling or swelling. Some settling and swelling has been observed in the past year but is mainly associated with soils having high clay fractions. Installation was completed on September 1, 1993.

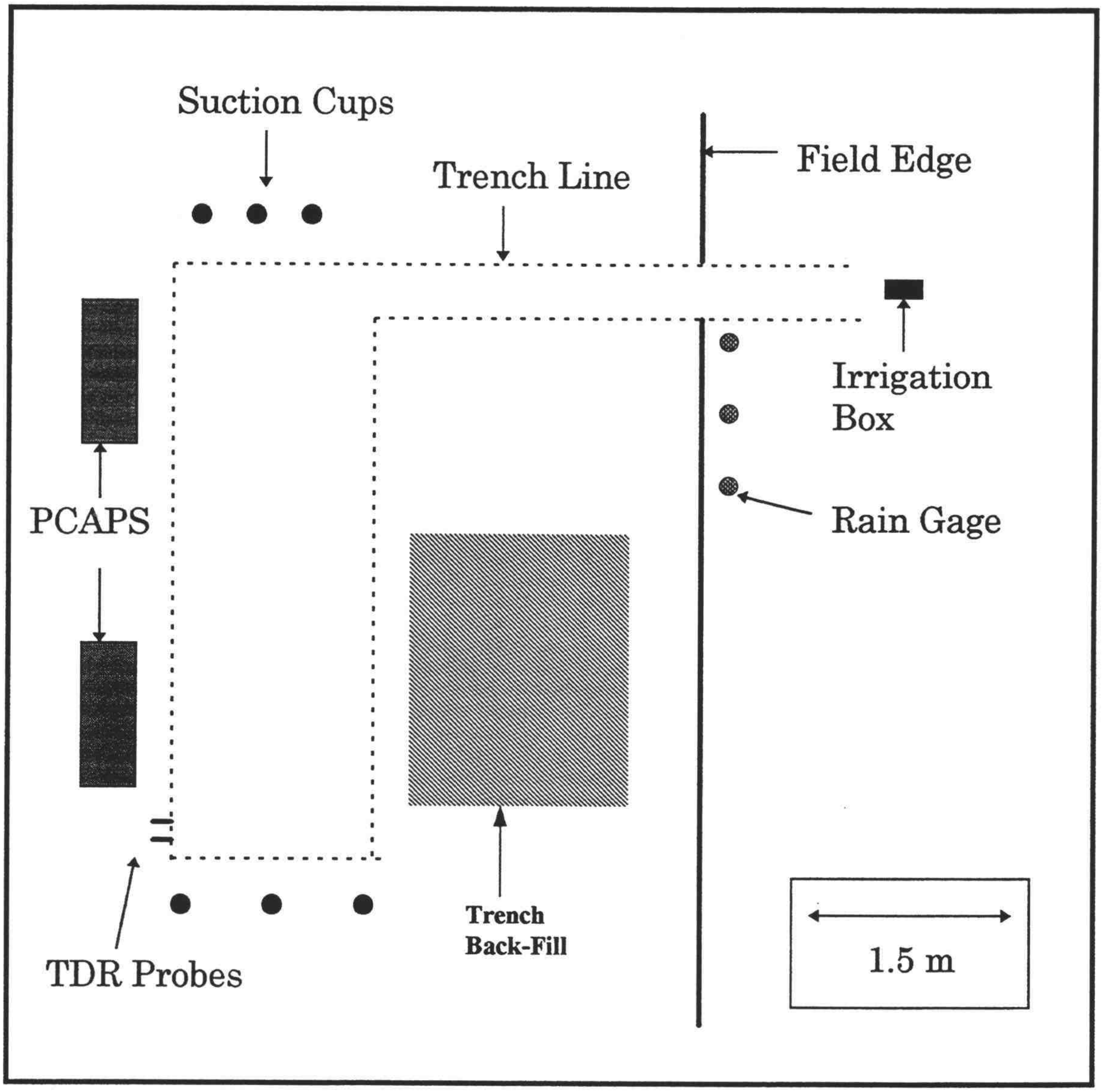

Figure 4. Typical special study site field lay-out for row crop, organic and mint sites (for blueberry and orchard sites: PCAPS were placed within the rows directly under the tree rows).

Table 6. Installation parameters for each site.

Site	Soil Type	PCAPS Depth (m)	# of Suction Cups installed	Depth of Suction Cups (m)	Water Table Depth (m)
Grass Seed #1	Silty clay loam	0.92	6	0.92, 3.0 and 3.5	3.5
Grass Seed #2	Silty clay loam	0.92	6	0.92, 2.7 and 3.0	3.0
Organic #1	Silty clay loam	0.92	6	0.92, 2.7 and 3.0	3.0
Organic #2	Loam	0.80	4	0.8 and 2.2	≈ 2.5
Blueberry #1	Loam	0.80	4	0.8 and 2.75	≈ 3.05
Blueberry #2	Fine sandy loam	0.80	4	0.8 and 2.45	≈ 2.75
Orchard #1	Fine sandy loam	0.92	6	0.92, 2.3 and 2.4	2.4
Orchard #2	Gravelly sand	0.65	6	0.65, 1.2 and 1.5	1.5
Mint #1	Loam	0.80	6	0.92, 2.3 and 2.75	2.75
Mint #2	Silty clay loam	0.92	4	0.92 and 2.75	
Mint #3	Fine sandy loam	0.92	6	0.92	
Mint #4	Silty clay loam	0.92	4	0.92 and 2.4	
Row Crop #1	Fine sandy loam	0.92	6	0.92, 2.2 and 2.4	2.45
Row Crop #2	Loam	0.90	6	0.92, 2.2 and 2.4	2.45
Row Crop #3	Silty clay loam	0.92	4	0.92 and 2.9	≈ 4.6
Row Crop #4	Silty clay loam	0.92	4	0.92 and 3.0	≈ 4.6

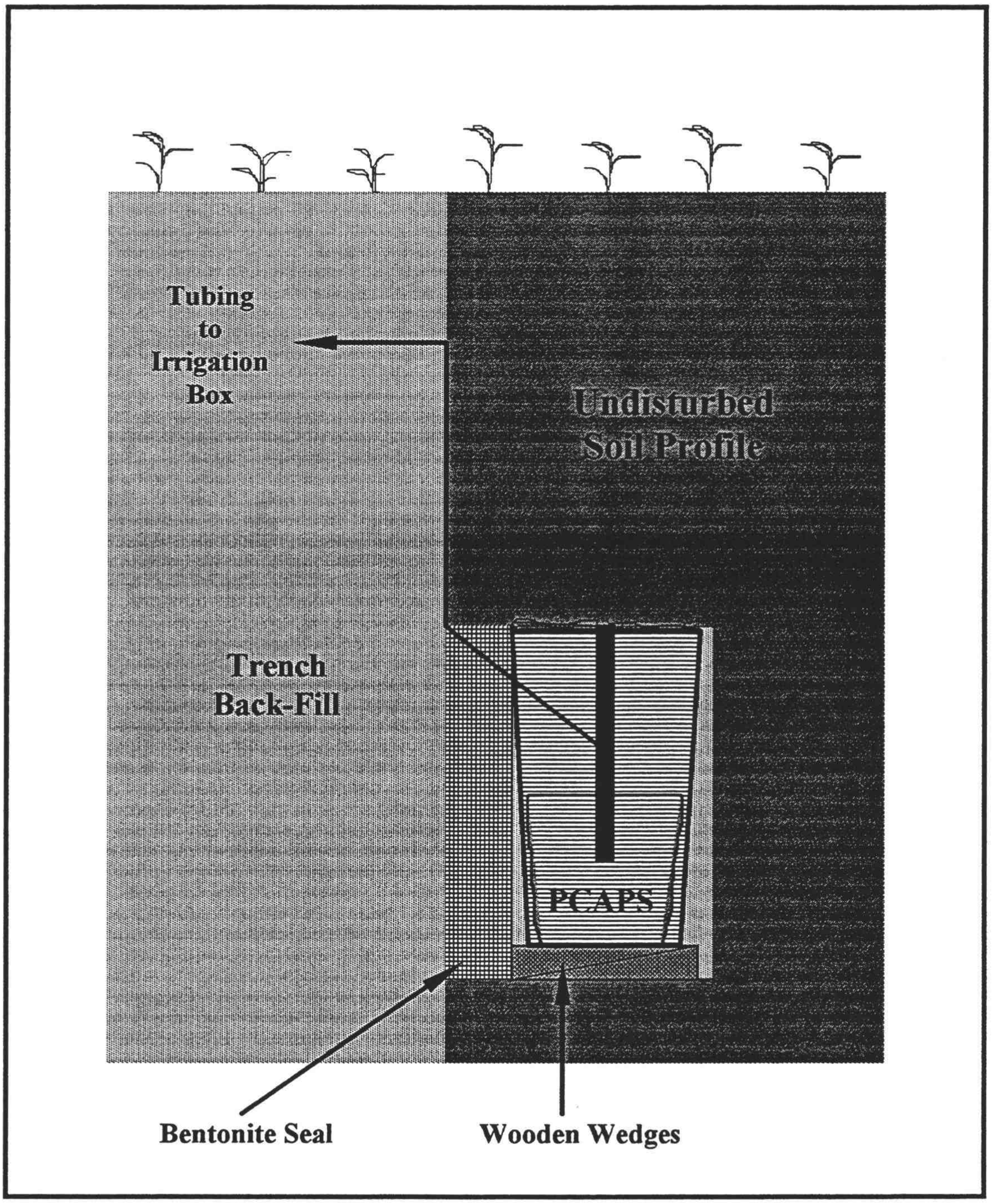

Figure 5. Schematic drawing of installation trench side-view showing PCAPS placement.

Suction Cup Samplers

Construction

High flow porous ceramic cups (5 cm O.D., 6 cm length, 1 bar air entry pressure) from Soil Moisture Equipment, Santa Barbara, CA (#653X01-B1M3) were used for sampler construction. A 2.54-cm I.D., PVC pipe was attached to the porous ceramic cup using epoxy. The top of the sampler was sealed with a rubber stopper. Two 3.175-mm I.D. HDPE tubings were used as the sample and vacuum tubing for the sampler. Sample tubing extended to the bottom of the cup to prevent dead volume. Each suction cup sampler was checked to insure that they held a vacuum before they were used in the field.

Installation

Two suction cups were installed at the level of the PCAPS, at the capillary fringe, and at the water table (Table 6). Suction cup samplers were installed using a bucket auger. Holes were drilled at a 45°-angle from the trench side wall to install suction cups at the PCAPS depth. Holes drilled for capillary fringe and water table suction cups were done so by drilling slightly off of vertical into the bottom side of the trench wall. Once the water table was located, the depth was recorded and an estimate of the height of the capillary fringe was made based upon soil texture and water content. Due to the area in which the farms are located, typically within 1 to 2 miles of a major river, problems were encountered with boulder size river gravel. An auger was rendered useless in these situations. For this reason, some sites were installed with only a total of 4 rather than the planned 6 suction cups (Table 6). The suction cups were installed at the greatest depth that could be reached using the hand auger. In all situations in which river gravel was encountered, the water table was not reached.

A silica flour slurry was poured into each sampler access hole. The suction cup sampler was dipped in the silica slurry and then placed in the auger hole. Native soil was refilled and compacted around the sampler, and then the sampler was hydraulically sealed by pouring a thick dry bentonite plug down the access hole. The hole was then refilled completely with native soil and sealed once again at the trench wall with a second bentonite plug. The tubing from each of the suction cups was also run to the field-side irrigation box. A vacuum of approximately 53kPa was applied to each sampler on the installation date to begin the sampling process.

TDR Probes

Construction

Time domain reflectometry (TDR) probes were constructed from 25 cm long, 6.35-mm stainless steel welding wire. One end of each probe was gold plated to allow electrical solder to bind to the probe. The other end of the probe was pointed to increase the ease of installation. To gold plate the probes, they were first cleaned using Midas (Midas Inc., Albuquerque, NM) Electrocleaner at 150°C combined with a stainless steel anode set at 6-12 volts for 1 minute. The probes were then rinsed in tap water and placed in a Midas 20% hydrochloric acid (HCl) dip at room temperature for 30-60 seconds. The probes were removed from the acid dip and placed in a Midas stainless steel activator at room temperature combined with a nickel anode set at 3-5 volts for 1-4 minutes to initially nickel plate the probes. The probes were rinsed in tap water, placed in the acid dip and re-rinsed in tap water. The probes were placed in a Midas cyanide-based gold plating solution at 150°C combined with a stainless steel electrode set at 2-4 volts for 10-30 seconds. The gold-plated probes were rinsed, placed in the acid dip and re-rinsed to complete the plating process.

The probes were connected to flood resistant coaxial cable (Belden 9203 M17/28-RGO 58) by soldering the positive lead wire to one probe and negative lead wire to the other probe. Any exposed wire was sealed using heat/shrink tubing. The probes were separated by at least 5 cm. The probes were equipped with 15.24 m of coaxial cable. The ends of each coaxial cable were fitted with a twist-on BNC connector (Newark Electronics part #50F2088).

An additional 0.65 m long, 12.3-mm TDR probe was constructed without gold-plating. The rod was not made to be installed for long periods of time but instead used at each site to determine an over-all moisture content for the upper soil profile layer.

Installation

TDR probes were installed only at the initial eight special study sites where rain gauges were installed. TDR probes were installed using a custom constructed instrument. The instrument had two rods separated by 5 cm and of slightly smaller diameter than the TDR probes to insure close soil contact for the probe. The rods were inserted at the desired location so as to make straight, evenly separated guidance holes for the TDR probes. Three TDR probes were placed at the level of the sampler, above the sampler. The coaxial cable was run from the trench side to the irrigation box. The coaxial connector was placed in a plastic ziploc bag that was sealed at one end with silicone sealant. TDR traces were analyzed using a Textronix 1502C TDR monitor and gravimetric water content was determined according to Topp et al (1988). The TDR probes were checked during periods when the profile was nearly saturated and nearly dried-out. This was believed to be the most important periods for mass balance analysis requirements.

Samples

Sample Collection

Samples were taken once a month beginning in October, 1993. During times of heavy precipitation, samples were taken twice a month. Samples were taken, on average, every 25 days from October, 1993 to January, 1995. A vacuum was supplied to the sample vessel access tubing for each PCAPS and the sample was drawn into a 4000-ml graduated glass cylinder. The total sampler volume was recorded and a representative subsample was taken in a 60-ml amber HDPE bottle. A vacuum was also applied to the access tubing for the suction cup samplers and the sample was drawn into a 1000-ml graduated cylinder. The volume was recorded and a subsample was taken. Samples were stored below 0°C and processed when all samples had been collected for the month. The sampling process typically took 3-5 days to complete for all sites, depending upon the amount of precipitation since the last sample date.

Sample Analysis

Samples were analyzed for anion concentrations using a Dionex 2000i ion chromatograph with a Dionex AS4A-SC separator column and an AG4A-SC guard column. This procedure gives results for all anions, but only nitrates were analyzed directly. Samples were also analyzed for pesticide concentrations using Ohmicron Immuno Assay test kits and a Milton Roy Spectronic 20 spectrophotometer. Subsamples were taken and kept in 20-ml HDPE vials at -12.7°C for future reference. When analyzing for nitrates, frozen storage produces minimal change in concentrations (Avanzino and Kennedy, 1993). The 60-ml amber HDPE sample bottles were washed and allowed to air dry for 24 hours before reuse in the field.

Statistical Methods

There is a nested structure to the design of the data. The two PCAPS were nested within the management site (blueberry, organic, etc...) which is nested within a certain farmers location. This is the set-up of the PCAPS experiment but, no analysis was carried out using this design. Instead, each site was treated as a block with the two PCAPS as treatments within the blocks. Therefore, only the main effect of the PCAPS were tested and not any effects of the site treatments. The main analyses were performed using paired t-tests and linear regression (ANOVA).

To evaluate the effect of either a mass balance model or conductivity model as being the defining model for sample collection, linear regression was used. Each sampler was assumed to contain a statistically independent measurement of ground water recharge and nitrate concentration. This assumption can be made because there is no spatial correlation between sites. Linear dependence between PCAPS recharge estimates and mass balance recharge estimates were evaluated based an the "goodness" of fit or coefficient of determination, R^2. Only the special study sites were used for the analysis because precipitation data was accurate for these sites. Linear dependence between PCAPS estimated flux and each sites saturated conductivity was estimated. The model which is most accurate for determining PCAPS sampling performance was the one most positively correlated to the PCAPS data. This model explains the linear relationship between the variables with the least amount of error on the two models.

Dependence between fertilizer nitrogen loss and PCAPS estimated nitrogen loss was done using linear regression. Analysis of variance (ANOVA) was performed on nitrate concentrations and nitrogen losses for sites under the same management systems. The same was done at each site to compare the differences in nitrate concentrations during the winter and summer seasons. Flow-weighted average NO_3^- concentrations over time are used to reduce the variance. In some instances, to assess the significance of the correlation, a t-statistic was used (Hirsch et al., 1993):

$$t = \frac{r\sqrt{n-2}}{\sqrt{1-r^2}} \tag{6}$$

with n-2 degrees of freedom and a probability of exceedance of $\alpha/2$, where r denotes the correlation coefficient, and n the number of data points.

In order to calculate the mass balance recharge at a given sampling event with a given allowable error and given confidence level, the following number of samplers is required assuming a normal distribution

$$n > \frac{t^2_{1-\alpha/2,n-1}s^2}{E^2} \tag{7}$$

where t is the t-statistic with $n-1$ degrees of freedom and a probability of exceedance of $\alpha/2$, s^2 the sample variance, and E the allowable absolute error in the mean. The sample variance was estimated from the pooled variance of the reacharge estimates for the N sampling events. The degrees of freedom were determined according to the number of samplers from which the sample variance was estimated.

Flow-weighted concentrations and standard errors were calculated as follows:

$$\bar{c} = \sum_{i=1}^{n} c_i w_i \tag{8}$$

$$Var(\bar{c}) = Var(c_i)\sum_{i=1}^{n} w_i^2 \tag{9}$$

$$SE = \sqrt{Var\left(\bar{c}\right)} \tag{10}$$

where

$$w_i = \frac{q_i}{\sum\limits_{i=1}^{n} q_i} \tag{11}$$

where $\bar{c}$ denotes the mean flux weighted concentration, at a particular, in [M L^{-3}], c_i and q_i the concentration and flux in [L T^{-1}] as measured by n samplers (always 2), and w_i the respective weight factor.

To evaluate the differences between PCAPS annual recharge and mass balance annual recharge, a paired t-test on the means was used. The same test was used to determine if there was any difference in PCAPS estimated nitrogen loss and mass balance estimated nitrogen loss. Flow-weighted concentrations were used in the nitrate analysis to reduce the the variance, but also the degrees of freedom are reduced. PCAPS at the same site were tested using a paired t-test in order to detect significant differences in recharge measurements at the same site. PCAPS were only paired when no statistically significant difference was detected between the two recharge measurements. To evaluate a difference in variation between PCAPS and suction cup nitrate concentrations, a F-test for equal variances was performed at each site. The test was performed to explain the amount of variability encountered when monitoring nitrate leaching using the two methods.

Chapter III. Passive Capillary Samplers (PCAPS) as Estimators of Recharge

Introduction

The monitoring of groundwater from wells is the dominant method for assessing aquifer contamination problems. However, monitoring of the vadose zone for groundwater recharge and quality before the problem reaches the groundwater may be a better method (Wilson, 1990). Excellent reviews indicate that rainfall, or the quantity of available drainage water, is the most important factor affecting the leaching of contaminants below the root zone (Pratt et al., 1972). Various methods are presently in use for monitoring water and solute transport in the vadose zone. These include (1) soil core profile sampling, (2) tile drainage, (3) vacuum extractors, and (4) lysimeters.

Analysis of the spatial variability of the leaching characteristics of a soil along with measurements at separate intervals in the profile make soil core sampling a valuable tool for measuring contaminants in the soil. The versatility and low cost of this method is what makes it so appealing. The ability to replicate measurements at different depths allows experiments to be designed properly in order to analyze the spatial variability of a soil characteristic. Soil core sampling is a once-in-time measurement. Interpretation of the flux of contaminants through the vadose zone is indirect at best, nor do soil cores allow for repetitive measurements at the same point. Flow-weighted averages of contaminant concentrations must be determined independently of the soil core sampling procedure. Interpretation of soil cores in assessing recharge quality implicitly assumes that the primary loss of solutes is by leaching, volatile chemical transport may be important depending upon the environment and target compound. Soil coring is a destructive method, in that soil must be taken back to the laboratory for analysis.

Tile drainage is a method of obtaining solute samples while also providing some measure of solute flux. Tile drains are an expensive and destructive form of soil solution sampler and are typically used in this capacity only where they have been previously installed for reasons of site management. Fields are significantly disturbed during their installation requiring the soil years to recover from the structural damage. Typically, they are placed on fields which require seasonal drainage due to their slow permeability. The amount of discharge from tile drains depends upon groundwater level and possible lateral water movement in more highly permeable soil layers. Because tile drains are typically used for drainage, experiments conducted using these samplers are done on sites which have had tile drains previously installed and the soil has substantially recovered. This results in study sites having slow permeability and unrepresentative discharges from tile drainage systems. For example, Hood (1977) found that only 20% of the rainfall was recovered from tile drains under field conditions compared with 38% for lysimeters. Thomas and Barfield (1974) showed in two measurements that 11 and 37% of total flow in a drainage ditch originated from the tile drains. Richard and Steenhuis (1988) used four tile drains on a field in northern New York to demonstrate preferential flow of solutes and the ability to define this flow using tile lines. The authors applied a chloride tracer to the research site and by estimating a mass balance concluded that tile drains could be used to sample solute transport through the vadose zone. However, the authors stress that while tile lines do integrate spatial variability in their sampling frame further investigations into their effectiveness need to be done.

The use of porous ceramic suction cup samplers was introduced by Briggs and McCall (1904) and later used by Joffe (1933) and Krone et al. (1951) among others. Kohnke et al. (1940) gave the first review of the sampler, including evaluations on construction and performance. A commercial form of the porous ceramic cup was manufactured by Soil Moisture Equipment Corporation, Santa Barbara, California, and tested by Wagner (1962).

The idea of applying a tension to the soil in order to extract soil solution samples has been widely used. However, many problems associated with the use of the suction cup sampler have been documented. The measurement of solute flux is impossible with this form of sampler. The sampler provides no information on the volume of soil sampled or the time at which the sampler is extracting the soil solution. Major units of recharge to groundwater such as fingered, preferential and channeled flow (Kung, 1990; Selker et al., 1992) may not be sampled with suction cups due to a non-continuous vacuum during the sampling period or the cross-sectional sampling area being too small to capture these components of recharge (Boll et al., 1991). Porous cup samplers may be completely by-passed during saturated conditions due to the channeling of water through interped pores (Shaffer et al., 1979). This may result in missing important contaminant pulses during fall and winter rainstorms or during times of agrochemical application (Barbee and Brown, 1986). For example, Cochran et al. (1970) concluded that the application of a constant continual suction causes soil solution movement to differ from natural conditions, regardless of the uniformity of the soil. Hansen and Harris (1975) found that different cup flow rates results in different sample collection times.

Soil solution samples collected by suction cups may be unrepresentative of actual leachate concentrations. By applying a suction to the surrounding area of the suction cup, soil solution is being extracted at a seepage rate which may be higher than the drainage rate under natural conditions (Tseng et al., 1995). England (1974) concluded that neither the volume of soil nor the size of pores from which samples are drawn can be determined. Experiments have shown that ion concentrations vary widely with distance from soil particles, and the concentration is a function of pore size. In a given volume of soil, the larger pores may have significantly different ion concentrations than the smaller pores. Thus, there are different amounts and concentrations of soil solution in soil pores, and these solutions are held at varying pressures within the soil profile (relative to the water table). In order to collect a soil solution sample representative of the concentration reaching the water table, a complex integration of soil solution drainage factors must be employed. However, the suction cup sampler is unable to mimic a soil's water/pressure relationships. Once a suction is placed upon a porous ceramic cup, it begins to sample from the largest pores where more dilute (or possibly higher concentrations) of ions are available for drainage. If the cup still maintains a vacuum after larger pores are drained, smaller pores may be drained providing soil solution having a lower (or higher) ion concentration. The result is an unrepresentative estimate of the actual soil ion concentration.

In most soils, water movement occurs at or near saturated conditions with soil/water pressures close to zero. Due to these pressures, a vacuum applied to a suction cup sampler greater then 10 kPa may result in sampling soil solutions that are not subject to leaching (Severson and Grigal, 1976). Barbee and Brown (1986) concluded that applying even small amounts of suction to extract a soil solution sample may cause significantly higher seepage rates, compared with rates under gravity drained conditions.

Spatial variability and time of sampling play a significant role in the performance of suction cup samplers. A porous ceramic cup with its small cross-sectional area may not adequately integrate chemical concentrations in soil solutions due to spatial variability. Biggar and Nielson (1976) suggest that soil solution samples are "point samples" and provide qualitative rather than quantitative measures unless the soil's spatial variability is fully defined. Therefore, it has been suggested that dense networks of tension samplers with uniform permeabilities and size in conjunction with uniform sampling intervals may reduce sample variability (Starr, 1985; Rhoades and Oster, 1986). Hansen and Harris (1975) suggested that collecting samples at relatively short intervals (e.g. a few hours or less) will reduce sample variability. Potentially, sample variability tends to be the greatest when samplers, regardless of sampling rate, are permitted to fill completely. Severson and Grigal (1976) found that as time to extract samples increases, the sample represents that which is held by a tension similar to the one applied to the porous cup. This further emphasizes the point of applying low tension to ceramic cups along with collecting samples over short time intervals. By doing this, the sampler has the ability to sample leachate volume. However, a dense, frequently sampled system will completely distort a soil's flow pattern (Tseng et al., 1995). In addition, the circumventing of samplers by channeling of soil solutions requires the use of an additional sampler to collect leachate volumes at zero-tensions.

A zero-tension lysimeter or pan sampler was designed and introduced by Jordan (1968). Zero-tension pan samplers depend on gravitational drainage to supply soil solution to the sampling reservoir. Pan samplers can be constructed to sample from very large surface areas. Theoretically they are only able to sample from the soil matrix that is saturated, i.e. which is a potential ≥ 0 cm H_2O. In many instances, macropores and low resistance channels will enhance the soil solution flux measured by the device resulting in dilution of concentrations (pesticide concentrations can be enhanced by macropore flow). However, Jemison and Fox (1992) found that since the soil pressure becomes greater than zero above the sampler during sample collection, there is a diversion of flow away from the sampler due to the lower pressure in the surrounding soil. Jemison and Fox (1992) define the collection efficiency as the ratio of observed to expected percolation. They found collection efficiencies for the zero-tension samplers to be low, ranging from 45% to 58%.

Haines et al. (1982) compared zero-tension and tension samplers and concluded that tension samplers will sample saturated flow less efficiently and unsaturated flow more efficiently than zero-tension soil solution samplers. Barbee and Brown (1986) reasoned that for more structured soils (such as those containing macropores), the pan samplers will provide a more representative sample through time and space than the suction cup. At higher moisture potentials, the pan samplers were able to provide more consistency in the samples obtained than the suction cup samplers. The major disadvantage of the zero-tension sampler is that it is consistently documented to be unable to collect soil solution in unsaturated conditions.

The idea of developing a sampler that is able to sample from a large surface area along with sampling saturated and unsaturated conditions was discussed by Hornby et al. (1986). The application of a tension to soil using a hanging water column made out of fiberglass wick helped develop the wick pan lysimeter. Brown et al. (1986) introduced the wick pan lysimeter, now called the Passive Capillary Sampler (PCAPS). Passive Capillary

Samplers have proven to give superior results to previously mentioned soil solution samplers in terms of efficiently collecting soil flux and chemical concentrations (Brown et al., 1986; Holder et al., 1991; Boll et al., 1992; Brandi-Dohrn, 1993). PCAPS use the capillary tension of moist fiberglass wicks to provide a negative pressure due to the hanging water column. The wick applies a suction between 0 and 50 kPa to the soil it is in contact with, and depending upon the wetness of the soil, the wick will sample at a flux similar to the soil flux. PCAPS have the advantage of being able to continuously collect samples of soil solution without a continuous vacuum source. PCAPS are also able to sample from a large area of soil thus allowing it to collect both macropore and matrix flow (Steenhuis et al., 1993).

In order to accurately sample unsaturated flow from a known area of soil, the native soil and flow regime must be left undisturbed. Soil cores and tile drains are unable to be used without disrupting the native soil regime. Large plots of soil must be excavated and then repacked with extreme care in order to install the samplers. It is not possible to return soil to its native state after excavation. PCAPS, as mentioned in the Methods chapter, are installed under native soil regimes. Trenches are dug and PCAPS are installed off the sides of the trenches. A trench effect is eliminated by sealing the installed PCAPS with bentonite from the refilled installation trench. PCAPS do little to effect the native flow regime. When flow occurs, the pore-water at the bottom of the wick is maintained at atmospheric pressure while the matric potential at the top of the wick is a function of the flux (Knutson and Selker, 1994). The pressure distribution of the wick is much the same as a soil's pressure distribution. When there is no flow, the matric potential of the wick is equal to the wick's length. This is the most negative pressure that the wick can generate. As the flow increases, the pressure at the top of the wick increases in the same fashion as the soil. Being a porous media, as is the soil, wicks have their own characteristic curves, and they also exhibit hysteresis. For these reasons, PCAPS can be designed with fiberglass wicks that are able to match a soil's hydraulic properties.

In designing PCAPS, the flux-pressure relationship of a wick and the soil must be matched. This is done so that the wick can mimic the unsaturated flow conditions existing in the soil allowing it to sample leachate efficiently. The only hydraulic parameters needed for the matching procedure are "a" and K_{sat} according to Gardner's (1958) unsaturated hydraulic conductivity function

$$K(h) = K_{sat}\exp[a_{soil}(h)] \qquad \text{for } h \leq 0 \tag{12}$$

where $K(h)$ is the unsaturated hydraulic conductivity of the soil, K_{sat} the saturated hydraulic conductivity, h the pressure potential in the soil (typically negative), h_{ae} the air entry pressure of the soil, and a_{soil} the exponential constant for the soil. Gardner (1958) used the exponential form of the conductivity-pressure relationship to solve Richards' equation for the steady-state evaporative flux from the water table

$$h = \left(\frac{1}{a}\right)\ln\left[\exp(az)\left(\frac{q}{K_{sat}}+1\right)-\frac{q}{K_{sat}}\right] \tag{13}$$

where h is the pressure potential at elevation z above the water table (negative), q the flux (positive upward - evaporative), K_{sat} the saturated conductivity, and "a" the exponential constant. This solution is well suited to wicks used in PCAPS since they are easily described by an exponential K-h relationship (Knutson and Selker, 1994). For a fiberglass wick, the sign of q, the flux, becomes negative since infiltration instead evaporation is being considered and z is the length of the wick. In order to match a wick to a soil, the flux is assumed constant and the water table is deep below the sampler. Under these assumptions the gradient in total potential is 1, thus $q = -K$, and the pressure in the soil may be calculated from (13) to be

$$h_{soil} = h_{ae} + \frac{1}{a_{soil}}\ln\left[\frac{-q}{K_{sat}}\right] \tag{14}$$

Once the proper conductivity-pressure relationships are determined, the number of wicks, the wick length and sampling area can be calculated by matching h_{soil} and h_w using (Knutson and Selker, 1994)

$$h_w = \frac{1}{a_w}\ln\left[\exp(a_w z_w)\left(q\frac{A_s}{A_w K_{sat}}+1\right)-\left(q\frac{A_s}{A_w K_{sat}}\right)\right] \tag{15}$$

where h_w is the pressure potential at the top of the wick (negative), A_s the sampling area, A_w the cross-sectional area of the wick, z_w the length of the wick (negative), and a_w the exponential constant for the wick.

Brown et al. (1986) tested nylon, glass and woven fiberglass ropes for capillary rise, saturated hydraulic conductivity, and adsorption of inorganic ions and organic compounds. Fiberglass wicks were able to simulate soil flux the best, and were found not to absorb any compounds or ions. Knutson and Selker (1994) tested many of the commercially available fiberglass wicks, and summarize their properties such as wick area, Gardner's exponential constant, saturated hydraulic conductivity, capillarity, and dispersivity. Some of the wicks and their properties tested by Knutson and Selker (1994) are shown in Table 7. Fiberglass wicks were shown to have much lower dispersivity values than observed in soils (Boll et al., 1992; Knutson and Selker, 1994). Knutson et al. (1994) point out that the commercially available fiberglass wicks are applied with fiber strengtheners, such as starch, and must be cleaned to avoid adsorption and poor capillary rise. Guidelines on wick cleaning procedures are presented by Knutson et al. (1994). PCAPS were first introduced as low resolution samplers (Brown et al., 1986). The low resolution design consists of a 30 by 30-cm pan with one wick placed in the middle. The filaments of the wick are spread over the entire surface of the pan. The high resolution sampler uses the same pan size with the pan split into 25 individual compartments (Boll et al., 1991). Each individual compartment contains one fiberglass wick and the pan is pressed up against the soil using springs.

At this time, little research has been done to evaluate the performance of PCAPS under varying field conditions. Holder et al. (1991) tested a low resolution, 0.09-m^2 PCAPS on three different soil types: a sand, a silt loam, and a clay. Saturated soil Br^- breakthrough curves were determined at each location and used to estimate the number of samplers required to characterize the flow of contaminants for each soil type [Note: Since the tracer tests were performed under saturated conditions, results of the experiments cannot be considered representative of natural flow conditions]. To achieve 95% confidence in sampling soil solution with representative chemical compositions, they estimate that 31 PCAPS were necessary for sandy soils, six for silt loams and only two for clay soils. Additionally, they find that the samplers were able to collect soil solution samples from soils having soil water potentials ranging from 0 to -6.0 kPA. However, sample volumes were only representative of soil flux at potentials of -5.0 kPa.

Steenhuis et al. (1991) tested two high resolution samplers in a silt loam and found them to be more effective than zero-tension pan samplers. Experiments were carried out under controlled conditions. The collection efficiency as measured with a water balance was 103% for the two PCAPS (C.V. 0.25 and 0.42) compared to 27% for the two zero-tension pan samplers (C.V. 0.84 and 0.91). The PCAPS were able to sample the early breakthrough of FD&C #1 blue dye which the authors attribute to the ability of the PCAPS to sample soil-water at low potentials or prior to saturation.

Brandi-Dohrn (1993) installed 32 high resolution PCAPS at the North Willamette Research and Extension Center in Canby, Oregon. For a 244 day test period, the authors found the collection efficiency as measured with a mass balance to be 80% (relative error of 9%) with the highest sampler efficiency being 86%. Using the previously mentioned matching procedure, the type of wicks used (2.93-cm. medium density Amatex) on a silt loam soil would suggest that the samplers would over-sample. The author attributed the under-sampling to poor air release from the collection bottles. They found the PCAPS to be more reliable than suction cup samplers for estimating the mean chemical composition of the soil solution. To achieve 95% confidence in estimating the mean bromide concentration, they estimate 37 PCAPS are required due to a high average coefficient of variation for bromide concentrations, which was 122%. The number of suction cups required was determined to be 47 (C.V. for Br concentrations = 126%). The findings suggest that PCAPS are a major improvement in soil solution sampling techniques in as far as reducing error in flux measurements and estimating mean nutrient concentrations. More field experiments under natural, rain-fed conditions and over longer periods are necessary to further evaluate the performance of PCAPS.

Table 7. Commercially available wick types with soil-matching variables (from Knutson and Selker, 1994).

Wicks	Diameter (cm)	Area (cm^2)	"a"	K_{sat} (cm/hr)
Pepperell 1/4"	0.64	0.322	0.075	622
Pepperell 1/2"	1.45	1.651	0.098	1168
Pepperell 3/8"	0.87	0.594	0.085	829
Mid-Mountain 1/2" Matrix braid	1.26	1.247	0.064	220
Mid-Mountain 1/2" Knit Braid	1.34	1.410	0.091	328
Mid-Mountain 3/8" Matrix Braid	1.02	0.817	0.062	323
Mid-Mountain 3/8" Knit Braid	0.94	0.694	0.129	528
Mid-Mountain 1/4" Matrix Braid	0.85	0.567	0.089	288
Amatex 3/8" Hi-Density	1.06	0.882	0.047	273
Amatex 3/8" Medium-Density	0.97	0.739	0.066	460
Amatex 3/8" Low-Density	1.12	0.985	0.083	607
Amatex 1/4" Medium-Density	0.65	0.332	0.077	291
Amatex 1/4" Low-Density	0.72	0.407	0.136	411
Amatex 1" Medium-Density	2.93	6.743	0.074	618
Amatex 1" High-Density	2.48	4.82	0.043	315
Mid-Mountain 1/4" Knit Braid	0.64	0.322	0.319	1380

The objectives of this study are to evaluate the performance of Passive Capillary Samplers under natural rain-fed conditions concerning (1) their operational characteristics; (2) their ability to estimate soil solution flux; and (3) to evaluate the factors controlling the sampler's collection ability and efficiency. As a result of evaluating the operational abilities of the PCAPS, the samplers ability to estimate recharge over a wide range of soil conditions and management systems is presented. A description of the sampler's performance as compared to suction cups is provided to support some earlier findings concerning the performance of suction cups in the field. The ability to sample from the vadose zone is usually attributed to the ability of the sampling procedure to mimic the conductivity/pressure relationships in the soil. However, PCAPS may show that the quantity of drainage water is the most important factor for determining leachate volume and thus concentration.

Results

Operational Characteristics

Of the 32 PCAPS installed in Lane County, five samplers were inoperable or did not operate efficiently. Two samplers at the Mint #2 site were deemed inoperable, and thus the site was omitted from the study. The site was omitted mainly due to the soil type and hydrogeology of the location. Upon installation, large boulder-sized rocks were encountered along with many abrupt textural changes in the soil profile. Installation of the PCAPS was carried out, but due to large winter rains and intense summer irrigation, the site was constantly under ponded conditions subsequently flooding the tubing access box. Two samplers at the Mint #1 site collected estimated percolation very inefficiently (Table 14). The inability of both samplers to collect estimated percolation could be attributed to either a textural change at the point of the sampler which would divert macropore or preferential flow (could possibly be an air gap that developed also) or the collapse of the HDPE sample box due to over-suction. However, NO_3^--N concentrations of the leachate collected from the samplers are very consistent with the other mint management systems (see Chapter 4). One sampler at the Row Crop #1 site sampled estimated percolation inefficiently for most of the study but appeared to begin sampling more efficiently towards the end of the study (Table 14). This could be explained due to errors occurring during the installation process. The site has a sandy soil type (Table 2) which resulted in some settling and disturbance of native soil regimes around the sampler. Over time, the soil may begin to recover and the sampler may perform much better.

A point of concern that has been noted by other researchers is the flooding of the sampler boxes. Flooding occurs when the outer fiberglass sampling boxes become filled with water. Although the fiberglass boxes are sealed to avoid such problems, water can leak in through the fiberglass material or defective joints. Some of the outer fiberglass boxes filled with water but none were found to be flooded at any time. Four samplers installed on the grass seed sites were flooded typically from Dec.-March due to the presence of either a high water table during this period or perched water table due to an impermeable clay layer. Similar problems were encountered at other sites; however, in these instances the PCAPS were typically flooded for only one month (Appendix D). Whenever the samplers were flooded, the PCAPS were emptied of only the amount of water the sample box could typically hold (48-60 L). The thought was that any extra water emptied from the sampler would merely be water drawn straight from the water table.

There were some technical failures observed with the suction cup samplers. As mentioned in the Methods chapter, some sites were unable to be installed with suction cup samplers at the water table level. Initially, 96 suction cup samplers were to be installed for the project (six per site). When the installation process was completed, 86 suction samplers had been installed with seven sites having only four suction cup samplers each. Of the 86 installed suction cup samplers, 12 were found to be inoperable. This evaluation was made based on checking the suction through the sampler (done by placing a vacuum on the sampling tube and checking to see if a suction is established in the open suction tube) or whether or not the sampler had collected any leachate during the study. In September of 1994, an attempt was made to replace all inoperable suction cup samplers as well as install samplers at the water table on sites where the water table was too deep to be reached with the hand auger (Row Crop #1, #3 and #4). In all, 14 new suction cup samplers were installed including six new water table samplers. Presently, there are 78 suction cup samplers installed and working in Lane County. The actual number of suction cups used in this study is 74 with four samplers at the Mint #2 site being operational but eliminated from the study.

The average monthly number of suction cup samplers which collected soil-water solution from January, 1994 to January, 1995 was 47 (C.V. = 23%). The maximum number of samplers which sampled in one month was 68 in January, 1995 which was the month of the highest recorded precipitation for the study period. The minimum number of samplers sampling in one month was 27 in March, 1994 which was the month where 50% of the study sites were inaccessible due to flooding. On the average, the volume collected by the suction cup samplers was 482 mL (C.V. = 70%) for the January, 1994 to January, 1995 period. There were occasions where some vacuum was found remaining in the samplers, typically during drier conditions. Assuming the average pressure potential in the soil was -15 cm relating to an average volumetric moisture content of 45% (see below), the suction in the samplers would tend to drop faster due to the amount of water being suctioned at a higher pressure. The larger average volume collection during the Nov. - May period (556 mL) compared to the June - Oct. period (366 mL) indicates that the suction cup samplers were able to sample larger volumes of water in shorter periods when the soil pressure potential was low. Due to the inability of the suction cups to estimate average annual recharge, Chapter IV compares the suction cup sampler and PCAPS based on NO_3^--N concentrations.

Soil -Water Retention and K_{sat}

Knowledge of the soil-water retention and hydraulic conductivity are essential for modeling processes in the vadose zone. Marion et al. (1994) indicate that because of their simplicity, laboratory techniques are useful methods; however, questions arise as to the validity of results produced by these techniques. For this reason, the soil water content as a function of pressure was predicted using a model. Two models were used (Appendix A), with the best fit for the data (Appendix A) obtained using the equation of van Genuchten (1980)

$$S_e = \frac{1}{\left[1+(-\alpha h)^n\right]^m} \quad \text{where} \quad \left(m = 1 - \frac{1}{n}\right) \qquad (16)$$

with

$$S_e = \frac{\theta - \theta_r}{\theta_s - \theta_r} \tag{17}$$

where S_e is the normalized moisture content, θ the volumetric moisture content, with subscripts r and s denoting residual and saturated, h the pressure potential in [L], and α [L^{-1}] (one over the air-entry value), n and m are empirical parameters effecting the shape of the curve. This model is also best suited because it predicts n and the air-entry value which are parameters which can be used for the wick matching procedure. The restriction m=1-1/n was used because it allows for the closest fit to the data for the first 200 cm H_2O of tension which is the critical pressures for soil-water flux. van Genuchten et al. (1991) developed the RETC code for quantifying the hydraulic functions of unsaturated soils. This code was used to generate the values of the parameters shown in Table 8. The parameters not fitted by the model were the saturated water content (water content at 3 cm H_2O) and the saturated hydraulic conductivity.

Table 8. Values for van Genuchten parameters obtained using RETC.

Soil	Depth	Saturated water content	Residual water content	Alpha	n	Air-entry value	R^2
	[cm]			[cm^{-1}]		[cm]	
Awbrig	70	0.47	0.11	0.0054	1.83	184	0.99
Chehalis	86	0.52	0.12	0.1692	1.12	6	0.97
Cloquato	65	0.48	0.08	0.0299	1.52	33	0.98
Coburg	62	0.50	0.06	0.0494	1.30	20	0.99
Malabon	67	0.48	0.12	0.0297	1.99	34	0.99
Newberg sandy loam	46	0.50	0.03	0.1340	1.25	7	0.99
Newberg loam	65	0.49	0.06	0.0109	1.29	91	0.99

Saturated hydraulic conductivity was measured in the lab using soil cores, however this method is prone to errors. Small cracks in the core, preferential flow along the sides of the core, over-packing of cores, and the application of high pressure heads during analysis may cause substantial experimental errors. For these reasons, the field saturated conductivity was also measured and used for the wick matching procedure. Table 9 lists the steady state flow rate, *Q*, and additional parameters needed to estimate the field saturated conductivity, K_{sat}, measured using a constant head well permeameter. Figures 6 and 7 illustrate the fit to the well-permeameter data. Guidelines for analysis of the infiltration data are provided in Chapter II. Volume of infiltration versus time figures for all remaining sites are provided in Appendix C.

Figure 6 illustrates the ability of the constant head well permeameter to determine changes in infiltration due to changes in the soil profile. For the first part of the curve, the field saturated conductivity was determined to be 2.4 cm hr^{-1}. After 3000 mL of infiltration, which translates to a depth of wetting front of about 52 cm, the infiltration rate changed to 4.5 cm hr^{-1}. The change in infiltration was due to the change in soil type at approximately the 150 cm depth. This information is consistent with observations made in neighboring pits where river gravel and sand belts were intermixed with the native soil. Typically, the average slope was used to predict the saturated conductivity as shown in Figure 6. Figure 7 is more consistent with infiltration measurements made at most sites (Appendix C). However, the presence of soil incontinuities is common for the southern Willamette Valley soils which formed on floodplains.

Table 9. Parameters and estimates for field saturated conductivity measured using a constant head well permeameter.

Site	Depth	Q	H	C	K_{sat}	R^2
	[cm]	[$cm^3\ sec^{-1}$]	[cm]		[$cm\ hr^{-1}$]	
Blueberry #1	7.5	0.30	11	0.99	0.76	0.99
Blueberry #2	8.0	0.21	10	0.93	0.59	0.98
Grass Seed #1	9.6	0.75	16	1.27	1.36	0.99
Grass Seed #2	9.4	0.025	11	0.99	0.065	0.98
Orchard #1	10.2	5.31	15	1.22	10.3	0.99
Orchard #2	8.1	3.34	10	0.93	9.29	0.98
Organic #1	9.0	0.62	10	0.93	1.73	0.99
Organic #2	8.5	0.18	15	1.22	3.46	0.97
Mint #1	7.6	0.32	16	1.27	0.58	0.99
Mint #3	7.8	0.66	13	1.11	1.45	0.99
Mint #4	8.2	0.61	12	1.05	1.45	0.99
Row Crop #1	7.9	3.89	14	1.17	8.04	0.99
Row Crop #2	7.9	0.13	14	1.17	0.27	0.99
Row Crop #3	8.6	0.82	16	1.27	1.49	0.99
Row Crop #4	8.3	0.11	13	1.11	0.25	0.98

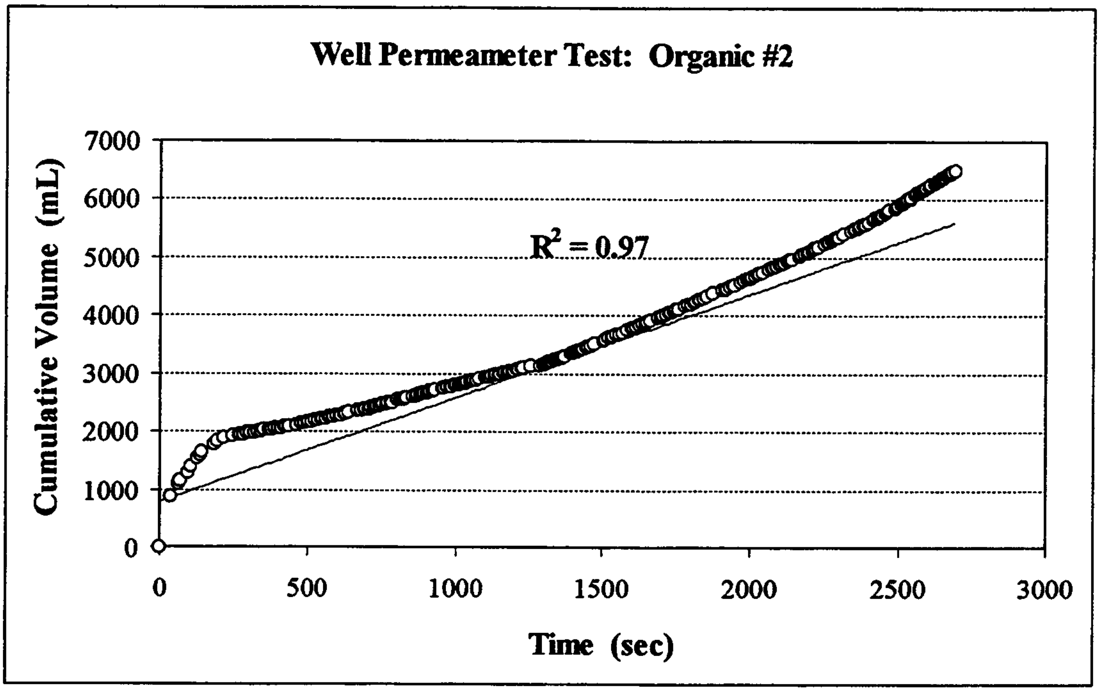

Figure 6. Steady-state flux with fitted regression measured using a constant head well permeameter (Organic #2).

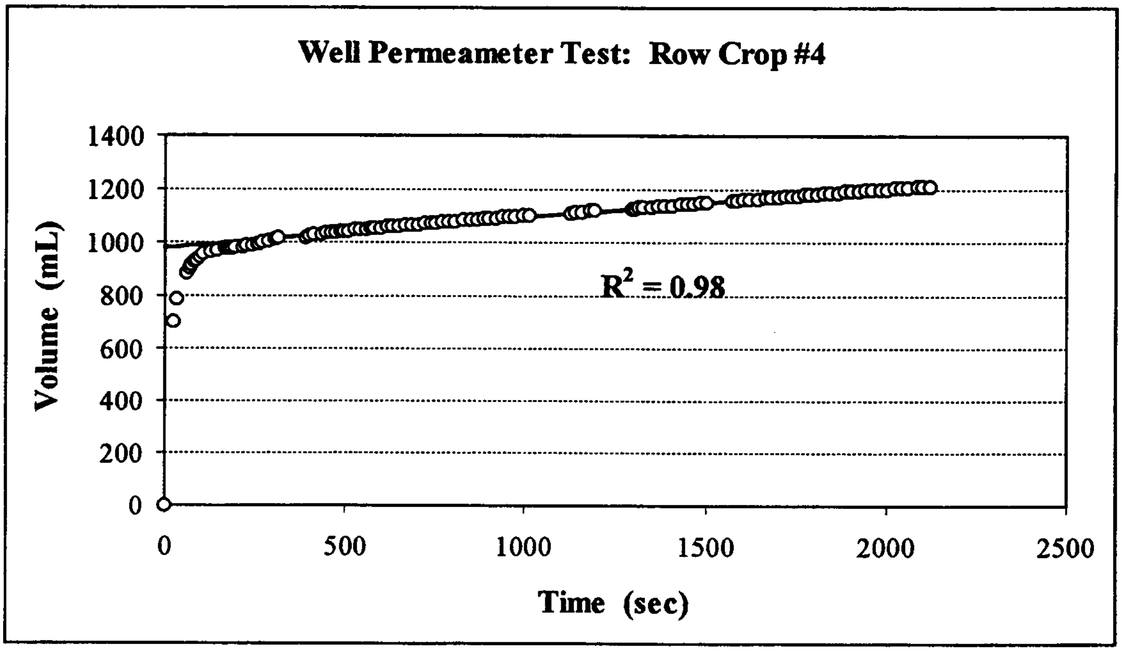

Figure 7. Steady-state flux with fitted regression measured using a constant head well permeameter (Row Crop #4).

Wick and Soil Matching

As outlined in the introduction, the conductivity-pressure relationship of a wick and the soil can and should be matched so that the wick can mimic the unsaturated flow conditions existing in the soil. Wicks which are not matched correctly with the native soil will result in disturbance of the flow regime leading to non-representative sampling (Rimmer et al., 1995). To predict the pressure (negative) at the top of the wick, Knutson and Selker's (1994) formula (Equation 15) was used. The matric potential as a function of flux for the soil was calculated using predictive models such as Gardner's exponential model (Equation 12). The wick matching procedure for this study was governed by certain practical constraints. As mentioned in Chapter II, the maximum wick fiber length was limited to 80 cm due to the dimensions of the sampling device. Thus, the maximum tension applied by the wick would be h_w = 80 cm H_2O which is a critical constraint. Of lesser effect, additional constraints included the sampling area, A_s, was limited to 900 cm^2, and the selection of wicks which was limited to those available commercially (Table 7).

Wick types were chosen based on their goodness of fit to the soil unsaturated conductivity in the area of the curve where the most flux occurs. This area was chosen as the pressures between -15 and -80 cm H_2O. For all soil types, either the Amatex 2.93-cm medium density or 2.48-cm high density wicks were chosen (Table 5). Mainly, these wick types were chosen because only one per sampling area was needed, and they provided the best fit to the soil flux. Figures 8 and 9 demonstrate the fit to the soil data for the study sites with the highest soil flux and lowest soil flux respectively. One question of interest was how adversely the constraint imposed by the limited range of commercially available wicks affected the ability to match soil pressures. Using a non-linear fit, optimal wick types were calculated by letting the wick saturated flux capacity, K_{sat} x A_w, and the exponential constant of the wick, a_w, be taken as variables rather than constraints. Table 10 lists the results for the fitting procedure.

Figure 9 suggests that the samplers would tend to over-sample, since the wick typically exerted a suction about -10 to -30 cm H_2O higher than the soil matric potential. This is supported by the assertions of Rimmer et al. (1995) based on their modeling exercises. The authors argue that for an optimal wick type where fiber length and sampling area are unconstrained, the air-entry value of the wick and soil should be matched ($\alpha_s = \alpha_w$). However, when L_w is kept constant, and optimal wick types are fitted for each soil, there is little variation in the calculated α_w (which is physically indicative of one over the air-entry pressure) (Table 10). The variable which is most sensitive to site characteristics is the wick saturated flux capacity which is due to the significant variability in the field saturated conductivitys' on our sites (Table 9). Therefore, one must realize that although it is true that $\alpha_s = \alpha_w$ will provide the best fit in the absence of practical considerations, actual constraints on PCAPS design may cause the saturated flux capacity of the soil and wick to be more important than the pressure saturation relationships.

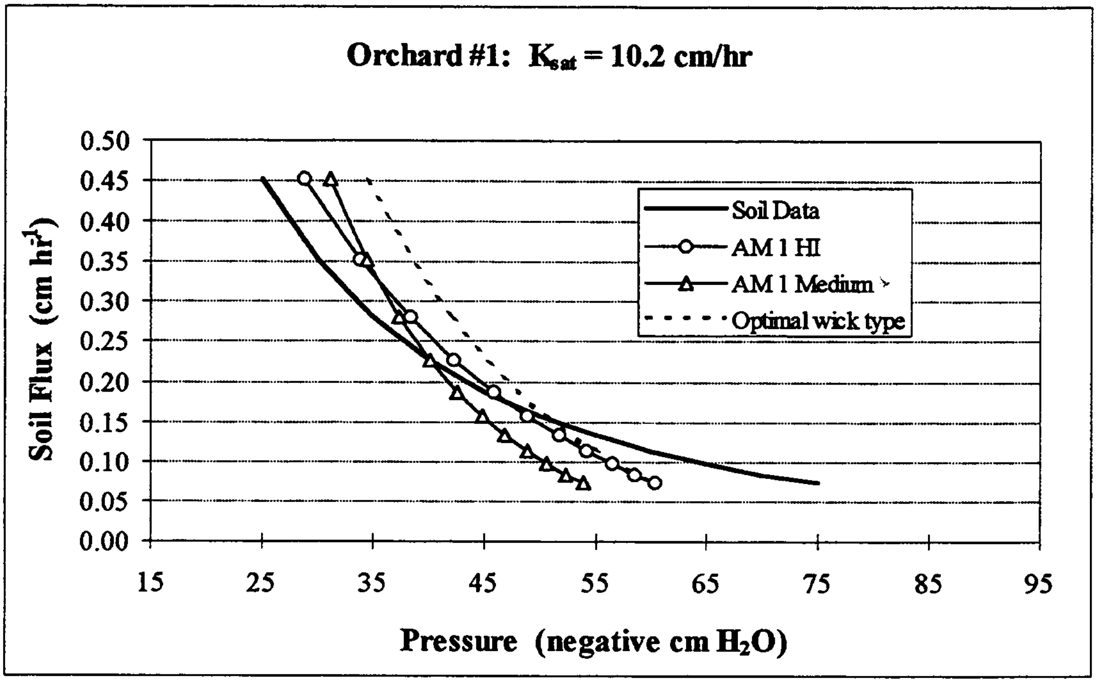

Figure 8. Soil flux and pressure for Orchard #1, Amatex medium and high density wicks, and the optimal wick type.

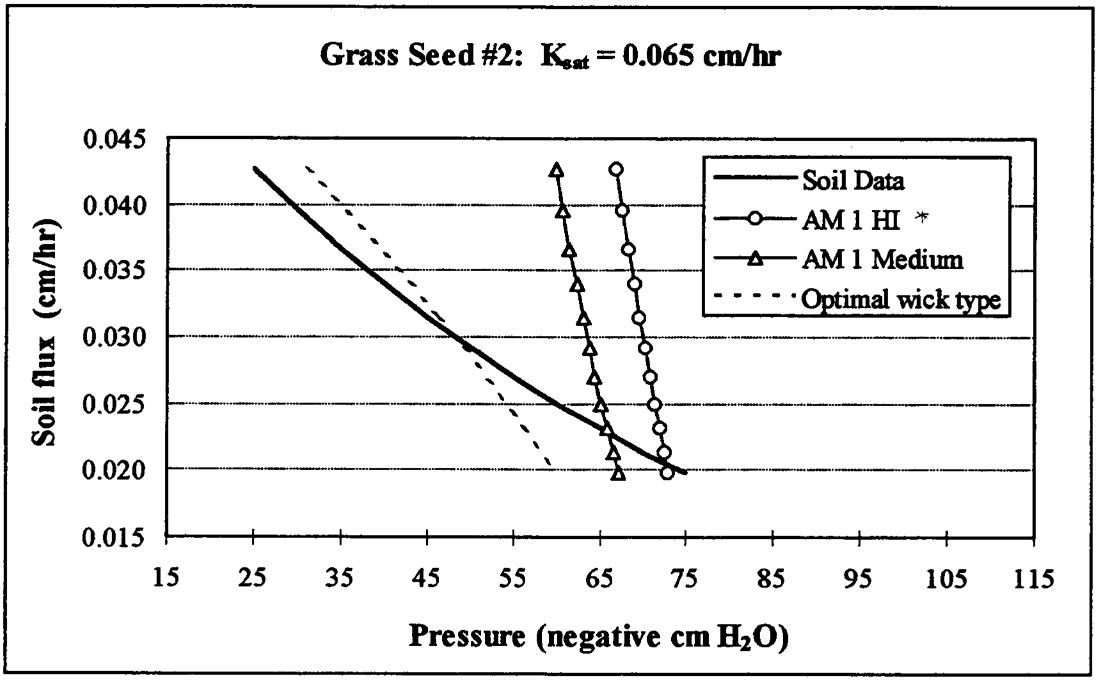

Figure 9. Soil flux and pressure for Grass Seed #2, Amatex medium and high density wicks, and the optimal wick type.

Table 10. R^2 for experiment wicks and optimal wick type parameters.

Site	R^2		Optimal Wick Type		
	Hi Density	Med. Density	a_w	$A_w * K_{sat}$ [$cm^3\ hr^{-1}$]	R^2
Blueberry #1	0.83	0.90	0.060	519	0.92
Blueberry #2	0.86	0.93	0.065	275	0.97
Grass Seed #1	0.72	0.83	0.084	291	0.93
Grass Seed #2	0.99	0.99	0	176	0.97
Orchard #1	0.91	0.93	0.057	3100	0.92
Orchard #2	0.91	0.93	0.060	2845	0.92
Organic #1	0.91	0.94	0.041	766	0.93
Organic #2	0.98	0.99	0.069	2845	0.99
Mint #1	0.91	0.94	0.041	766	0.93
Mint #2	0.68	0.79	0.071	327	0.86
Mint #3	0.83	0.90	0.060	458	0.92
Mint #4	0.96	0.99	0.069	1177	0.99
Row Crop #1	0.91	0.93	0.060	2462	0.92
Row Crop #2	0.82	0.90	0.041	121	0.93
Row Crop #3	0.96	0.99	0.070	1218	0.99
Row Crop #4	0.90	0.95	0.069	202	0.99

Mass Balance Estimated Recharge

The actual ground water recharge per sampling period was estimated using a hydrologic mass balance (ΣInputs = ΣOutputs). The form of the mass balance is

$$D = (P + I) - ET - \Delta S \tag{18}$$

where D is drainage (recharge) to ground water in [L], $(P + I)$ is precipitation and irrigation in [L], ET is Penman evapotranspiration in [L], and ΔS is the change in storage in [L]. Runoff is assumed to be zero. This is a good assumption on fields with little slope and relatively high saturated conductivity. However, during periods of intense rainfall, runoff may be a significant variable accounting for the loss of drainage water. Conditions did exist where runoff would be expected, and this would bias the estimate of D to be greater than the true D especially for high periods of rainfall. Since our experiments were conducted on actively cultivated commercial fields, quantification of runoff was not practical. Annual (Jan-94 to Jan-95) total ground water recharge (D) is listed in Tables 13 and 14 for all sites.

Precipitation and Irrigation

Rainfall and irrigation amounts were recorded monthly at each special study site which was installed with a non-recording rain gage. Each of the special study sites was initially instrumented with six rain gages. Three rain gages were 250-mm capacity rain gages with an evaporation minimizing circular funnel design. The other three rain gages were 100-mm capacity open top rectangular gages. The rectangular gages were not designed to prevent evaporation or resist freezing. In most cases during the first winter, the rectangular gages cracked and were rendered useless. Thus, only data from the three cylindrical gages were used to estimate the monthly precipitation. Table 13 lists yearly total precipitation values for each special study site. Appendix D lists monthly precipitation values for all sites.

For sites which were not instrumented with rain gages, rainfall data from the Eugene weather center or adjacent special study sites rainfall data was used to estimate the rainfall during non-irrigating months. For all sites, some rain gages had to be moved from the field during the summer because they were in the way of the farmers. In these cases data was taken from the farmers as to the amount of irrigation applied while one rain gage was placed on-field. Table 14 lists the annual total estimated rainfall data for the remaining seven study sites.

Evapotranspiration

Details concerning the calculation of evapotranspiration (ET) using the Penman-Monteith equation are provided in Chapter II. Table 11 lists the monthly parameters used to estimate ET using the PENMET computer program. ET is effected by the surface and aerodynamic resistance (Equation 4). These two resistance terms are different for each crop type; therefore, ET was estimated for each crop type used during the 1994 growing season (Table 4). Tables 13 and 14 list the total yearly ET estimates for each study site. Monthly estimates of ET are provided in Appendix D.

The accuracy of using the Penman-Monteith equation is addressed by Jensen et al. (1989). The authors evaluated the average peak monthly ET estimates using Penman-Monteith and express it as a percentage of actual lysimeter measured ET for both arid and humid regions. The percentages were 96% efficient for the arid locations and 98% efficient for the humid locations. The correlation coefficients for the two measurements were 1.00 and 0.98 for the arid and humid locations, respectively. Of the 20 methods tested by the authors, the Penman-Monteith estimates rated the best by having the highest efficiency (99% for arid and 104% for humid) and lowest weighted standard error of estimate (0.49 for arid and 0.32 for humid). The Penman-Monteith method is a widely used method for estimating ET and should contribute little error to the mass balance recharge estimate.

Table 11. Parameters for estimation of ET using the Penman-Monteith equation where T_{max} is the maximum temperature, T_{min} is the minimum temperature, RH is the relative humidity, and U_{avg} is the average wind speed.

Month	T_{max}	T_{min}	Net Solar Radiation	Max. RH	Min. RH	U_{avg}
	[°C]	[°C]	[W m^{-2}]	[%]	[%]	[m s^{-1}]
Jan-94	9.74	3.39	31.20	90	65	3.40
Feb-94	9.66	1.52	43.51	92	68	3.34
Mar-94	12.22	2.67	84.24	90	58	1.79
Apr-94	16.06	5.73	101.64	91	55	2.26
May-94	20.94	8.72	132.84	93	53	3.17
Jun-94	22.05	9.00	149.26	93	51	3.28
Jul-94	29.28	11.36	271.38	92	40	3.41
Aug-94	27.36	10.80	220.01	92	43	3.08
Sep-94	26.12	9.75	168.64	87	45	3.04
Oct-94	17.78	4.44	104.68	93	54	3.00
Nov-94	8.11	2.00	50.40	97	84	2.46
Dec-94	8.83	3.28	38.53	93	85	3.26
Jan-95	9.78	4.61	43.13	92	78	3.31

Change in Storage

The change in soil water storage can either be a loss or gain to the mass balance system. When the soil profile is dry and there is a rainfall event, usually no drainage will occur until the available storage is filled and the soil reaches field capacity. This results in a loss to the ground water recharge. When the soil is wet and becomes dry, there is positive drainage of storage water until field capacity is reached at which plants begin to transpire the remaining storage water. During the winter months, TDR measurements of saturated soil volumetric moisture content (θ_{sat}) were taken. The TDR measurement with the highest moisture content during the winter was considered the saturated moisture content. At the end of the growing season when the soil profile was the driest, the volumetric moisture contents were measured. Change in storage is estimated by multiplying the change in water content by the height of soil interval. During the experiment, soil water storage was refilled to saturation during the month of November, 1994. It is assumed that the change in storage estimate is valid for this month due to the heavy rainfall occurring after a long dry period (Appendix D).

Table 12. TDR measurements and soil profile storage estimates for special study sites.

Site	Depth	TDR Reading		θ_{wet}	θ_{dry}	ΔS
		Wet	Dry			
	[cm]	[m]	[m]	[cm^3 cm^{-3}]	[cm^3 cm^{-3}]	[cm]
Blueberry #1	0-61	1.17	0.88	54.9	41.0	8.47
Grass Seed #1	0-61	0.86	2.22	39.9	37.0	1.76
	61-76	1.03	0.88	48.7	41.0	1.18
Mint #3	0-61	1.05	2.76	49.7	47.5	1.29
	61-76	0.9	0.82	42.1	37.6	0.68
	76-91	0.89	0.8	41.5	36.5	0.77
Mint #4	0-61	1.3	3.3	59.7	56.1	2.17
	61-76	1.10	0.9	51.9	42.1	1.50
	76-91	1.08	0.9	51.0	42.1	1.37
Orchard #1	0-61	1.09	2.18	51.5	36.1	9.36
	61-76	1	0.76	47.3	34.2	2.00
	76-91	0.89	0.76	41.5	30.7	1.65
Organic #1	0-61	1.14	2.68	53.6	46.1	4.60
	61-76	1.1	0.96	51.9	45.2	1.02
Organic #2	0-61	1.1	2.22	51.9	37.0	9.11
	61-76	1.15	0.9	54.1	42.1	1.83
Row Crop #4	0-61	2.76	2.22	47.5	37.0	6.44

Soil water storage was never considered to contribute to ground water recharge. During the summer months when ET would exceed the amounts of (P + I), all extra ET which occurred was the transpiration of soil storage water. This assumption appears to be correct, and had to be employed since daily changes in water content could not be observed. In addition, the yearly change in storage is approximately zero due to the summer drying period and winter wetting period. Table 12 summarizes the TDR measurements and change in storage estimates for the Jan-94 to Jan-95 period.

Although some TDR measurements appear to estimate water contents higher than the actual soil porosity, the most confidence in the measurements comes from the difference between the wettest and driest water content measurements. While the TDR measured soil water content may be overestimated, measurement of the difference in water contents is quite robust (the contribution of soil dielectric subtracts out). This provides greater confidence in the storage estimate.

PCAPS Estimated Recharge

During the experiment, PCAPS were sampled on a monthly basis. Due to flooding problems and sampling complications, not all samplers were sampled during the month of April, 1994. For this reason, rainfall and ET data are combined for April and May for comparison to the PCAPS sampled volume. Monthly PCAPS estimated recharge was calculated by dividing the volume of sampled water by the total PCAPS sampling area (270 cm^2). Annual total recharge as estimated by both PCAPS at each site are listed in Tables 13 and 14. The efficiency of the PCAPS for estimating the mass balance recharge is also provided in Tables 13 and 14. Efficiency is calculated as the ratio of observed (PCAPS) to expected (Mass Balance) percolation. Appendix D lists monthly PCAPS estimated recharge and efficiencies for Jan-94 to Jan-95.

Table 13. Annual (Jan-94 to Jan-95) mass balance estimates and PCAPS estimated recharge and efficiency (PCAPS D / Mass Balance D) for special study sites.

Site	Total P	Total ET	Total ΔS	Total D	PCAPS Est. Recharge	PCAPS Efficiency
	[cm]	[cm]	[cm]	[cm]	[cm]	[%]
Blueberry #1	156.6	62.7	0.5	93.4	#1 - 65.4	70
					#2 - 59.6	64
Grass Seed #2	116.8	26.5	0	90.3	#1 - 86.5	96
					#2 - 77.0	85
Orchard #1	154.4	54.1	0	100.3	#1 - 104.2	104
					#2 - 100.2	100
Organic #1	153.4	59.1	3.5	90.8	#1 - 119.1	131
					#2 - 45.5	50
Organic #2	101.9	33.5	0.5	67.9	#1 - 48.3	71
					#2 - 57.1	84
Mint #3	151.5	43.1	2.0	106.4	#1 - 70.2	66
					#2 - 100.4	94
Mint #4	170.6	32.7	4.9	133	#1 - 154.2	116
					#2 - 182.9	138
Row Crop #4	147.4	54.1	0	93.3	#1 - 79.8	86
					#2 - 92.8	99

For the special study sites, the PCAPS monthly collection efficiency averaged 97%, with a median of 92%, for PCAPS #1 (C.V. = 50%) and 94%, with a median of 92%, for PCAPS #2 (C.V. = 69%). For all 15 study sites, the PCAPS monthly collection efficiency averaged 78%, with a median of 81%, for PCAPS #1 (C.V. = 66%) and 85%, with a median of 90%, for PCAPS #2 (C.V. = 78%). However, this estimate includes the

Table 14. Annual (Jan-94 to Jan-95) mass balance estimates and PCAPS estimated recharge and efficiency (PCAPS D / Mass Balance D) for non-rain gage study sites.

Site	Total P	Total ET	Total ΔS	Total D	PCAPS Est. Recharge	PCAPS Efficiency
	[cm]	[cm]	[cm]	[cm]	[cm]	[%]
Blueberry #2	177.7	54.2	0.8	122.7	#1 - 54.7	45
					#2 - 130	106
Grass Seed #1	118.8	24.5	0	94.3	#1 - 81.9	87
					#2 - 78.5	83
Orchard #2	166.1	47.7	1.8	116.6	#1 - 71.8	62
					#2 - 89.7	77
Mint #1	145.6	57.3	1.8	86.5	#1 - 4.6	5
					#2 - 10.1	12
Row Crop #1	123.4	35.2	1.5	86.7	#1 - 42.4	49
					#2 - 10.5	12
Row Crop #2**	106.7	57.5	2	47.2	#1 - 32.1	68
					#2 - 41.7	88
Row Crop #3	130.6	34.4	0.5	95.7	#1 - 89.6	94
					#2 - 96.2	101

** - Data not available for December, 1994 and January, 1995.

three PCAPS which were known to be sampling inefficiently for the experiment [PCAPS #1 and #2 at Mint #1 and PCAPS #2 at Row Crop #1 (Table 14)]. By eliminating these clearly non-functional samplers, the PCAPS monthly collection efficiency averaged 84%, with a median of 85%, for PCAPS #1 (C.V. = 59%) and 95%, with a median of 95%, for PCAPS #2 (C.V. = 68%). A discrepancy is made between PCAPS #1 and #2 because statistically significant differences in PCAPS collection volume between two samplers at the same sites were observed (Table 18). These averages for collection efficiencies are much better than those observed in similar lysimeter studies carried out without wick samplers [collection efficiencies typically ranging from 48% to 58%, e.g. Jemison and Fox (1992)]. The collection efficiency for the PCAPS is very similar to the efficiency reported by Brandi-Dohrn (1993) where the average efficiency was 80%. In a side-by-side comparison, Steenhuis et al. (1991) found collection efficiencies for PCAPS of 103% compared to 27% for zero-tension pan samplers. Thus, the observed PCAPS efficiency appears to be consistent with previous PCAPS research, with this study including a much broader range of soil types and operating outside of a controlled experiment setting.

Figure 10 depicts the relationship between mass balance estimated recharge and PCAPS monthly estimated recharge for the special study sites on a month-by-month basis. Figure 11 does the same for all study sites excluding the three PCAPS outliers. A 1:1 line is shown to illustrate whether or not a majority of samplers are over or under estimating the mean monthly recharge. A paired difference t-test to test the difference in the means was performed on both data sets. The significance level, $\alpha = 0.05$, is the probability that the null hypothesis, H_0, will be rejected given that it is true. The H_0 for the t-test is that the two mean monthly recharge estimates are equal. The outcome of the test provides a significance of probability, p-value, for the test. The p-value is defined as the smallest level of significance, α, at which an experimenter using the test statistic would reject the null hypothesis. A p-value < 0.01 would suggest that there is a difference between the two treatment means. Linear regression is also performed on both data sets to determine the actual slope that would better represent the data and also evaluate how the samplers are performing. Summary output for the paired t-tests and regression analyses are provided in Table 15. There is significant evidence (p-value = 0.0009) to suggest that the mass balance annual recharge and PCAPS annual recharge for all sites are not equal reflecting the under-sampling discussed above. For the special study sites, a p-value of 0.01 suggests that the means may be non-equal but the test is inconclusive. The regression analyses reveal a positive correlation between the PCAPS estimated recharge and mass balance recharge for all sites ($R^2 = 0.59$) and special study sites ($R^2 = 0.61$). The regression analysis suggests that there is a linear correlation between the two variables but the R^2's are inconclusive.

Figures 10 and 11 suggest that the PCAPS are under-sampling for the duration of the study, as noted in the mass balance analysis. A majority of the related points lie above the 1:1 line which indicates that the actual monthly recharge is typically greater than that estimated by the PCAPS. On the average for the special study sites, the mass balance recharge was 6.9 cm (s.d. = 42.5, n = 206) greater than PCAPS estimated recharge with values ranging from -133 cm to 140 cm. For all study sites excluding outliers, the mass balance recharge averaged 7.8 cm (s.d. = 45, n = 335) greater than the PCAPS estimated recharge with values ranging from -161 cm to 162 cm. This seems peculiar given that the wick matching procedure suggests that the PCAPS would over-sample due to higher pressures applied by the wick at equal soil fluxes. In their field experiment under natural conditions, Brandi-Dohrn et al. (1994) found their PCAPS to under-sample as well. From their wick matching results, they found their PCAPS would also over-sample due to the wicks applying a typical pressure three times that of the soil. However, both experiments used the same wick types as well as the same PCAPS design. Both experiments' PCAPS sampling ability was, therefore, hindered by the pressure distribution in the wick which is controlled by the wick fiber length, h_w. Although average conductivities for Brandi-Dohrn et al.'s (1994) experiments were much lower than K_{sat}'s for this experiment, the results indicate that similar PCAPS design will result in similar field performance results. It should be noted that the pressures of the wick filaments are not as high as those at the center of the wick. Pressure is not communicated well out to these wicks resulting in less sampling ability towards the outer edge of the sampler.

Table 15. Monthly mean recharge, with standard deviation in parentheses, paired t-test results, and regression results, with standard errors in parentheses, for all study sites (no outliers) and special study sites.

	Monthly Mean Recharge		Paired t-test			Regression		
	PCAPS	Mass Balance	n	t-stat.	p-value	Intercept	Slope	R^2
	[cm]	[cm]						
Special Study Sites	71.9 (65.5)	78.8 (63.4)	206	-2.33	0.01	24.3 (4.09)	0.76 (0.04)	0.61
All Sites (No Outliers)	72.3 (66.7)	80.1 (65.6)	335	-3.16	0.0009	25.5 (3.38)	0.75 (0.03)	0.59

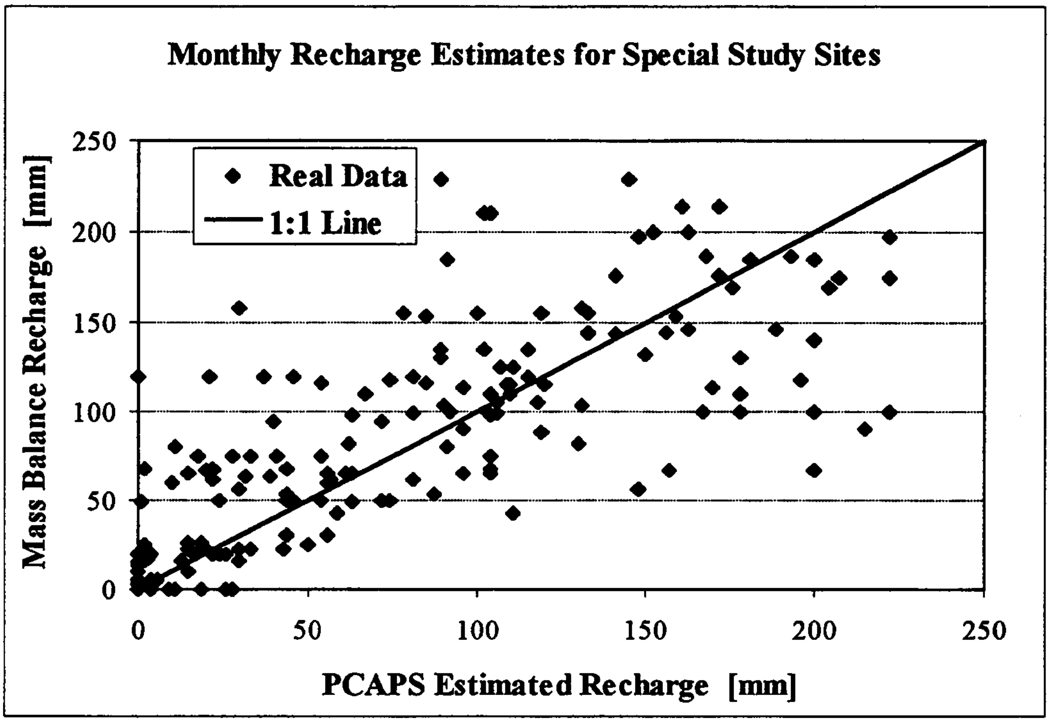

Figure 10. Mass balance recharge, PCAPS estimated recharge, and 1:1 line for special study sites.

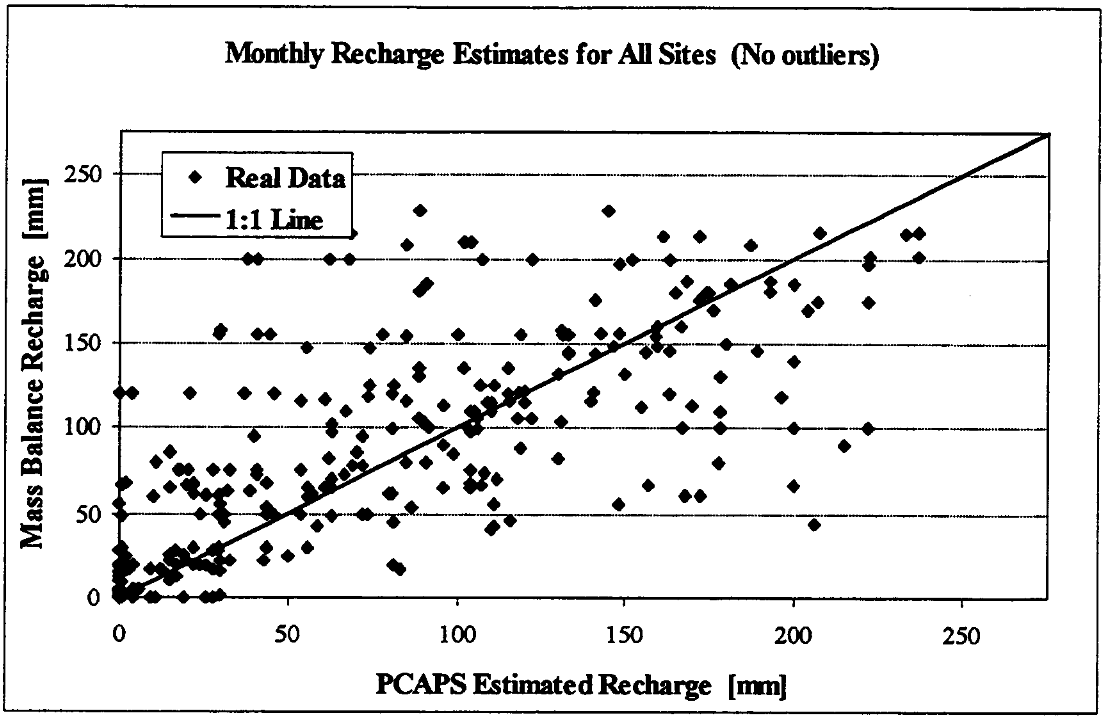

Figure 11. Mass balance recharge, PCAPS estimated recharge, and 1:1 line for all sites excluding the three PCAPS outliers.

Number of Samplers to Estimate Mean Monthly Recharge

From the PCAPS recharge results, the number of samplers needed to accurately estimate the mean monthly recharge is determined using Equation 7. Using recharge from all study sites excluding outliers (df = 26), at least 80 samplers are needed to estimate the mean monthly recharge with a 15% bound on the mean and 95% confidence level. The estimate does not change when calculated using only the special study sites. A more appropriate bound on the mean may be on the order of 30% given that the coefficient of variation for the mass balance estimated recharge is 80%. The number of samplers needed to estimate the mean monthly recharge at each site with a 30% bound on the mean and 95% confidence level is at least 20 while the number is 7 for a 50% bound on the mean and two for a 75% bound. This number appears to be a more likely estimate to be used based on the degrees of uncertainty and costs involved with using PCAPS.

The variance used in the number of samplers calculation is a pooled estimate of the variance in an individual PCAPS used in the experiment. Although a better estimate for the number of samplers could have been obtained using variances for each individual site, it was not considered feasible. When individual sites are used there is only 1 df and the variance estimates are very similar to using all sites for the analysis. Therefore, the number of samplers estimate incorporates the variation resulting from the PCAPS, each soil type, and each management system. This is an estimate of the number of samplers needed to estimate the average recharge over a large sampling area where soil types and management systems are different. When the scale is reduced to a single soil type or field size, the variance estimate might be expected to drop, reducing the number of samplers required. To illustrate this point, an estimate for the number of samplers needed on one soil type is done: There are four sites which each have a Newberg fine sandy loam or Malabon silty clay loam soil type. For both the Newberg fine sandy loam and Malabon silty clay loam, the variance estimate is of the same magnitude as for all sites, and therefore the number of samplers estimate is equivalent to the values given above since only 7 df is allowed with this analysis. This result suggests that the variability in the recharge estimates is a direct result of either the PCAPS performance or the management system and not necessarily the soil types. Proper statistical analyses such as a nested or split plot analysis would illustrate which variable contributes the most significant variance to the recharge estimates.

PCAPS Collection Ability as Influenced by K_{sat}

From the wick matching procedure, the thought that equal pressure distributions in the soil and wick will promote optimum sampling of leachate by the PCAPS was discussed. Due to the design of the PCAPS for this study, equal pressure distributions for commercially available wicks and eight soil types was impossible mainly due to the maximum pressure applied by the wick being limited to 80 cm of H_2O. For this reason, the optimal wick types fitted to each soil demonstrated that the saturated flux capacity of the wick, $A_w * K_{sat}$, was more important for these PCAPS in matching the unsaturated conductivity curves for each soil type. Figure 12 depicts the relationship between the optimally fitted wick saturated flux capacity and each sites K_{sat}. It should be noted that when K_{sat} was measured, the test was done typically 2-5 m from the location of the PCAPS. Due to natural soil variability, the K_{sat} at the location of the sampler may vary somewhat from the well-permeameter test estimate. From Figure 12 and the optimization results, it would appear that K_{sat} plays a significant role in the proper function of the PCAPS. Previous research has shown K_{sat} to be a controlling factor in the ability of the soil to transmit water both vertically and horizontally when a gradient exists. However, the collection ability of PCAPS may not be entirely dependent upon the rate at which water moves through the wick and soil.

Theoretically, if the PCAPS are sampling properly, they sample at a rate equal to the soil flux. If pressure distributions differ between wick and soil, their fluxes will differ as well. According to the matching equations (Knutson and Selker, 1994), PCAPS will over or under-sample depending upon whether or not the pressure potential of the wick is greater or less than, respectively, that of the soil. As previously mentioned, most of the PCAPS for this experiment were predicted to have pressure potentials of 15 to 30 cm of H_2O more negative than that in the soil at high flux, but during more typical low fluxes (0.005 - 0.2 cm/hr) the soil has a more negative pressure potential. Only 17% (five of 30) PCAPS had sampling efficiencies greater than 100% for the January, 1994 to January, 1995 period (Tables 13 and 14). This does not indicate that K_{sat} was less important at these sites than the pressure potential in the wick. Due to soil variability, the unsaturated conductivity varies dramatically throughout the profile. This is largely a result of the distribution of pores and water in the profile which cause varying pressure distributions. In other words, the pressure potential at one end of the PCAPS may be, or most likely will be different than the potential at the other end of the PCAPS. In theory, the PCAPS' wicks could be sampling at different rates across the entire sampler. If this were the case, the conductivity would not be as important as the amount of drainage water available to be sampled or pressure potentials in the PCAPS sampling area. For instance, the pressure potentials across the wick sampling surface could vary so much that some wicks will be at lower potentials and some at higher potentials. The wicks at higher pressure potentials would tend to sample all water, even diverting water from neighboring soil having lower potentials. Wicks at lower potentials will tend to sample less water at lower rates. Although, some wick filaments are over-sampling and some wick filaments are under-sampling, the PCAPS appear to have the ability to reach some equilibrium where it is able to sample a "large" area of drainage rather efficiently.

Biggar and Neilson (1976) state that soil solution samples are "point samples" and can provide only qualitative measures. Perhaps the PCAPS sampling area is large enough that even though soil spatial variability still exists, the samplers may have the ability to possibly factor-out soil variations by integrating a range of conductivities and sampling the available drainage volume. The only cases where over-sampling may occur is when saturated conditions exist, and diversion of flow around the sampler can occur during either saturated or low flow conditions. For these instances, the PCAPS filaments are all at the same potentials and sampling at a flux higher than the soil flux. More importantly, the pressure potential in the soil may become greater than zero which would result in the diversion of flow from the sampler. However, the thought is that when saturated conditions exist, the PCAPS will over-sample soil solution due to higher fluxes in the wick than in the soil. Beyond saturated conditions, conductivity may play a role in the movement of water, but the volume of drainage water appears to be more important for evaluating the collection ability of the PCAPS. It should be noted, however, that matching wick and soil conductivity distributions is very important when designing PCAPS. If conductivities are not matched to some degree, distortion of flow pathways is expected to occur. Wicks having higher conductivities will pull water from larger areas of soil than estimated, and PCAPS will not sample representatively.

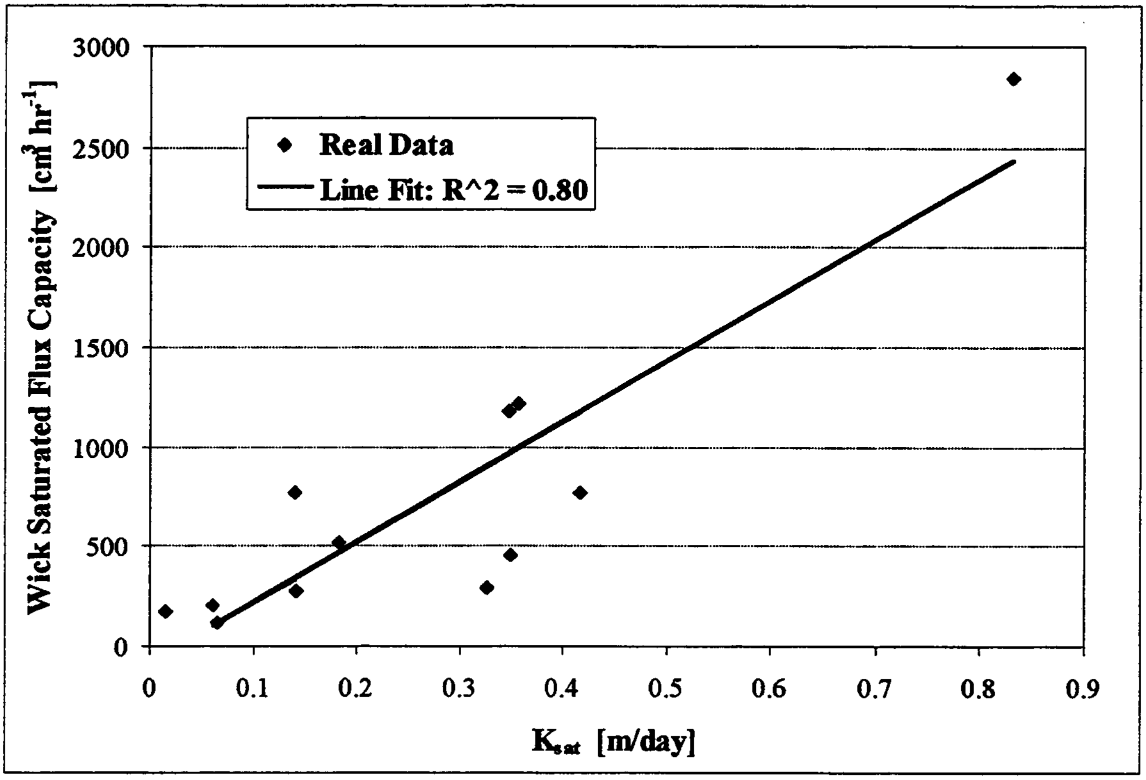

Figure 12. Wick saturated flux capacity (A_w * K_{sat}) and K_{sat} for all study sites not showing three values where K_{sat} > 2.0 m day^{-1} which were included in the analysis.

To illustrate the low significance of K_{sat} for explaining PCAPS collection ability, Figure 13 depicts the relationship between the cumulative PCAPS percolation and each sites K_{sat}. If it were true that K_{sat} were a controlling variable in the field for determining

PCAPS collection volume, sites having the highest K_{sat} would sample larger volumes of water, and those sites having low K_{sat} will sample smaller volumes of water. There is strong evidence to suggest that K_{sat} is not correlated to the amount of percolation collected by the PCAPS (R^2 = 0.01). If K_{sat} played a significant role in determining the PCAPS collection volume, there would definitely be a positive correlation between cumulative PCAPS percolation and K_{sat}. As we shall see, the flow through the system is dictated by the volume of water available rather than the soil properties. On the other hand, if the samplers were miss-designed we would expect greater systematic over or under-sampling.

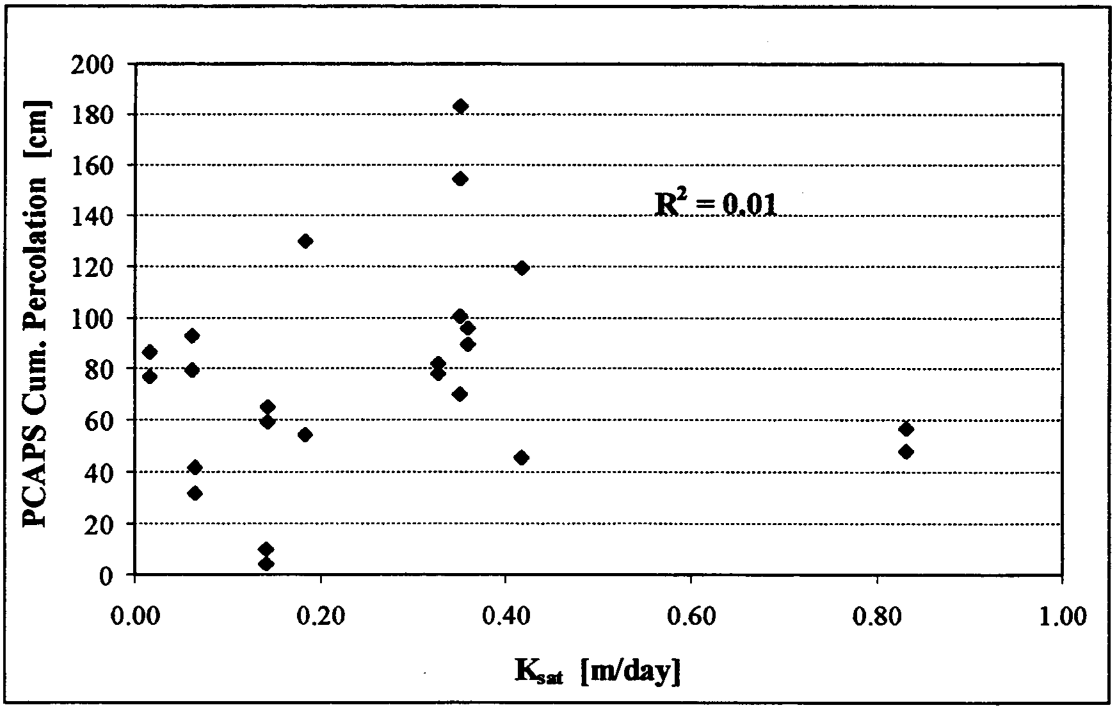

Figure 13. Relationship between cumulative PCAPS percolation and K_{sat} for all sites not showing three outliers with $K_{sat} > 2.0$ m day^{-1} which were used in the analysis.

PCAPS Collection Ability as Influenced by Drainage Volume

The quantity of drainage water is typically the most important factor which affects leaching of both precipitation and irrigation water and chemicals in the soil solution. Mass balance estimates provide the best "guess" as to the amount of drainage water which may leach below the root zone and eventually make it to the ground water. Tables 13 and 14 summarize the yearly recharge data as estimated with a mass balance. Appendix D contains monthly estimates of recharge for each study site in the experiment. As previously discussed, PCAPS appear to be estimating the ground water recharge with a high percentage of efficiency. The PCAPS are designed to collect the soil solution by sampling at a rate similar to the soil flux. This was shown to be very important when designing PCAPS. Without properly matching wicks to soils, PCAPS will either divert water from larger areas of soil or sample from smaller areas depending upon the soil/wick conductivity relationship. However, as shown above, the soil conductivity plays a minor role in determining the quantities of water that the PCAPS will sample each month. If soil conductivity were a dominant factor in estimating the quantities of water in the PCAPS, a mass balance model would tend to be insignificant for explaining recharge. Table 16 lists the recharge estimates if K_{sat} was the controlling factor for determining recharge. Obviously, these numbers could not be attained since this amount of water was not available. The mass balance model is a better estimate because it is founded upon the conservation of mass, a robust concept. Although runoff is not considered, it is clear from Tables 13 and 14 that most soils can take up the vast majority of precipitation. Using a conductivity controlled model to estimate recharge is inappropriate because complex hydrological processes are not being accounted for by the model. The conductivity controlled model refers to the ability of the soil to transport water at the saturated conductivity rate. The model does not differentiate between natural conditions existing in the field. Soil conductivity can only be a determinant of the rate at which water moves through the soil profile and not the quantity of water moving below the root zone.

To illustrate the significance of the mass balance drainage estimate, linear regression was used to develop a relationship between the mass balance recharge estimate and the PCAPS collected percolation. The regression analysis estimates the depth of leachate in the PCAPS (y) when the mass balance recharge estimate (x) is known. For this reason, only the special study sites will be used for the analysis. Each monthly measurement is considered independent of the next, therefore, the regression was done using monthly rather than cumulative estimates. Before the regression analysis was performed, two individual PCAPS at each special study site were compared using a paired t-test. The null hypothesis for the test is that there is no difference in the mean recharge estimates of the two PCAPS. Table 17 summarizes the t-test results for the special study sites. When no significant difference in the PCAPS means was identified (p-value > 0.05), the average recharge estimates for the PCAPS was used for the regression analysis, and confidence intervals (C.I.) on the mean PCAPS recharge estimate were calculated as well. Figures 14 and 15 illustrate the monthly PCAPS collected recharge for two sites where no difference and a significant difference, respectively, in PCAPS mean recharges were identified.

Table 16. Estimated monthly recharge based on a K_{sat} model.

Site	K_{sat}	K_{sat}-Limited Recharge Estimate	Actual Avg. Recharge Estimate	Estimated K_{sat}
	[cm hr^{-1}]	[cm]	[cm]	[cm hr^{-1}]
Blueberry #1	0.76	547	7.8	0.01
Grass Seed #2	0.065	47	7.5	0.01
Orchard #1	10.3	7416	8.4	0.01
Organic #1	1.73	1246	7.6	0.01
Organic #2	3.46	2491	5.7	0.01
Mint #3	1.45	1044	8.9	0.01
Mint #4	1.45	1044	11.1	0.02
Row Crop #4	0.25	180	7.8	0.01

Table 17. Results from paired t-test analyses on PCAPS collection volume at all study sites.

Site	PCAPS #1 Mean Depth [mm]	PCAPS #2 Mean Depth [mm]	df	s_p^2	t-Stat.	$t_{1-\alpha/2, df}$	p-value	C.I.
Blueberry #1	62.1	41.6	13	2940	2.43	1.77	0.03	-
Blueberry #2	53.4	110.8	13	4500	-3.43	1.77	0.005	-
Grass Seed #1	73.4	72.1	8	6627	0.11	0.46	0.92	50.5
Grass Seed #2	78.6	70	10	4687	1.03	1.81	0.33	37.4
Orchard #1	79.2	67	13	2840	0.98	1.77	0.34	25.2
Orchard #2	57.9	72.7	12	4492	-1.23	1.78	0.24	33.1
Organic #1	84.2	40.3	12	4180	3.47	1.78	0.005	-
Organic #2	38.8	45.5	12	3300	-1.36	1.78	0.2	28.4
Mint #1	4.57	8.65	12	43.3	-2.12	1.78	0.06	3.25
Mint #3	59.4	87.8	12	2350	-2.94	1.78	0.01	-
Mint #4	123.5	146.8	12	3856	-1.55	1.78	0.15	30.7
Row Crop #1	50.3	9.58	10	1073	2.72	1.81	0.02	-
Row Crop #2	34.7	44.7	9	2815	-1.78	1.83	0.11	30.8
Row Crop #3	89.2	100.3	10	6500	-1.81	1.81	0.1	43.9
Row Crop #4	61.3	71.4	12	5317	-1.4	1.78	0.19	36

The results from the paired t-test analysis indicate that three of the eight special study sites had suggestive p-values (p-value < 0.03 for Blueberry #1, Organic #1, and Mint #3), therefore, these sites PCAPS measurements were not averaged. For the regression analysis, data from some sites had to be transformed to the log scale to obtain constant residuals, as revealed. By examining the residual plots (Figure 16), the variation in the data shown forms a cone shape (solid lines) which indicates that the variability in the mass balance and PCAPS residual data is large for high values of recharge and low for low values of recharge. To reduce this variability and obtain a better fit for the regression model, the data is transformed to the log-scale. Figure 17 depicts the residual plot for the same site after transformation of the data. It should be noted that serial correlation may be significant for this analysis. Serial correlation suggests that the residuals of the regression estimates are correlated in the time they are measured (i.e. high residuals follow high residuals and low residuals follow low residuals in time). The effect of the serial correlation would be reflected in the altering of the standard errors and p-values of the regression estimates. Because minimal serial correlation was seen, the regression estimates are not meant to accurately describe the relationship between the independent or dependent variables. Rather, the regression analysis is a means of observing correlation and dependence between the mass balance and PCAPS data.

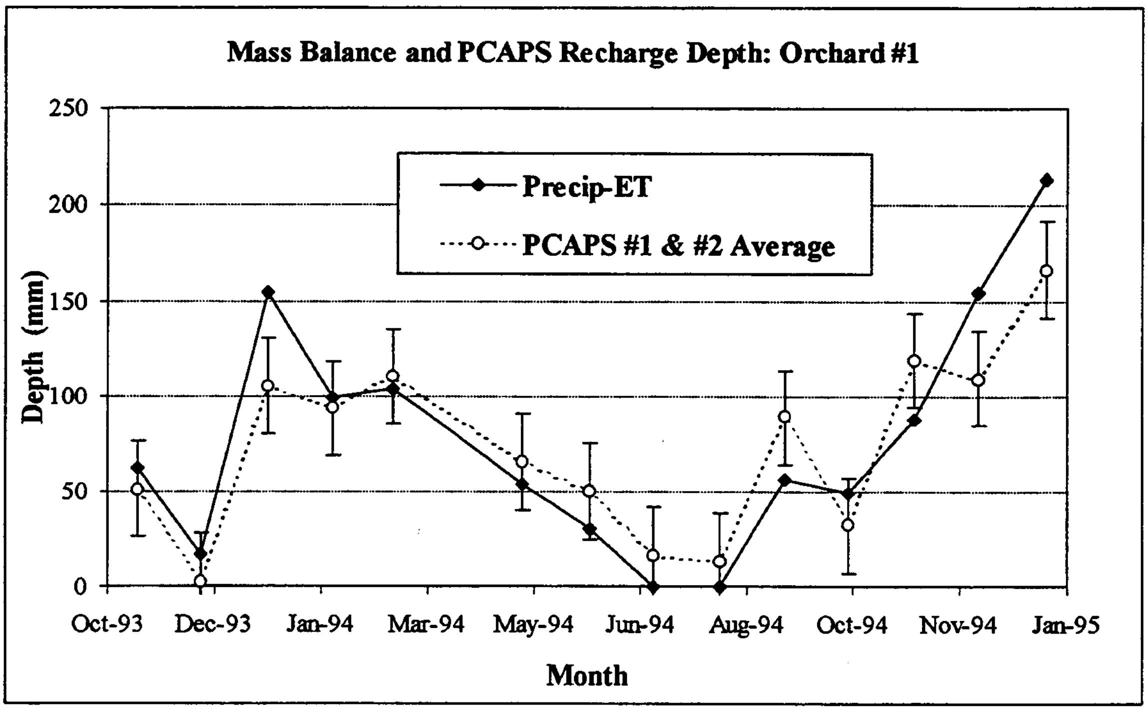

Figure 14. Mass balance (Precip-ET) and PCAPS estimated recharge depth for the entire study period (Nov-93 to Jan-95): Orchard #1.

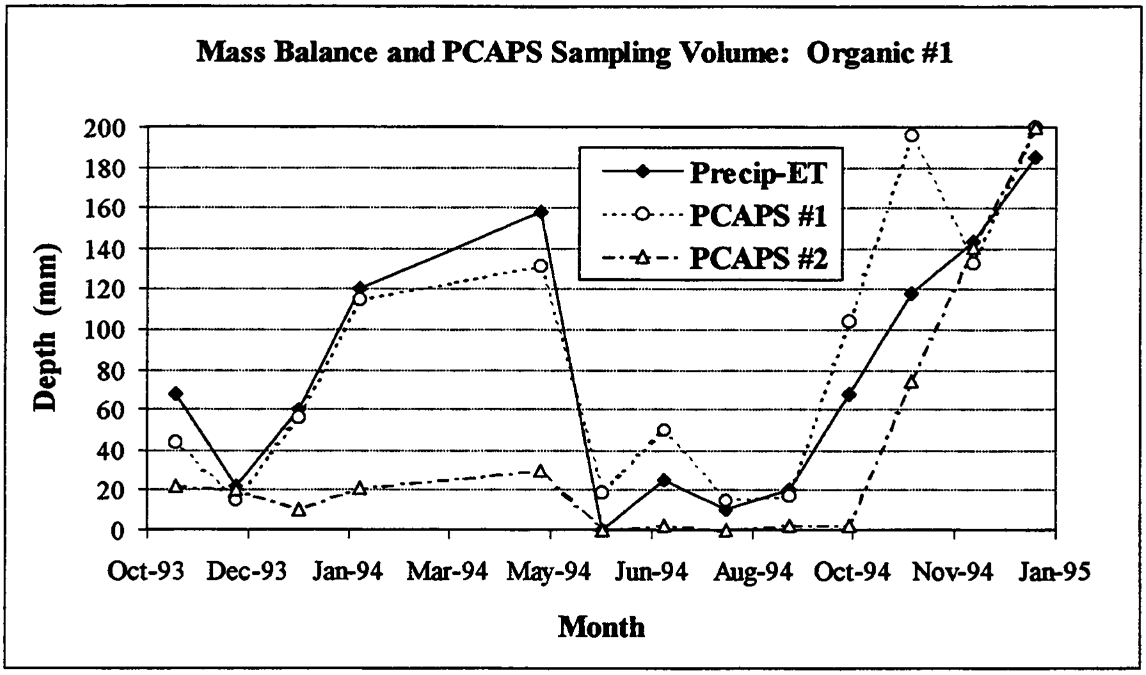

Figure 15. Mass balance (Precip-ET) and PCAPS estimated recharge depth for the entire study period (Nov-93 to Jan-95): Organic #1. A significant difference was observed between the two PCAPS on this site.

Table 18 lists the results for the regression analyses of PCAPS sample volume against estimated recharge. Coefficients of determination for the analysis averaged 0.75 with a range of 0.55 to 0.98. The coefficients of determination indicate that a linear relationship exists between the variables, and the regression equations are able to estimate the mean PCAPS recharge with reasonable accuracy. For all special study sites, the amount of recharge estimated by the PCAPS is significantly more correlated to the amount of drainage volume than to each sites' saturated hydraulic conductivity. From the regression analyses, each sites fitted estimates were obtained and averaged. The same was done for the mass balance recharge estimates at each site. An "average" regression equation was fit to these averages in order to have a single equation estimating PCAPS recharge from mass balance recharge for the experiment. The results of the average analysis are provided in Table 18. Figure 18 depicts the average mass balance and PCAPS recharge data for special study sites with the fitted regression estimate of PCAPS recharge.

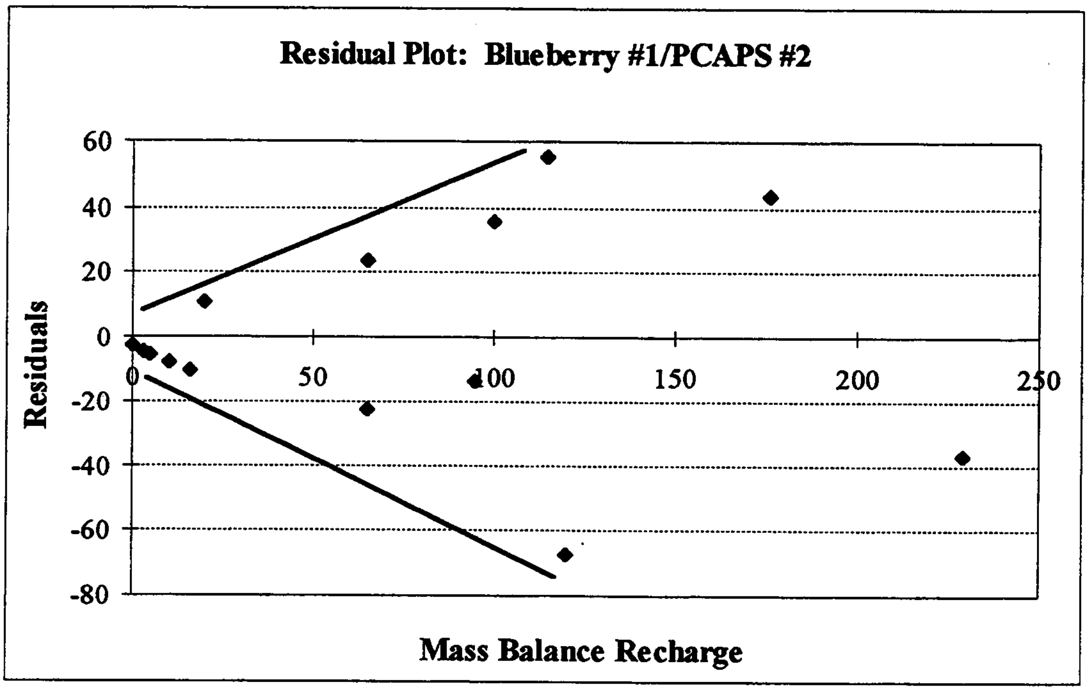

Figure 16. Residual plot for untransformed data at Blueberry #1 for PCAPS #2.

The "average" regression equation is highly correlated to the average PCAPS data ($R^2 = 0.97$). The slope of the regression equation ($\beta_1 = 0.68$) is similar to the previously calculated regression slope for the annual recharge data ($\beta_1 = 0.75$). The slight difference is due to the heavier weighting of low flow data due to the use of log transformed data. The result of the "average" regression analysis also demonstrates that on a monthly basis, the PCAPS are under-sampling the mass balance recharge. From Figure 18, it is apparent that during periods of high recharge (Precip-ET > 100 mm), the PCAPS consistently under-sample.

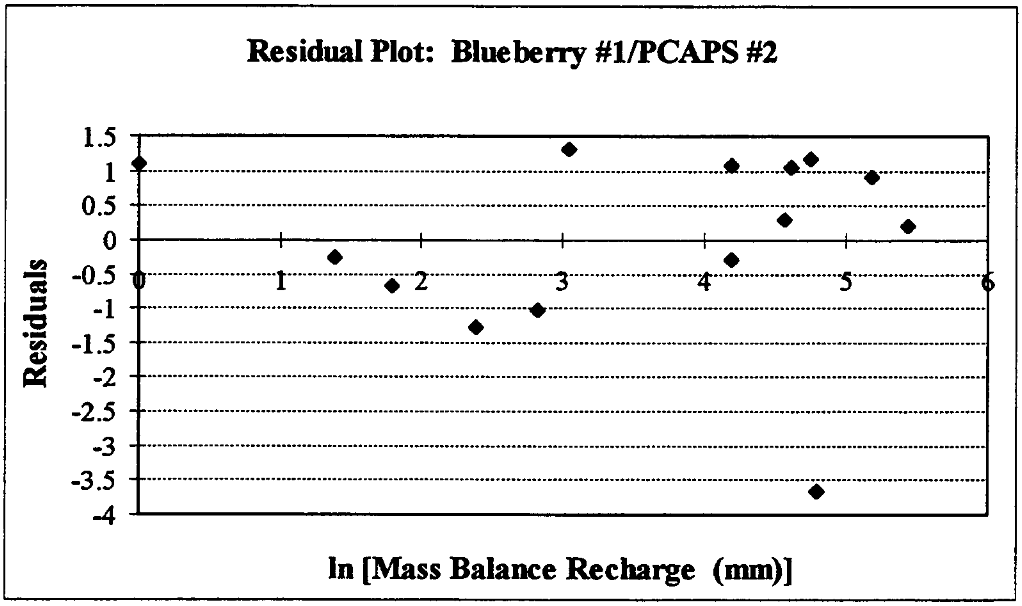

Figure 17. Residual plot for log-transformed data at Blueberry #1 for PCAPS #2.

This fact may well be a result of local runoff due to surface ponding which was frequently observed (this was also true in the study by Brandi-Dohrn, 1993). Thus we must bear in mind that the observed discrepancies could arise either due to PCAPS sampling error, or due to an error in the estimate of recharge. Runoff is not accounted for in the mass balance model, and during periods of saturation and heavy rainfall (i.e. winter), runoff may be a significant variable. This appears to be the case especially since the under-sampling occurred during the winter months.

Table 18. Regression output for PCAPS sample volume against estimated recharge for the special study sites (** - indicates transformed data employed in the analysis).

Site	Intercept (S.E.) [cm]	Slope (S.E.) [cm cm^{-1}]	n	R^2	F-Stat.	p-value
Blueberry #1	PCAPS 1: -1.20 (0.50)**	1.21 (0.13)	14	0.88	87.8	< 0.001
	PCAPS 2: -1.11 (0.93)**	0.99 (0.24)	14	0.59	17	0.001
Grass Seed #1	11.3 (24.7)	0.78 (0.24)	10	0.57	10.8	0.01
Orchard #1	19.7 (9.02)	0.69 (0.09)	14	0.83	56.8	< 0.001
Organic #1	PCAPS 1: 8.93 (13.2)	0.98 (0.14)	13	0.83	51.9	< 0.001
	PCAPS 2: -1.16 (0.89)**	0.98 (0.22)	13	0.64	19.4	0.001
Organic #2	-0.46 (0.45)**	0.96 (0.13)	13	0.84	56.4	< 0.001
Mint #3	PCAPS 1: 5.20 (13.6)	0.60 (0.13)	13	0.67	22.2	0.0006
	PCAPS 2: 12.0 (10.4)	0.84 (0.10)	13	0.87	74.4	< 0.001
Mint #4	1.74 (0.84)**	0.68 (0.19)	13	0.55	13.3	0.004
Row Crop #4	-4.00 (4.18)	0.94 (0.04)	13	0.98	550	< 0.001
Average	8.60 (3.38)	0.68 (0.04)	14	0.97	375	< 0.001

In Figure 18, it appears that the regression equation under-estimates the PCAPS recharge estimates for drainage amounts > 100 mm. This could be a result of the transformation of some of the data. When fitted results are back-transformed to the natural scale, they are no longer estimating the mean but rather the median PCAPS recharge. In addition, the ln transform tends to under-estimate larger values when back-transformed to the natural scale. Table 19 lists the goodness of fit of the regression equation to each special study sites individual mass balance and PCAPS recharge data. For all cases where the data was not transformed, the R^2 was greater than 0.78. The "average" regression equation does a poor job of accounting for error when the variability between the mass balance and PCAPS recharge is too big. This reinforces the observation that the log-transform does a poor job of estimating the mean recharge. The most important conclusion is that the PCAPS sampling ability is highly correlated to the mass balance recharge for all cases. This was expected due to the fact that the PCAPS are designed to intercept flow from a large surface area. It is apparent that if PCAPS are matched reasonably well with the soil hydraulic conditions of a site, the wicks will sample the available drainage with little error. However, at times of peak flow, a combination of surface runoff and diversion of flow around the sampler causes a persistent discrepancy between the PCAPS recovery and the ground water recharge estimated assuming only ET and percolation.

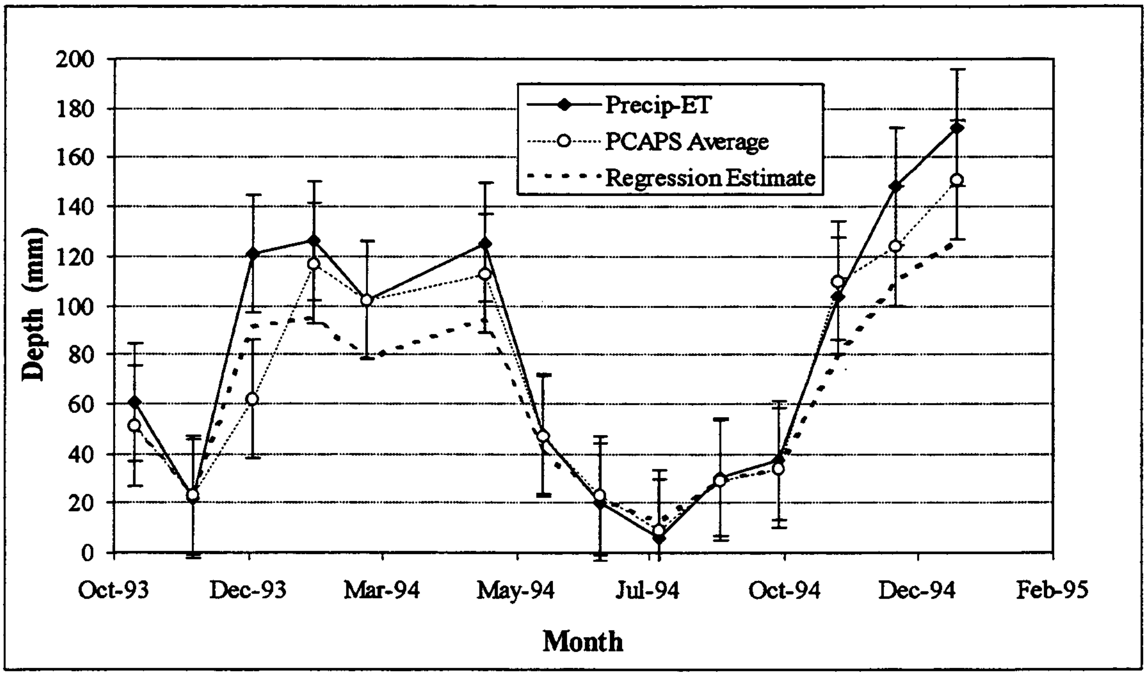

Figure 18. Mass balance and PCAPS monthly average recharge and PCAPS recharge regression estimate for special study sites.

In any case, for the period of a year, the PCAPS performance was superior to that reported in literature for other in-situ sampling methods. The fact that variability in soil and mass balance data exists, supports the fact that variability in PCAPS data will exist as well. Spatial variability cannot be avoided so the use of one sampler at a site is inappropriate. On the average, the PCAPS work very efficiently, but as with any statistically distributed population, variation in the data does occur. By increasing the number of samplers, variability is reduced, and, as in this experiment, favorable results can be obtained.

Table 19. Best fit (R^2) and sums of squared error (SSE) for the "average" regression equation and true PCAPS recharge at each special study site.

Site	R^2	SSE	n	F-statistic	p-value
Blueberry #1	PCAPS #1: 0.88	5300	14	87.4	< 0.001
	PCAPS #2: 0.58	13706		16.5	< 0.001
Grass Seed #1	0.57	21450	10	10.8	< 0.001
Mint #3	PCAPS #1: 0.67	7478	13	22.2	< 0.001
	PCAPS #2: 0.87	4354		74.5	< 0.001
Mint #4	0.23	28737	13	3.32	< 0.001
Orchard #1	0.83	5217	14	56.8	< 0.001
Organic #1	PCAPS #1: 0.83	9400	13	51.9	< 0.001
	PCAPS #2: 0.60	18720		16.4	< 0.001
Organic #2	0.78	8555	13	38.7	< 0.001
Row Crop #4	0.98	1212	13	550	< 0.001

Conclusion

The PCAPS showed little evidence of any technical failures (the only site omitted failed due to submersion of the site). Only three of the 30 PCAPS used in the analyses were determined to operate inefficiently and in each case the failure was evident. The PCAPS inefficiency was attributed to either collapse of the interior HDPE sampling box or lateral movement of water around the sampler. The collapse of the interior box can be avoided by carefully monitoring the removal of water during a sample collection period. As soon as water is no longer being pumped, the vacuum must be removed. Otherwise, the interior box may be collapsed by the great cumulative force exerted by the applied suction. The disturbance of the surrounding area of soil introduced by the installation process can be significant and may cause the lateral movement of water or diversion of flow paths which will hinder the PCAPS from sampling matrix flow. Common technical failures of the suction cup samplers observed in this study included inconsistent volume collection, loss of vacuum, and complete inoperability. Several suction cups had to be replaced during the experiment. Typically, only 60% of the functional suction cups collected samples during each month. The performance of the suction cups is consistent with those reported for other field experiments.

The overall average PCAPS collection efficiency of 90% found in this study is consistent with collection efficiencies obtained for similar PCAPS experiments. This collection efficiency is a considerable improvement over studies done with zero-tension samplers. According to the pressure potential applied by the wicks the PCAPS should have over-sampled during periods of high flux ($P < 45$ cm of H_2O), when they in fact under-sampled for the duration of the study. The discrepancy may be due to unaccounted for runoff in the mass balance model or considerable flow occuring at high pressure potentials at which wicks were under-sampling or not sampling at all. However, the wick matching procedure shows that K_{sat} is the most important variable when matching the wicks to the soil types given this experiments PCAPS design. In fact, this is a very important procedure which must be done. Without matching the wick to the soil, flow paths may be disturbed by wicks which have much higher pressure potentials than the soil. On the field scale, K_{sat} no longer becomes a significant variable in defining the amount of percolation which travels below the root zone. The correlation coefficient between K_{sat} and PCAPS estimated flux was 0.01. There is no evidence to suggest that K_{sat} has a significant role in the collection ability of the PCAPS. K_{sat} is an estimate of the rate at which the water moves through the soil profile under the influence of a pressure gradient. The variable provides no information on the actual volume of water moving through the profile on a monthly basis. Thus, assuming that the PCAPS were to over-sample on the field scale from the wick matching results, may be inappropriate.

A better indicator for the PCAPS collection ability is the available volume of drainage water calculated using a hydrologic mass balance. For the special study sites, PCAPS monthly estimated recharge was found to be highly correlated to the mass balance monthly estimated recharge (average $R^2 = 0.75$). The analyses indicate that the PCAPS do sample the mass balance recharge, which is proven by the linear relationship between the two variables. Although a paired t-test on the means of the PCAPS annual recharge and mass balance annual recharge suggests a difference (p-value = 0.01 for special study sites), the PCAPS operated efficiently for all cases except during periods of high flux. As previously mentioned, unaccounted for interior field runoff due to ponding may be the cause for the discrepancy. An average regression equation was determined to define the PCAPS recharge estimates for the special study sites during the experiment

$$P(\mathrm{y}) = 8.60 + 0.68[M(\mathrm{x})] \tag{19}$$

where $P(\mathrm{y})$ is the PCAPS estimated recharge in [cm] and $M(\mathrm{x})$ is the mass balance recharge in [cm].

Paired t-tests on PCAPS at the same sites indicate that variability in recharge estimates at a single site do exist. Even though the PCAPS were placed as close to 1-2 m from one another, 17% of the sites had statistically significant differences in the mean PCAPS estimated recharge between samplers. In addition, the coefficient of variation for the mass balance estimated recharge at the special study sites was 80%. These variability estimates indicate that natural variation will always be a problem. To assess the groundwater recharge for a given region, many measurement must be recorded. To assume that two PCAPS at each site will accurately estimate the mean monthly recharge would be false. Variability such as that documented in this experiment exists. One individual PCAPS may be an inefficient sampler, but as a whole, several PCAPS are very efficient estimators of actual ground water recharge. As the number of samplers increases, variability decreases as with any population distribution. To estimate the mean monthly recharge at each site with a 30% bound on the mean and 95% confidence level, 20 samplers are needed per site. When two samplers are used, there is a 75% bound on the the mean recharge estimate. The estimate indicates that natural variation exists, but when proper designs and numbers are employed, PCAPS can be an invaluable device for assessing the ground water recharge under agricultural production.

Chapter IV. Passive Capillary Samplers (PCAPS) as Estimators of Ground Water Quality

Introduction

Protection of groundwater quality is of national concern, especially with respect to nitrate contamination. The USEPA's national survey of drinking water wells (USEPA, 1990) indicates that nitrate (NO_3^-) was the most commonly found contaminant with 57 and 52% of the rural wells and community water supplies, respectively, containing detectable concentrations. In 1984, 6.4% of examined wells in the U.S. exceeded the EPA water quality criteria for NO_3^--N of 10 mg L^{-1} (Madison and Brunett, 1984). In rural areas where groundwater is the source of drinking water for approximately 97% of the residents (USEPA, 1987a), groundwater quality is a subject of immediate importance. Nitrogen is the primary component of inorganic and organic fertilizers and transforms rapidly to nitrate under normal soil conditions (Alexander, 1965). The use of nitrogen fertilizers by U.S. farmers increased an average of 4% a year for the 1969-1979 period (Am. Chem. Soc., 1980). Applications of nitrogen fertilizers has reached an equilibrium in the past decade but high application rates continue. More land is being used for intensive agricultural production which requires increased application rates and increases the chances of leaching loss of N as NO_3^-. In contemporary agriculture nitrogen is added in sufficient quantities to achieve maximum yield, typically significantly in excess of that which the plants can take-up.

Nitrates are considered harmful in drinking water at concentrations above 45 mg L^{-1} NO_3^- (10 mg L^{-1} NO_3^--N), which is the U.S. public drinking water standard. Those at greatest risk are infants who are susceptible to methemoglobenemia (CAST, 1985). The drinking water standard of 10 ppm NO_3^--N was chosen because it was the concentration below which no case of infant methemoglobinemia had been identified (Walton, 1941). When ingested in high amounts, nitrates may have other adverse effects such as causing cancer (CAST, 1985). A number of reviews have been published concerning nitrate contamination and nitrate toxicity and health effects (Aldrich, 1984; Brezonik, 1978; CAST, 1985; Keeney, 1982; Viets and Hageman, 1971).

In the past few decades, the main reason for measuring or monitoring the loss of NO_3^- below the root zone has shifted from evaluation of the loss of NO_3^- from crop production or estimated loss of production, to an increase in the concern for NO_3^- in surface and groundwaters (Pratt et al., 1972). Much research has been done on the leaching of anions below the root zone, especially nitrogen (N), with relation to the climate, soil and crop. The most important chemical factor influencing the movement of nitrate in soils is that nitrate is highly soluble and anionic and thus very mobile in the soil. The two most important physical characteristics which influence the movement of nitrate in the soil are (1) the quantity of available drainage water (Pratt et al., 1972) and (2) the length of time that the water remains in contact with biologically active layers (Lind and Pedersen, 1976). The finer textured the soil, the larger the amount of drainage water needed to overcome the higher water storage capacity and cause the leaching of nitrates. In addition, finer textured homogeneous soils favor chemical processes such as exchange of anions and cations, adsorption, and denitrification. Drainage water can deplete soil reserves of mineral nitrogen or leach inorganic fertilizers, especially during the winter. During fall to spring fallow periods, natural precipitation may be adequate to leach NO_3^- below the root zone and into the ground water. As a general rule the greater the total winter rainfall, the greater the amount of nitrate being leached.

The movement of the dissolved nitrate ion through the soil is governed by three mechanisms: convection, and dispersion and diffusion. Convection, or mass flow, is the mass of solute per unit area per unit time being transported by the bulk movement of water. Purely convective flow is referred to as the "piston-flow" model in that the soil solution is displaced through the soil like a piston. Gravitational forces cause the soil solution to percolate from the soil when the water content is between saturation and field capacity (the condition at which no more water can drain from the soil by gravity). Under these conditions, nitrate will percolate with the soil solution according to the mass flow model. Soil solution can continue to be removed from the system but only by evaporation or plant-uptake. Diffusion and dispersion takes place constantly in the soil, which also moves nitrate through the soil. Through molecular diffusion, solutes, such as NO_3^-, are spread out through Brownian motion caused by molecular collisions which acts to transport mass from areas of high concentration to areas of low concentration (Jury and Nielsen, 1989). The diffusion pathway is tortuous in soils which reduces the amount of diffusion and dispersion in comparison to movement in pure water. The diffusion process typically occurs much more slowly than the mass flow process. Dispersion is the process by which solutes are smeared through the soil due to the variation in percolation rate from point-to-point. For diffusion and dispersion to give rise to mass movement requires a gradient in concentration, which are usually very small for nitrates. Therefore, the dominant transport mechanism influencing the movement of nitrate in the soil is convection.

Nitrate is an end product of the natural mineralization of the vast organic N-pool in the soil (Figure 19). Nitrogen transformations are brought about mainly by soil fauna, bacteria and fungi. Nitrogen fertilizers are typically applied in a known amount in the ammonium (NH_4^+) form. Clay particles and organic matter carry a negative charge on their surfaces, which bind ammonium ions to render them immobile. Nitrate ions which are produced from the ammonium, however, have a negative charge which repel them from soil surfaces, allowing movement with leached water. Ammonium can also be utilized by soil bacteria and converted to soil organic nitrogen. Organic materials, such as soil organic nitrogen, may also be biologically decomposed by bacteria and converted to inorganic forms (NH_4^+ and NO_3^-). This process is called nitrogen mineralization. In the case of a soil with decreasing soil organic matter, net N mineralization is positive and the supply of plant-available N increases. Significant amounts of N can be added to the soil plant system from nitrogen mineralization if soil properties and climatic factors are favorable. However, estimates of N mineralization should be viewed with uncertainty unless information regarding cropping history, fertilizer applications, soil chemical characteristics, climatic conditions and geographical information are available (Schepers and Mosier, 1991).

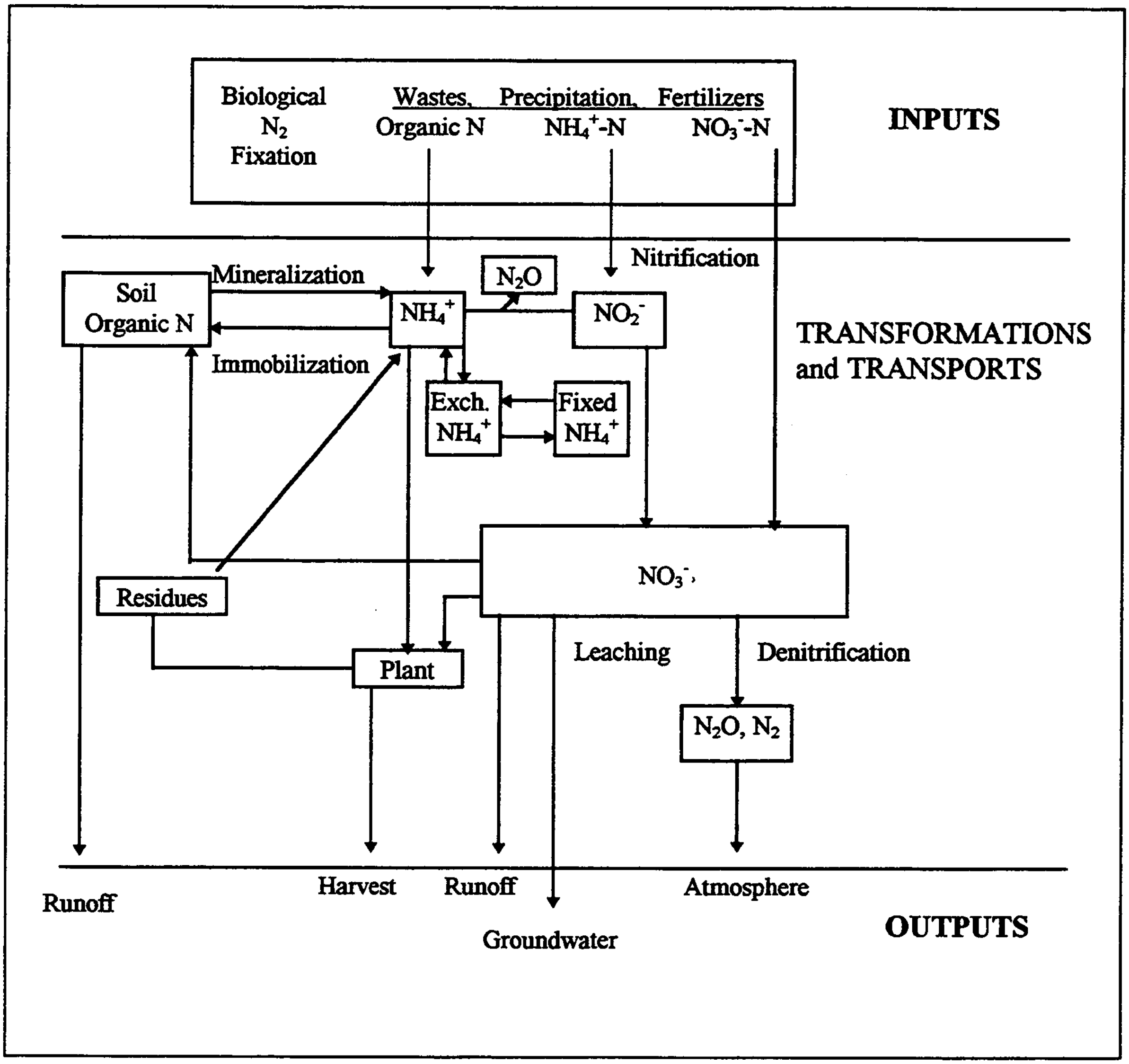

Figure 19. The nitrogen cycle (adapted from Keeney, 1989).

The most important N-biological reaction occurring in the soil is the conversion of ammonium to nitrite and further to nitrate by microbial oxidation. This process is called nitrification. Soil organic nitrogen can be mineralized to form ammonium, and urea [$CO(NH_2)_2$], another popular application fertilizer, can be converted to ammonia. Nitrate is more easily absorbed by plant roots during the growth cycle than ammonium. For this reason, the soil/plant system works to convert ammonium to nitrate to stimulate growth and diversity. The soil system is considered to be an ecosystem. An ecosystem works to sustain its environment by creating opportunities for all living organisms. Thus, the soil/plant ecosystem converts available ammonium to nitrate to stimulate growth of both the plants and the microbial biomass. The conversion creates opportunities for all organisms which creates a diverse and hopefully sustainable system. If organic matter is low, soil bacteria will mineralize more N in order to make more available to the plant roots. Nitrogen may also be added to the soil through symbiotic dinitrogen (N_2) fixation. This process converts atmospheric N_2 (80% of the earth's atmosphere) into plant N through symbiotic bacteria living in the root nodules of certain plants. Estimates of N_2 fixation tend to be rather crude and depend on many genetic and environmental factors including plant species, available soil N, soil water, and type of fixing bacteria (Phillips and DeJong, 1984).

Nitrogen is removed from the soil-plant system in three primary ways: (1) harvested product; (2) denitrification; (3) and leaching. As plants grow, they absorb the necessary N from the soil in order to produce a harvestable product. The N in harvested product can be determined by chemical analysis or may be estimated from literature values (Meisinger and Randall, 1991). The harvested N removals on an areal basis can be estimated by multiplying the amount of harvested per acre product by the N content per unit of harvested product.

Denitrification is microbial respiration that uses NO_3^- rather than oxygen and reduces N primarily to N_2 gas which is released to the atmosphere. Oxygen levels in the soil as well as climatic conditions play a key role in denitrification. Saturated or high water content soils have low oxygen levels promoting denitrification. Denitrification losses are usually small during much of the year with periodic major losses occuring when the soil is rewetted by rainfall or irrigation (Rolston et al., 1982). N budget estimates of denitrification often suggest values of 15 to 30% loss of added fertilizer N (Hauck, 1981); however, traditional N budget studies ascribe all N not recovered to denitrification (Meisinger and Randall, 1991). Ammonia loss through volatilization and leaching losses of NO_3^- may be added into the estimates of denitrification. Ammonia losses and denitrification also have a tendency to be higher under fields which are applied with animal manures. In addition, areas having warm, temperal climates are ideal for promoting denitrification. Significant sampling uncertainty requires better localized measurements to obtain representative estimates (Rolston et al., 1979).

Leaching losses of nitrate have been determined using many different methods (see Chapter III). From a management standpoint, nitrate leaching losses are minimized by maximizing the residence time of the chemical in the root zone (Jury and Nielsen, 1989). Thus, developing a precise irrigation schedule to meet the needs of the crop and reduce leaching has become an important method for controlling N loss. In addition, maximizing the efficiency of fertilizer use is key to minimizing any environmental impact (Barry et al., 1993). As mentioned before, much of the NO_3^- remaining in the profile in the fall is susceptible to leaching. The amounts of applied N recovered by the crops plus the residual in the soil in field experiments often range from 70-90% (Allison, 1966). In Colorado, Smika et al. (1977) used suction cups under corn and found that annual NO_3^--N leaching losses ranged from 19 to 60 kg ha^{-1} depending on the amount of percolation. In Nebraska, Mielke et al. (1979) used vacuum extractors under corn and found leaching losses of 31 to 64 kg of NO_3^--N ha^{-1} (May-September). For irrigated corn on a sandy loam soil, Gerwing et al. (1979) reported ground water $N0_3^-$-N concentrations increasing 7 and 10 ppm for applications of 179 and 269 kg of N ha^{-1}, respectively. In Oregon, Brandi-Dohrn et al. (1993) used PCAPS to study the immediate response of recharge concentrations on fields with and without cover crops. PCAPS were able to sample both matrix and flux concentrations and proved superior to suction cup samplers also used in the experiment. Fields planted with cover crops were found to reduce recharge nitrate concentrations by 12%, reduce leachate volume and reduce the loss of nitrogen by 16.5 kg ha^{-1} yr^{-1}. Crop management (tillage, irrigation, crop, fertilizer) play a significant role in determining the amounts of NO_3^- that are potentially being leached below the root zone.

Contamination of ground water under agricultural land by NO_3^- is influenced by the cropping system (Barry et al., 1993). One method for evaluating the amount of NO_3^- losses to ground water is by calculation of N budgets for farming systems. Due to the number of parameters associated with accurately estimating all the components involved in a soil N budget (immobilization, mineralization), constructing a precise estimate of the amount of N available for leaching or denitrification after the growing season may be impossible for many farmers. However, the N budget may be simplified by assuming that the soil organic matter or soil N content, remains constant on a yearly basis for crop rotation systems. This simplified N balance approach for predicting long-term effects of farming systems was described in detail by Fried et al. (1976). Their model relied on using some basic assumptions which allow their model to be used for evaluating NO_3^- leaching. The assumptions are (1) the pool of organic N in the soil was constant from year to year so that net mineralization or immobilization was zero, (2) the rate of movement of NO_3^- below the root zone was equal to the movement of water (e.g., convection dominated), and (3) the amount of N denitrified or leached was equal to the total N input minus the sum of removal in crop plus that found in soil. They stated that any continued agricultural practice will result in the soil N content reaching a steady state level. The transfer of N to the ground water should then equal the difference between N inputs and N outputs. Essentially when the soil/plant system (SPS) is at steady state, the inputs (I) must equal the outputs (O), and the leaching loss (L) will equal I - (D + HP + PL), where D equals the amount of N denitrified, HP equals the amount of N in the harvested product removed from the system, PL the amount of N lost physically (runoff, erosion, volatilization, wind), and I = F + NA, where F equals fertilizer and irrigation additions of N, and NA equals natural additions. Natural additions may include precipitation, fixation, and direct NH_3

adsorption. These inputs are usually quite low compared to fertilizer additions unless NH_3 concentrations in the air become high and large amounts are absorbed by the plants.

The N outputs for Fried et al.'s system include denitrification, harvested product, and physical losses. Again, the physical losses tend to be small compared to the removal of harvested product, especially if fertilizer is used. Thus, one can assume that NA - PL = 0 based upon the previous discussion. If one accepts the steady state concept it is unnecessary to quantify the intricate transformations of N in the soil. Also, many of the factors used in Fried's SPS can be assumed at steady state and possible leaching losses can be determined. For this system, the maximum amount of N that can potentially leach beyond the root zone is F + NA - (HP + PL). If all assumptions are correct, this value can be estimated by (F - HP). If the amount of N that is denitrified is known, this estimate of L will further decrease.

Similar research was conducted by Pratt et al. (1972) in which the same assumptions and theory were used to account for natural additions and physical losses of N below the root zone. However, instead of using actual inputs and outputs, their model incorporates the use of a mass balance and leachate concentrations to estimate leaching losses of N. In other words, this model requires percolation amounts and concentrations to estimate the amounts of N lost to leaching in kg ha^{-1} yr^{-1}. When using PCAPS, the amount of drainage water need not be quantified, yet, Pratt et al. had to empirically quantify the amount of drainage water using

$$D = \frac{\overline{LF} \cdot \overline{ET}}{1 - \overline{LF}} \tag{20}$$

where D is the volume of drainage water expressed in cm, $\overline{ET}$ is the evapotranspiration in cm, and $\overline{LF}$ is the leaching fraction given by

$$\overline{LF} = \frac{P_d}{(I + R)} \tag{21}$$

where P_d is the percolation in cm, I is the irrigation and R is the rainfall in cm. The amount of excess N available for leaching is calculated as

$$N_{excess} = \frac{D \cdot NO_3}{10} \tag{22}$$

where N is expressed in kg/ha per year, and the NO_3 is the concentration in ppm of NO_3^--N in the soil water below the root zone. The units for the constant (10) are mg cm ha kg^{-1} $liter^{-1}$. The amount of time for water to reside in the unsaturated zone is calculated as

$$T = \frac{S\theta}{D} \tag{23}$$

where T is transit time in years, S is the soil depth in cm, and θ is the volumetric water content.

The concept of Fried et al.'s (1976) model was tested by Tanji et al. (1977) with corn at two sites in California. Predicted NO_3^- concentrations in leachate were compared with measured NO_3^- concentrations at various depths in the soil profile. They concluded that NO_3^- losses at one site were approaching steady state, but predicted NO_3^- concentrations for the other sites were less than measured. Lund (1982) suggested that even where the assumption of steady state was invalid, results can still be indicative of the effects of certain soil and crop management practices. Macduff and White (1984) tested the approach using arable and grassland soils in England. They found that there was net mineralization of organic matter in their soils therefore the simple N-budget was inadequate. However, Barry et al. (1993) point out that N inputs from grazing cattle were not accounted for in Macduff and White's (1984) budget.

Pratt et al. (1972) also applied their model on soils in California. NO_3^- concentrations in soil cores are compared with excess N in the soil, calculated as N input minus crop removal. Using their model, they concluded that for open-porous soils and inputs of N around 150 kg ha^{-1} reasonable estimates of the NO_3^- concentrations reaching the saturated zone could be made. However, at higher input rates on porous soils or low inputs on soils with textural discontinuities, denitrification, which is an uncalculated variable, was assumed to be the cause for problems in the estimates. Adriano et al. (1972) applied the concepts in California on soils with asparagus and celery. As before, denitrification was the variable used to account for under estimations of N lost on soils receiving high rates of N inputs as well as high levels of water use.

The concepts of Pratt et al. (1972) and Fried et al. (1976) are often taken as the basis for development of best management practices (BMPs). Often, BMPs are developed to minimize NO_3^- inputs to ground water, and to be useful must encompass a wide variety of crop and soil management options along with socio-economic and regulatory activities (Keeney and Follett, 1991). The main principle is to minimize the amount of NO_3^- available for leaching in the root zone.

The theories used by Fried and Pratt lead to similar conclusions. The important factors for evaluating NO_3^- between the root zone and saturated zone are (1) the volume of drainage water, (2) the yearly or monthly excess of NO_3^- available for leaching, and (3) an estimate of denitrification. The researchers stressed that as the amounts of N added increased, the amount of N available for leaching also increased. They indicate that the key to minimizing agriculture's contribution to nitrate in ground water is the efficient use of fertilizer as indicated by the proportion of added N which is removed by the harvested portion of the crop. Only after maximum yield is reached does the utilization of additionally applied N decrease. Thus, the relationship between leaching loss potential and the amount of fertilizer applied is a function of fertilizer use efficiency and what farmers consider efficient. Fried et al. (1976) show that the absolute amount of N subject to leaching is not as dependent on the levels of nitrogen in the crop as the efficiency of nitrogen use by the crop. Therefore, the researchers provide substantial information and data to indicate that high N fertilizer use efficiency and efficient irrigation applications are required to achieve maximum production with minimal effect to the ground water.

The objectives of this study are to evaluate the leaching losses of NO_3^- below the root zone concerning (1) the ability of PCAPS to estimate the mean NO_3^- concentrations of several different management practices under natural conditions; to evaluate (2) the major cropping systems employed in Lane county based on each management systems contributions of NO_3^- to yearly ground water recharge; and (3) to provide preliminary approaches to best management practices for systems which appear to lead to the largest adverse annual environmental impacts.

Results

Estimation of NO_3-N Composition of Soil Solution

Flow-Weighted Concentrations

In order to accurately estimate the quality of agricultural leachate, values for the quantity of drainage water as well as the drainage flux must be obtained. The drainage flux must be measured and sampled in a timely sequence in order to estimate flow-weighted concentrations. Flow-weighted concentrations are important because of the great variation in both concentration and flux, which are often correlated. The flow-weighted concentration gives an accurate indication of the average quality of ground water recharge. The concentration of the leachate is averaged according to the volume of drainage leached over the period of sample collection. If more than one sampler are present the flow-weighted average concentrations can be calculated using Equations 8 and 11. As discussed in Chapter III, PCAPS appear to be the best sampling means available to obtain representative estimates of leachate concentration because of their ability to sample ground water recharge. When suction cup samplers are used, however, the mean solute concentration can only be estimated arithmetically since suction cups provide no flux data.

The arithmetic mean is a biased estimate of the true flow-weighted concentration because volume and solute chemical composition are not independent. Typically, the NO_3-N content of the percolating soil solution, flux concentration, is not in equilibrium with the soil matrix NO_3-N concentration, resident concentration. Percolating water moves through the vadose zone leaching NO_3-N faster than the soil matrix can equilibrate concentrations with the leachate. If concentrations in the flow pathways are higher than concentrations in the soil matrix, the flux concentration is greater than the resident concentration. Assuming this is true, arithmetic averages of NO_3-N would be biased only when the soil solution flux is high and the flux and resident concentrations are not able to equilibrate

To demonstrate this concept, the correlation between NO_3-N concentration and flux were calculated for our data. Nitrate concentrations and flux for all sites were normalized by dividing by the mean concentrations to correct for differences between treatments. Figure 20 depicts the correlation coefficients for each month of the experiment: nitrate concentration and flux cannot be correlated in an easily identifiable pattern. The effects of high or low flux and varying concentrations are not very pronounced suggesting that the soil nitrate and moisture content are evenly distributed for the study period. In addition, there is little evidence to suggest that for times of larger flux, the flux concentration is higher than the resident concentration. Figure 20 shows that for this study, nitrate levels were more dependent upon soil and crop type than the amount of water moving through the profile. Even though flux is not positively correlated with nitrate concentrations ($r = -0.53$ for the study period), it is the main mechanism by which nitrate is transported through the profile, and estimation of this drainage volume is key for assessing a management systems environmental impact.

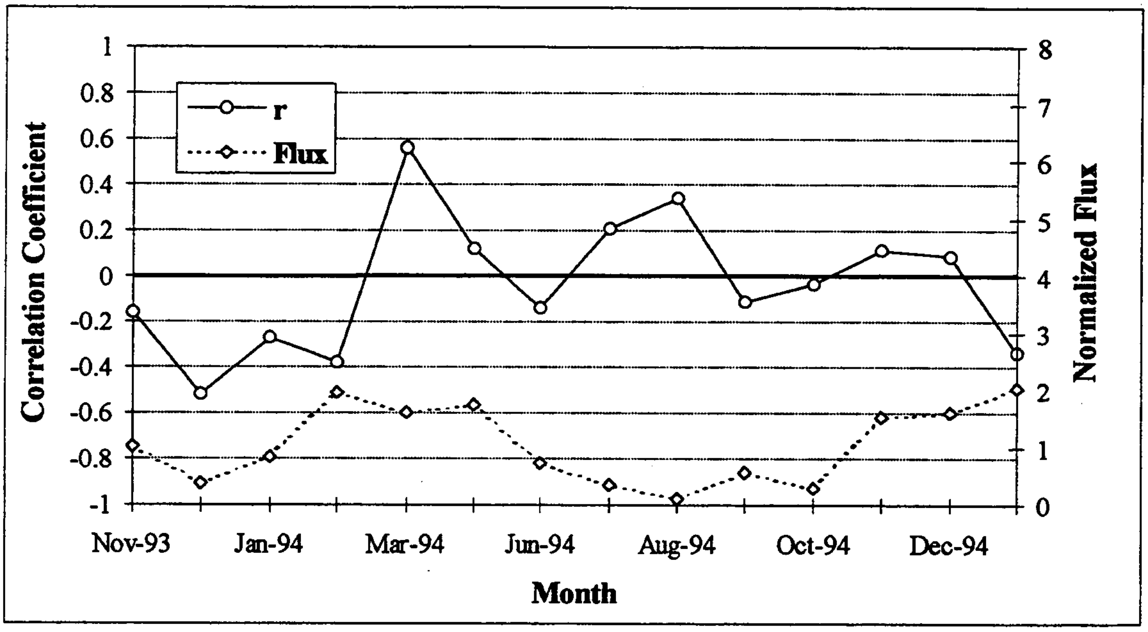

Figure 20. Correlation coefficient between normalized flux and NO_3-N concentrations, and mean flux.

Due to the fact that flux and NO_3-N concentrations are not correlated for the study period, the two methods for calculating the mean nitrate concentrations would not be expected to significantly differ for all treatments. Table 20 lists the mean NO_3-N concentrations as calculated by both the arithmetic and flow-weighted methods. For the entire study period, the two methods to calculate the mean did not appreciably differ for all treatments based on a paired t-test of the means. On the average, the arithmetic mean underestimated the flow-weighted mean by 6% for all sites, and ranged from overestimating by 48% and underestimating by 35% on a site-by-site basis (Table 20). The results of the means analysis further illustrate that the NO_3-N concentrations are not positively correlated with NO_3-N flux. This is not to say that collected volumes and solute content are independent for all situations in this study, but rather, that NO_3-N levels are distributed evenly as well as soil moisture through-out the profile. If tracer tests were performed, most likely a positive correlation would be observed between the tracer and the tracer flux for the early portion of the study, and negatively for the later portion of the study as found by Brandi-Dohrn (1993).

Table 20. Comparison of arithmetic and flow-weighted means of NO_3-N concentrations.

Site	N rate	Mean: Arithmetic	95 % C.I.	Flow-Weighted	95 % C.I.	% Diff. Range
	[kg ha^{-1}]	[mg L^{-1}]				
Blueberry #1	102	5.30	1.23 to 9.37	6.70	0.29 to 13.11	-79 to 37
Blueberry #2	76	4.37	0.60 to 8.14	6.07	0.01 to 12.13	-50 to 50
Grass #1	156	11.31	6.87 to 15.75	11.24	6.52 to 15.96	-88 to 10
Grass #2	156	28.31	20.40 to 36.22	28.57	20.65 to 36.49	-2 to 7
Mint #1	250	24.02	12.55 to 35.49	37.09	21.19 to 52.99	-
Mint #3	280	11.79	7.30 to 16.28	12.08	7.63 to 16.53	-17 to 25
Mint #4	370	32.04	19.94 to 44.14	32.33	20.53 to 44.13	-31 to 32
Orchard #1	0	3.28	1.21 to 5.35	3.64	1.24 to 6.04	-57 to 34
Orchard #2	45	3.21	0.61 to 5.81	3.55	0.96 to 6.14	-25 to 25
Organic #1	180	17.27	7.44 to 27.10	11.69	6.67 to 16.71	-137 to 50
Organic #2	180	35.51	19.16 to 51.86	35.89	19.34 to 52.44	-1 to 7
Row Crop #2	170	18.95	7.93 to 29.97	19.80	9.00 to 30.60	-0.04 to 50
Row Crop #3	135	22.18	15.26 to 29.10	23.70	16.83 to 30.57	-4 to 7
Row Crop #4	180	28.98	14.03 to 43.93	31.91	14.48 to 49.34	-12 to 32

Comparison of NO_3-N Concentration Measurements as Obtained by Suction Cup Samplers and PCAPS

As discussed in Chapter III, suction cup samplers tend to sample water held at more negative pressures resulting from smaller pores preferentially (Hansen and Harris, 1975). If the suction cup samples preferentially, only small portions of the flux concentrations and mostt of the resident concentrations are being sampled. PCAPS, on the other hand, have the ability to sample both resident and flux concentrations. In addition, PCAPS are able to sample continuously which allows them to collect in a representtative manner large pulses of solute during times of heavy percolation. These pulses are key to estimating the true flux concentrations. Suction cups are unable to sample continuously unless a vacuum is applied in a timely sequence. Pulses of flux concentration will typically bypass the suction cup sampler which is unable to sample solute flux. During periods of low flux, suction cups will continue to sample from the smaller pores by pulling water towards the sampler at a rate independent of the native soil flux. This suggests that the suction cup sampler would underestimate the true, flow-weighted mean if flux concentrations are higher than resident concentrations and overestimate the mean if resident concentrations are higher than flux concentrations.

To demonstrate this concept, a comparison between NO_3-N concentrations as sampled by the suction cups and PCAPS was performed. Recall that suction cup samplers can only estimate the arithmetic mean of the concentrations; therefore, the comparison is made between the arithmetic mean of the suction cups and the flow-weighted mean of the PCAPS. All sites are used for the comparison, thus the NO_3-N concentrations are normalized by dividing by each sites mean concentrations. First, the natural variability in the concentrations was addressed. For this comparison, variability in differences between adjoining suction cups and PCAPS as well as differences between the same samplers was investigated. The "differences" are simply the absolute value of the difference in normalized concentration between pairs of sampling devices on the same site. PCAPS and suction cups are taken individually and not averaged (Table 21). There is evidence to suggest that the variability of the differences in concentration are of the same magnitude between suction cups and PCAPS and between individual suction cups. This result indicates that the NO_3-N concentrations in the soil vary due to the nature of the soil. Not only are the variability in the differences of the same magnitude between the suction cups but also between the PCAPS. Thus, to assume that the suction cups are not sampling the same concentrations as the PCAPS simply based upon the difference between NO_3 values would be a false conclusion. There is no evidence to suggest that the PCAPS reduce the variability in the NO_3-N measurements, and, in addition, there is no evidence to suggest the suction cups are not representatively sampling the true NO_3-N concentrations. The results of the variability analysis further reinforce the conclusion that NO_3-N levels reaching the ground water may be more dependent upon the natural soil variability and crop type rather than actual soil flux.

Table 21. Comparison of the variability's of differences between normalized NO_3-N concentrations as sampled by suction cups and PCAPS.

Differences Between:	Mean Difference	Variance of Difference
Suction Cups	0.79	1.30
PCAPS	0.71	0.98
Suction Cups #1 & PCAPS	0.73	0.65
Suction Cups #2 & PCAPS	0.94	1.78

An investigation of the deviation in nitrate concentrations through-out the study period was also performed (Figure 21). According to the above reasoning, if flux and solute content are not independent, suction cup means should be lower for periods of high flux and high for periods of low flux, resulting in a negative correlation between the two. To investigate this hypothesis, percent deviations in suction cup arithmetic averages and PCAPS flow-weighted averages was done. There is evidence to suggest that solute flux does play an important role in estimating NO_3-N concentrations. There is negative correlation ($r = -0.63$) between the % deviation in suction cup NO_3-N concentrations and solute flux, for all months. Therefore, it is evident that the suction cups NO_3-N concentrations' variability may be a result of underestimation of NO_3-N during large percolation events and overestimation of NO_3-N during periods of little or no percolation. Consequently, suction cups prove themselves to be an inferior method for estimating ground water quality due to their inability to sample solute flux. Although Figure 20 also demonstrates that PCAPS estimated concentrations are independent of flux ($r = -0.53$), there is still evidence to suggest (i.e. non-correlation of suction cup measurements) that without estimating solute flux, true NO_3-N concentrations cannot be calculated despite the fact that natural soil variation and crop types have a significant effect upon recharge concentrations. For estimating tracer or pesticide concentrations (compounds not distributed evenly throughout the profile), it is very important to sample flux concentrations because these compounds are typically released in pulses along with the leached water.

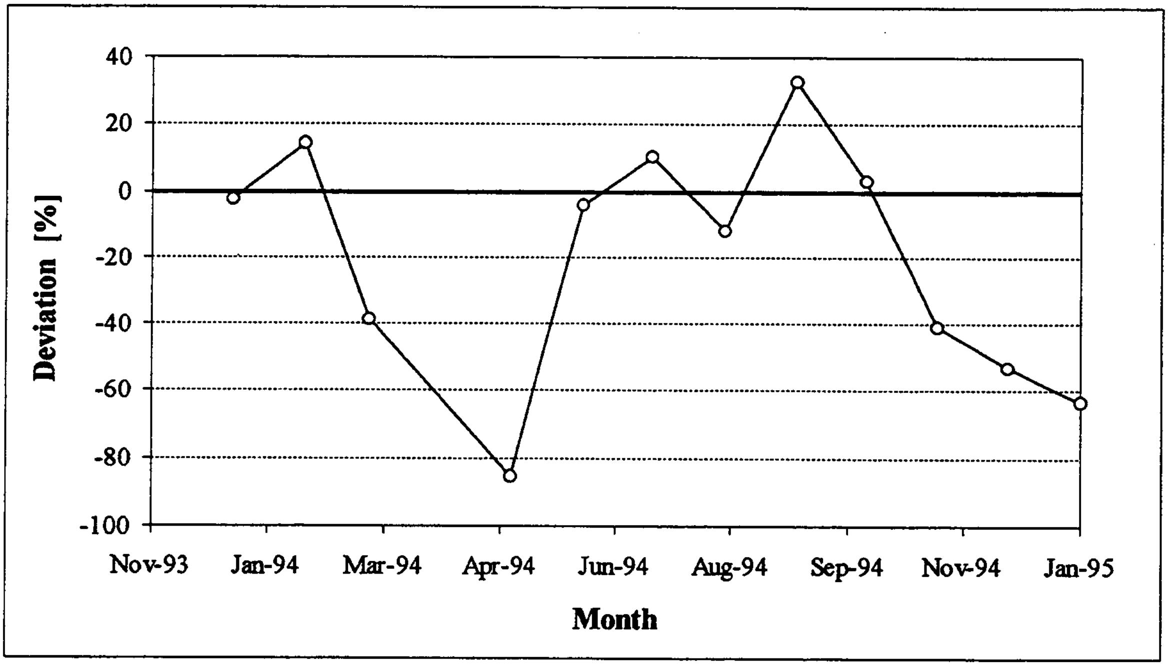

Figure 21. Deviation of mean NO_3-N concentration in suction cup samplers from flow-weighted mean NO_3-N concentration in PCAPS.

Estimation of Nitrogen Loss for each Management System

Amount of Percolate

The volume of drainage water is key to assessing an agricultural management system's impact on the environment. Excessive amounts of percolate can leach mineral reserves of nitrogen and inorganic fertilizers. In Western Oregon, the winter months play a central role in the transport of nitrogen to the ground water. However, excessive applications of irrigation water beyond the ET capacity of the summer crop may leach as much nitrogen as winter rainfall. Figures 22 and 23 depict the mean monthly and cumulative amounts of percolation for each management system as measured by the PCAPS. There is evidence to suggest (paired t-test between management systems, p-value < 0.05) that the amount of percolation was significantly higher under the mint management systems compared to all other management systems. There were no significant differences detected between the mean percolation amounts at all other management systems. The reason for mint having significantly higher amounts of percolation is obvious to anyone visiting the fields: The fields were often irrigated to the point of standing water. The amount of percolation during the months of irrigation (typically May - September) was significantly higher for mint than any other management system. Peppermint is a shallow rooted crop which has a poor water use efficiency. Growers of mint appear to irrigate at high levels (sometimes for eight hours at a time) to stimulate high yields for this cropping system. For other management systems, farmers would irrigate enough to wet the rooting zone and supply water for the growing crop. On the average this irrigation amount was around four cm of water per week. Orchard, row crop and organic crops were able to use the water efficiently with little water percolating beyond the root zone. The result of over-irrigation is increased loss of soil nutrients during the summer as well as the winter. It appears that all cropping systems leached similar amounts of soil solution during the winter season reflecting similar rainfall patterns throughout the Eugene, OR area (C.V. = 8.5%; Table 22). Summer percolation differed drastically due to its dependence on management practices (C.V. = 22%; Table 22).

Table 22. Comparison of winter, summer and average percolation as collected by PCAPS for all sites excluding Mint #1 whose PCAPS did not sample actual percolation.

Site	Percolation Amounts		
	Cum. Winter	Cum. Summer	Average
	[cm]		
Blueberry #1	65.9	4.78	5.05
Blueberry #2	91.1	23.9	8.21
Grass Seed #1	62.5	17.7	6.68
Grass Seed #2	58.0	23.8	6.81
Mint #3	65.6	30.1	6.83
Mint #4	138	60.3	14.2
Orchard #1	80.0	27.9	7.71
Orchard #2	65.1	19.8	6.06
Organic #1	66.5	20.9	6.24
Organic #2	54.2	0.50†	3.91
Row Crop #1	53.3	2.04	3.95
Row Crop #2	34.0**	5.71	3.31
Row Crop #3	94.0	11.0	7.44
Row Crop #4	72.1	14.1	6.16

** - data unavailable for December, 1994 and January, 1995

† - non-irrigated crop for summer of 1994

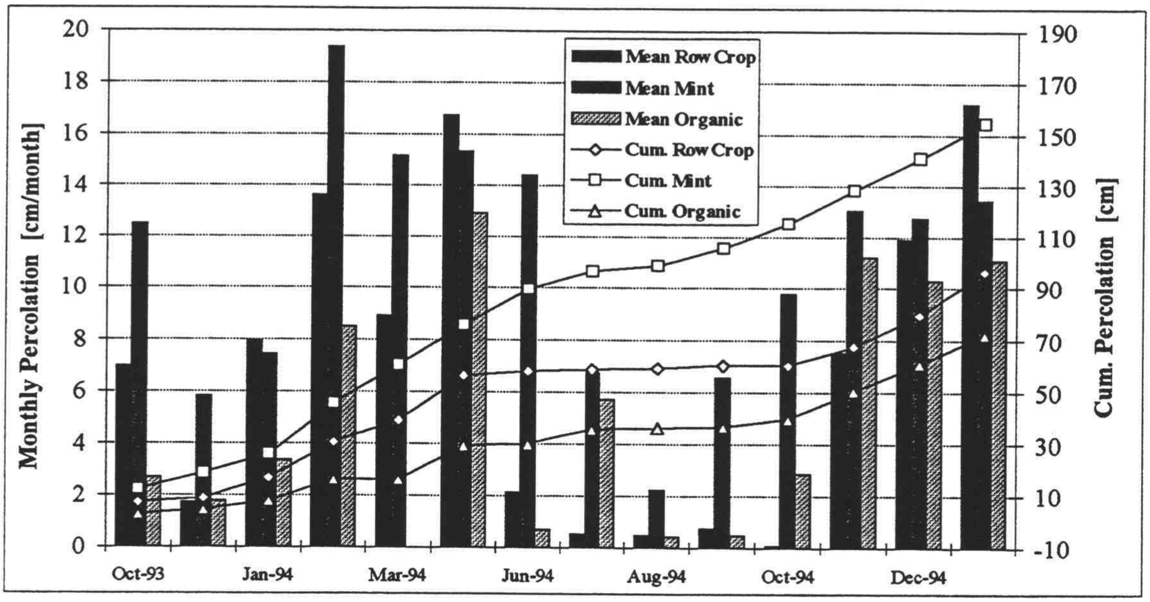

Figure 22. Average monthly and cumulative percolation for the mint, row crop and organic management systems.

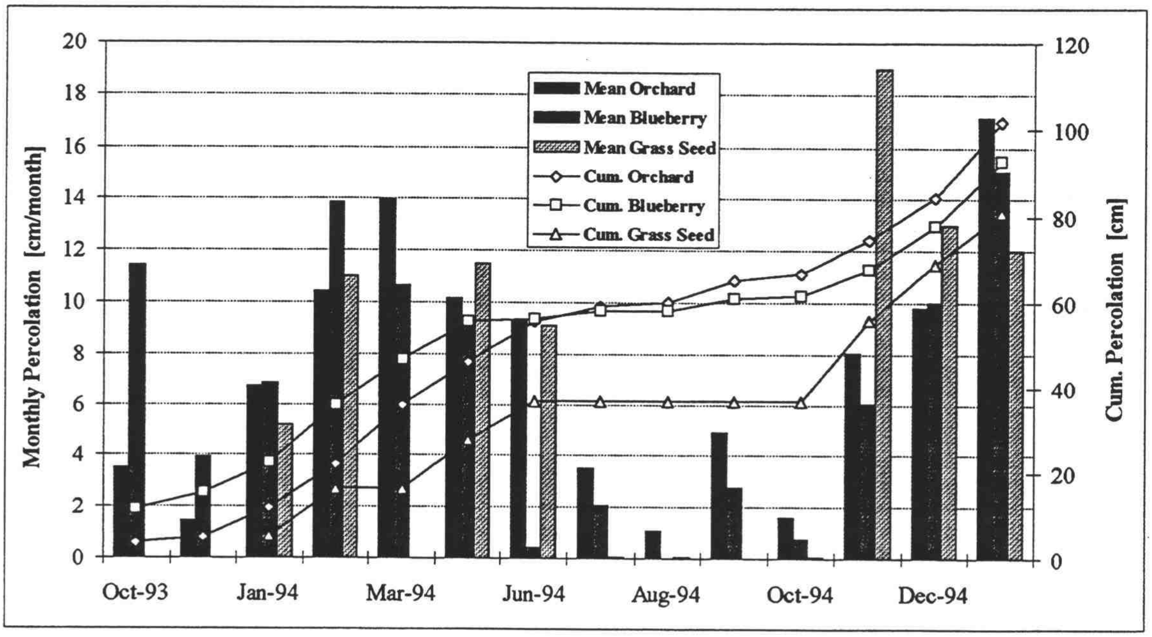

Figure 23. Average monthly and cumulative percolation for the blueberry, grass seed and orchard management systems.

Recharge NO_3-N Concentrations

Quantifying the amount of percolate is an essential prerequisite to estimating the mass loss of nitrogen from a cropping system. PCAPS are able to sample the volume of leachate allowing for correct estimates of both the chemical composition of the leachate and the amounts of nitrogen possibly leaching to the ground water. Previous crop types, residual soil nitrogen content, fertilizer types and times of application all play key roles in determining the amounts of nitrogen lost to the subsurface. Quantification of residual soil nitrogen and effects of previous crops requires lengthy research (see Figure 19). For this analysis, the amounts of nitrogen lost to leaching will be based solely upon the amounts of percolate and its chemical composition reflecting the integrated effect of management factors.

Using the theory described by Pratt et al. (1972) (Equation 22), the amount of nitrogen lost to leaching can be estimated in kg ha^{-1}. When PCAPS are used, the amount of drainage water need not be calculated as described by Pratt et al. (1972). Instead, the percolation as measured by the PCAPS can be used in place of the variable D to estimate the mass loss of nitrogen. The variables needed to estimate the mass loss composition are both the NO_3-N concentrations and volumes of leachate. Figures 24 - 27 depict the monthly, flow-weighted NO_3-N concentrations for all management systems. All of the mint sites exceeded the EPA standard of 10 ppm NO_3-N. The average annual recharge concentrations for the Mint #1, #3 and #4 sites was 28, 13 and 32 ppm respectively. The variability in the management practices suggests that certain farmers are able to control their nitrogen applications such that lower recharge concentrations are established. In addition, with the mint management systems, a majority of the high NO_3-N concentrations occurred during the summer rather than winter season. This points to the fact that greater amounts of nitrogen are being lost to the subsurface as result of over-irrigation and fertilization of a poor efficiency crop type. Mass loss of nitrogen throughout the year is discussed in more detail in the next section.

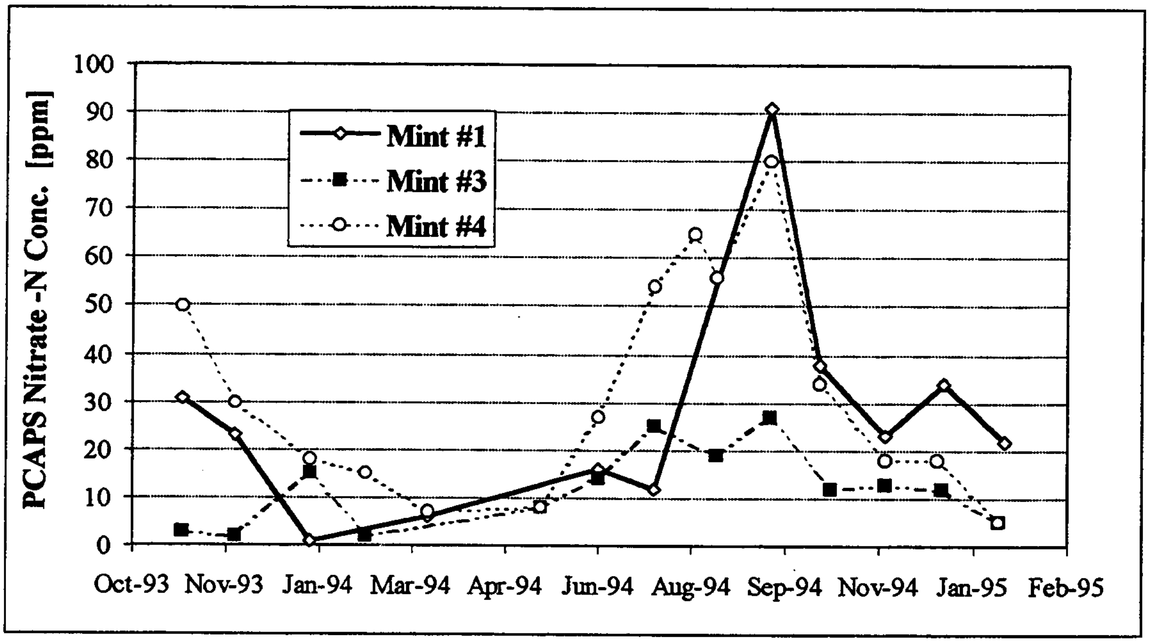

Figure 24. Flow-weighted NO_3-N concentrations in ground water recharge as sampled by the PCAPS for the mint management systems.

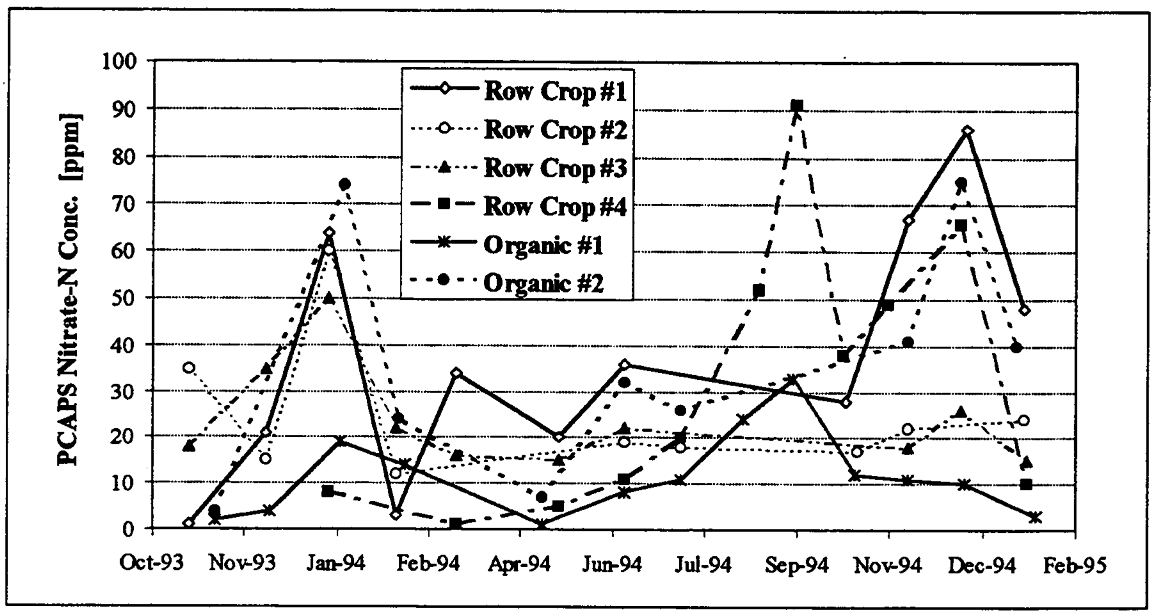

Figure 25. Flow-weighted NO_3-N concentrations in ground water recharge as sampled by the PCAPS for the row crop management systems.

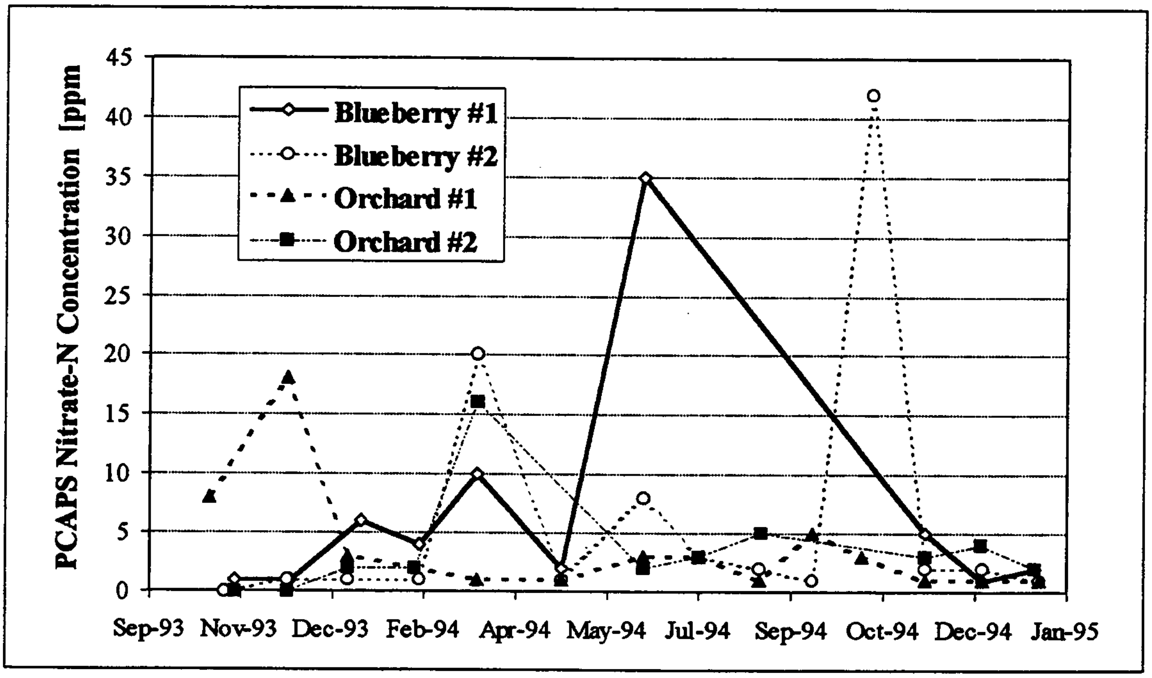

Figure 26. Flow-weighted NO_3-N concentrations in ground water recharge as sampled by the PCAPS for the orchard and blueberry management systems.

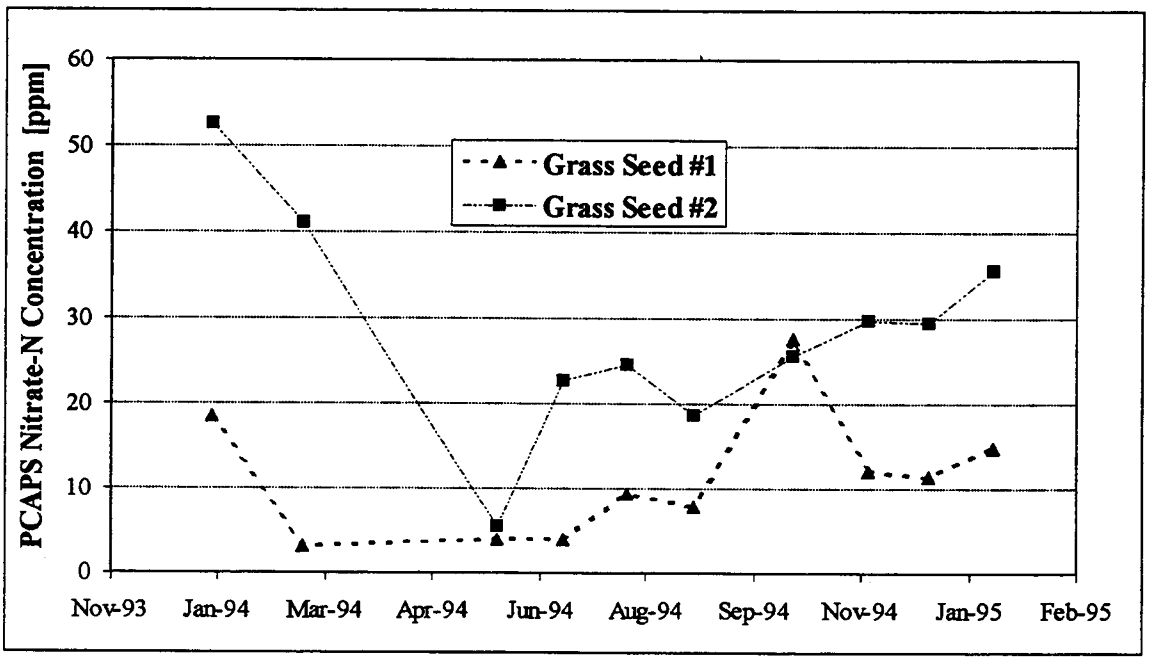

Figure 27. Flow-weighted NO_3-N concentrations in ground water recharge as sampled by the PCAPS for the grass seed management systems.

Six row crop producers, four conventional and two organic, also had NO_3-N concentrations in excess of the EPA standard. Row crops #1, #2, #3, #4 and Organic #1 and #2 had yearly recharge concentrations of 36, 25, 25, 34, 12, and 35 ppm respectively. These concentrations are very similar to those recorded under the mint production systems, although the pattern of leaching for row crop producers is quite different than that of mint producers. For the most part, row crop producers leached a majority of the high NO_3-N concentrations to the subsurface during the late fall and early winter seasons. This suggests that proper management of water during the summer had reduced the loss of applied water, but that a large residual soil nitrogen content was carried into the wet season resulting in the loss of large amounts of nitrogen flushed out with fall percolation. The grass seed producers also appear to have adverse environmental impacts with Grass Seed #1 and #2 having yearly recharge concentrations of 10 and 28 ppm. However, for the site #2, there is a direct relationship between the previous crop type and recharge concentrations: For the year prior to the production of grass seed, mint was planted on this field. It is assumed that the high recharge concentrations recorded at this site are a result of the mint production and not the grass seed production.

The remaining four management systems for investigation are the blueberry and orchard production systems. Low levels of nitrogen application (Appendix B) at these sites is reflected in the low yearly recharge concentrations. The yearly recharge concentrations for Blueberry #1 and #2 and Orchard #1 and #2 were 7, 6, 4, and 4 ppm respectively. The results of the leachate monitoring from these production systems suggest that the orchard and blueberry production systems have minor environmental impacts. For these two systems, it appears an early application of nitrogen may be lost during March which would suggest trying a later time of application. However, due to the rooting systems of these two crops, controlling the loss of any fertilizer application during the year may be impossible. Orchard #1 which has well established trees applies no fertilizer to their crop. Even with no fertilizer application, the soil will produce and lose organic nitrogen on a yearly basis. From the NO_3-N concentrations, there is evidence to suggest that the soil may provide enough nitrogen to sustain these management systems without any additional applications. In any case, the results of the monitoring of recharge concentrations provide a solid conclusion. To address the nitrogen problem in the rural agricultural areas of Lane County, focus must be placed upon the mint and row crop production systems as primary production systems contributing to adverse environmental impacts.

Knowing the composition of nitrogen in the ground water recharge is important, for in time, these concentrations will be reflected in the ground water system. This hypothesis is supported by the results shown in Figure 28 where each sites sampled well NO_3-N concentrations are plotted against the yearly average recharge concentrations. Linear regression on the data set provides a positive correlation between the two concentrations (R^2 = 0.58). Given the variation between well depths and site characteristics, this correlation is as strong as we could expect. The equation of the line also provides an interesting result

$$W[y] = 1.0 \bullet R[x] - 13.1 \tag{24}$$

where $W[y]$ is the well NO_3-N concentration, and $R[x]$ is the yearly average recharge NO_3-N concentration. The one-to-one slope between the estimated recharge concentration and well concentration indicates the effects of the recharge concentrations on subsurface water quality. Although the R^2 value is not conclusive, there is still evidence to suggest that prolonged agricultural management at these sites has lead to well NO_3-N compositions reflective of the local management practices.

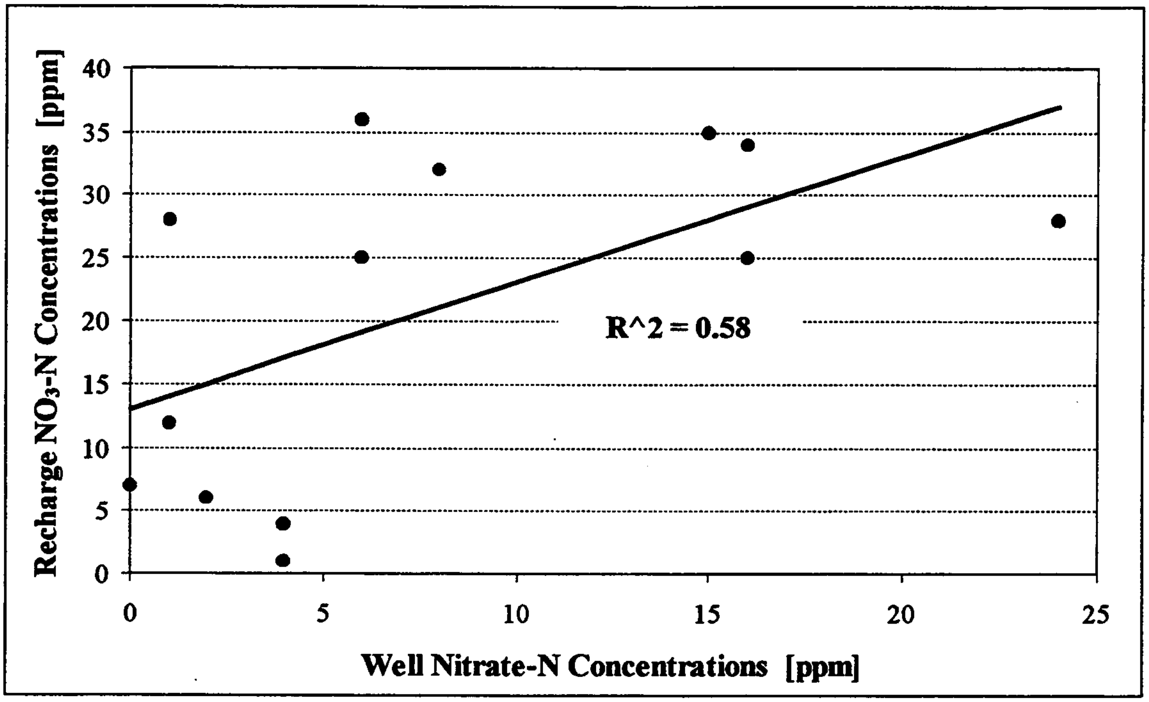

Figure 28. Comparison of well and yearly average recharge NO_3-N concentrations for all sites.

Mass Loss of Nitrogen

Equation 22 was used to calculate the mass loss of nitrogen from each management system (Figures 29, 30, 31 and 32). Due to the high recharge NO_3-N concentrations and percolation volumes, the mint management systems lost the most nitrogen during the year. Because the PCAPS at Mint #1 sampled estimated percolation inefficiently, percolation amounts from the Mint #3 site were used to estimate the mass loss of nitrogen at this site (rain gauge data was not taken on this site). Using non-exact percolation estimates may be biased since Mint #3 was a site which irrigated much less than normal mint operations. However, because Mint #1's recharge concentrations were very similar to Mint #4 (Figure 24), using "efficient" percolation measurements does not appear to be biased from the researchers stand point.

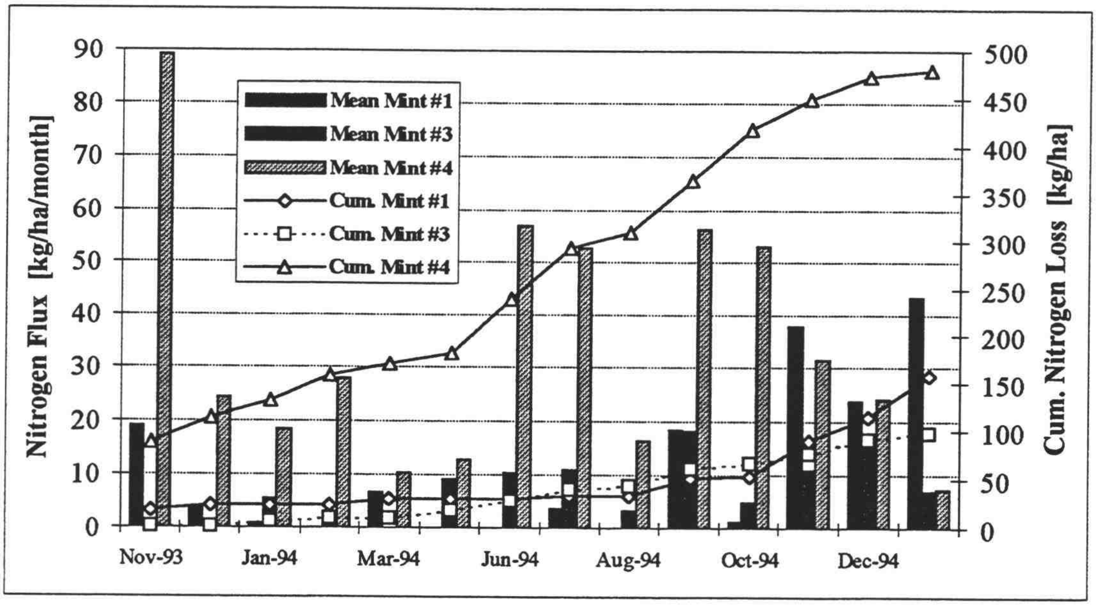

Figure 29. Monthly and cumulative mass losses of nitrogen as measured by the PCAPS from the mint management systems.

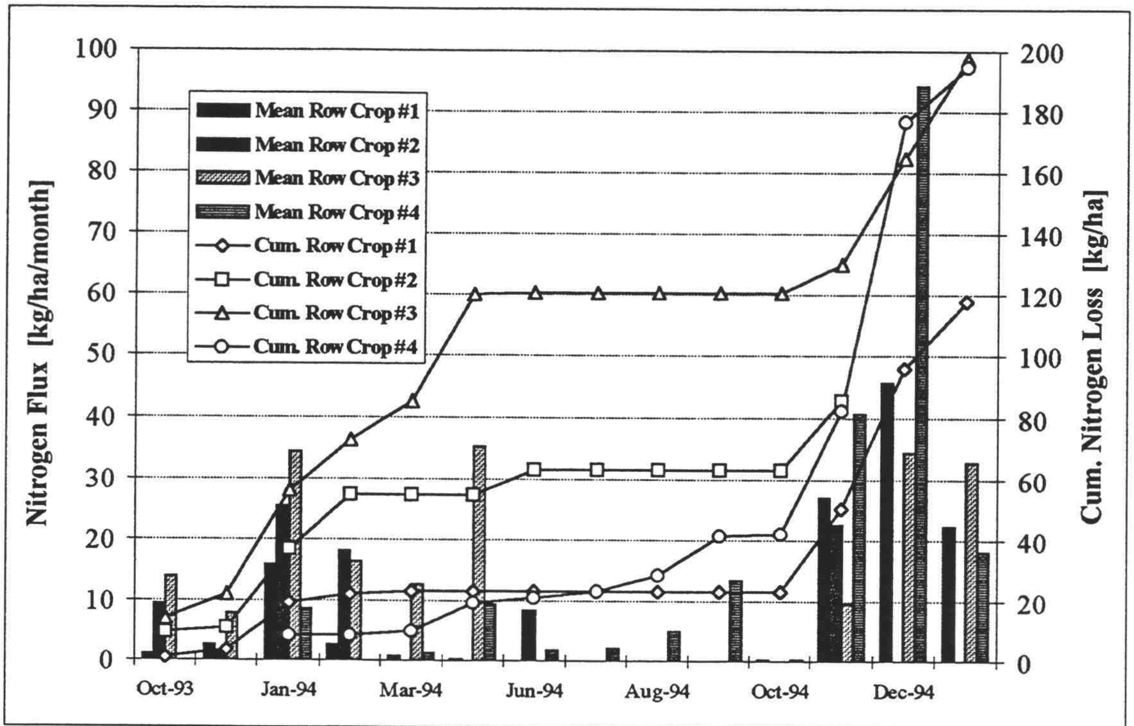

Figure 30. Monthly and cumulative mass losses of nitrogen as measured by the PCAPS from the row crop management systems.

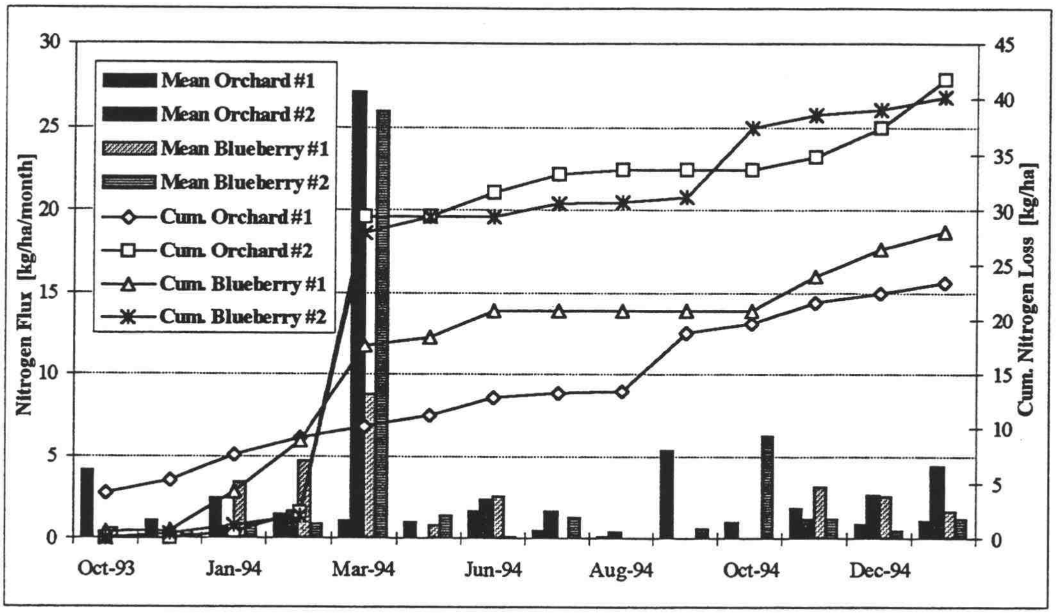

Figure 31. Monthly and cumulative mass losses of nitrogen as measured by the PCAPS from the orchard and blueberry management systems.

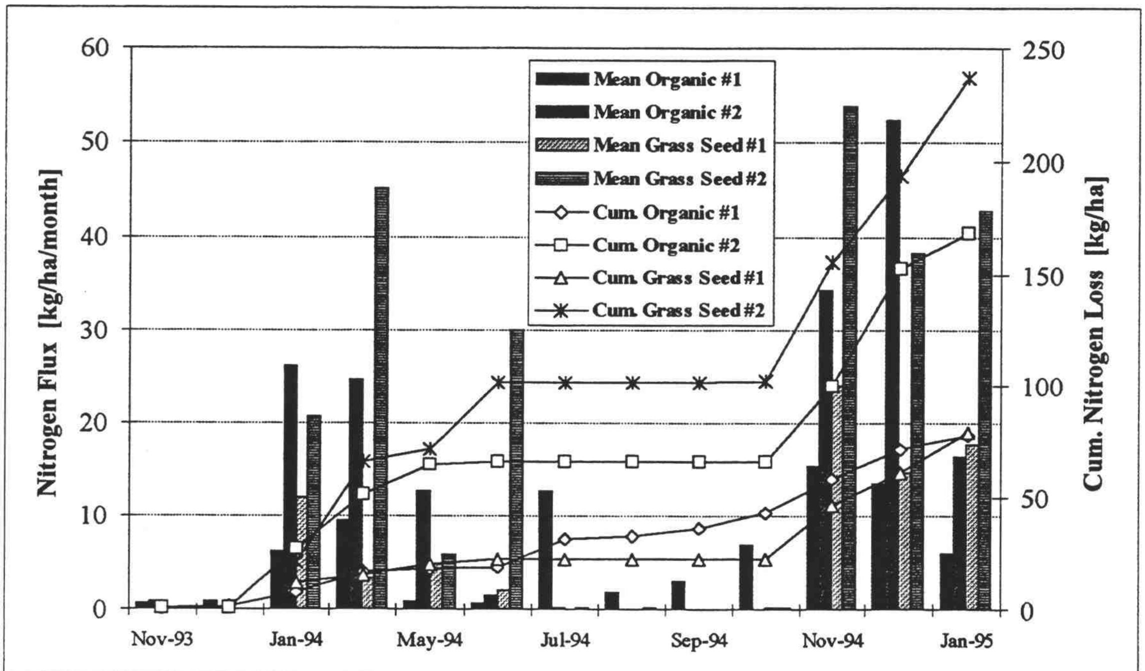

Figure 32. Monthly and cumulative mass losses of nitrogen as measured by the PCAPS from the organic and grass seed management systems.

Row crop producers lost the second highest amounts of nitrogen during the study period following the mint producers. As was the case for mint, high NO_3-N recharge concentrations leads directly to increased losses of nitrogen. As noted above, the mint producers lost a majority of their nitrogen during the growing season while the row crop producers lost a majority of their nitrogen during the Fall/Winter flush. These contrasts are a direct result of the volumes of percolate leaching below the root zone. On mint, urea in the liquid form (solution 32) is added to the irrigation water for much of the growing season. It appears that the urea leaches with the percolate resulting in high NO_3-N recharge concentrations (Figure 24) and large losses of nitrogen during the growing season. On the other hand, row crop producers are not required to irrigate their crops as intensely during the growing season. Thus, any excess nitrogen that was applied typically remains in the upper profile with little opportunity to leach under controlled irrigation. However, once the winter rains begin (Figure 22), higher NO_3-N concentrations are reflected as the pool of excess nutrients is now leached with ample volumes of percolate (Figure 25). The result of the winter flush is reflected in Figure 30.

The organic and grass seed producers appear to have the same pattern of nitrogen loss as the conventional row crop producers. Grass Seed #2 lost a significantly higher amount of nitrogen than Grass Seed #1 (Figure 32). This was attributed to the previous crop grown on this field which was mint. The soil at this site is fine-textured and appears to be retaining most of the nutrients left from the previous cropping system. It is anticipated that there will be a drop in NO_3-N concentrations as the residual N in the profile is depleted, although to date this trend has not occurred. The high loss of nitrogen from the Organic #2 site is a direct result of the cropping system used during the growing season. The field was left fallow during the growing season thus allowing the build-up of large quantities of nutrients. Once the winter rains occurred, approximately 150 kg ha^{-1} of nitrogen was lost from this site as a consequence of leaving the field fallow (Figure 32). Results of nitrogen lost from the Organic #1 and Grass Seed #1 sites appear to reflect the performance of the proper management of these systems. The contrast within the same production systems demonstrates the fact that any production system can have a negative environmental impact if wise management is not employed.

As anticipated, the orchard and blueberry production systems lost the least amounts of nitrogen of all the study sites. Low rates of fertilizer application lead directly to low losses of nutrients. Only during one period of the year (March-94) was there substantially high losses of nitrogen. The explanation for this high loss of nutrients can not be made without proper investigation of the yearly nutrient cycles at these sites.

Seasonal Effects on Nitrogen Loss

There is evidence to suggest that times of the year have a significant effect on the amounts of nitrogen lost to the subsurface (Table 23). Fall/Winter nitrogen losses were compared to Spring/Summer nitrogen losses for the Jan-94 to Jan-95 study period using a paired difference t-test. Of the 15 study sites, only five were found to have non-significant differences between seasonal losses of nitrogen. All row crop operations had significantly higher losses of nitrogen during the Fall/Winter season as compared to the Spring/Summer season. This was as expected from observations of the seasonal percolation and recharge NO_3-N concentrations. For the mint production systems, there was quite a contrast in the nitrogen loss results. Mint #1 was found to have significantly higher losses of nitrogen during the Fall and Winter while Mint #4 had significantly higher losses of nitrogen during the Spring and Summer. As previously mentioned, percolation amounts for Mint #1 were taken from those recorded at the Mint #3 site. The downfall of this substitution is reflected in the seasonal differences. Because NO_3-N concentrations at Mint #1 are comparable to those at Mint #4 (Figure 24), the possibility that the amount of percolate during the growing season being as high at Mint #1 as at Mint #4 may be significant. If this were the case, differences in the seasonal losses of nitrogen at Mint #1 would most likely shift and be very similar to losses recorded at the Mint #4 site. No seasonal differences in nitrogen losses were recorded at the Mint #3 site. The management of this site appears to be at an environmentally sound level which is reflected in the annual recharge concentrations and seasonal losses of nitrogen.

Table 23. Comparison of seasonal nitrogen losses from all study sites.

Site	N rate	Losses of Nitrogen Fall/Winter	Spring/Summer	Difference	t-test
	kg ha^{-1}		kg N/ha		
Blueberry #1	102	24.1	3.31	-20.7	**
Blueberry #2	76	30.3	9.49	-20.9	NS
Grass Seed #1	156	71.8	7.15	-64.7	**
Grass Seed #2	156	201	36.0	-165	**
Mint #1	250	113	23.0	-89.6	*
Mint #3	280	40.8	56.6	15.8	NS
Mint #4	370	119	248	129	*
Orchard #1	0	8.83	9.47	0.64	NS
Orchard #2	45	37.5	4.29	-33.2	NS
Organic #1	180	51.1	25.8	-25.3	NS
Organic #2	180	154	14.3	-140	*
Row Crop #1	200	114	0.40	-114	**
Row Crop #2	170	77.0	8.66	-68.4	*
Row Crop #3	135	140	35.4	-105	**
Row Crop #4	180	163	32.1	-131	*

*, ** Significant difference at P = 0.05 and 0.01 respectively. NS = not significant.

Losses of nitrogen were found to differ significantly seasonally at Organic #2, Blueberry #1, and both grass seed sites,. As previously mentioned, the differences in losses of nitrogen at the Grass Seed #2 site is most likely so pronounced due to the previous crop type, mint. However, the differences are consistent with the difference in nitrogen loss at the Grass Seed #1 site. The differences in seasonal losses of nitrogen at Organic #2 are also indicative of the crop type at this site. Because no crop was planted during the growing season, no irrigation was required. Thus, any pool of nutrients building-up during this period were leached when percolation below the root zone occurred. High losses of nitrogen could have been avoided with the planting of a crop. As expected, the orchard and blueberry management systems exhibited no differences in seasonal losses of nitrogen except for Blueberry #1. The significant difference at this site can be explained by the amount of percolate sampled during the growing season. Soil solution did not reach the PCAPS at this site between the months of August and October. Thus, no nitrogen was lost, resulting in a significant difference between seasonal nitrogen loss.

Comparison of PCAPS Nitrogen Loss and Mass Balance Estimated Nitrogen Loss

A comparison between the loss of nitrogen from each site as recorded by the PCAPS and mass balance estimated loss of nitrogen was undertaken (Table 24). This analysis was done to determine if 1) the PCAPS are accurately estimating the mass loss of N to some degree; and 2) the farmers are well aware of there fertilizer application amounts. Several difficulties were associated with estimating the mass balance loss of nitrogen. First, precise estimates of each farmers nitrogen application tended to be a problem when fertilizers were applied through the irrigation water. In most cases, the farmers are unaware of the actual amounts of fertilizer being applied through the irrigation system, or some farmers may have neglected to document the fertilizer being applied with the irrigation. Second, precise measurements of soil biological activity was not undertaken. Third, the mint and organic producers were eliminated from this analysis due to lack of critical data. Mint harvest is recorded in the amount of oil harvested from the mint. From all sources which list nitrogen content of harvested product, none showed data for mint oil. We could find no data on this crop as to the amounts of nitrogen harvested in relationship to the amount of harvested oil product. Organic farmers were omitted because little information is known about the nutrient contents of the composts and manures applied to the fields. Assumptions as to the amounts of nitrogen being leached were as follows: 1) the pool of organic N in the soil was constant from year to year so that net mineralization or immobilization was zero; 2) the rate of movement of NO_3^- below the root zone was equal to the movement of water; and 3) the amount of denitrified or leached N was equal to the total N input minus the sum of removal in crop (Fried et al., 1976). These assumptions are biased, but the results may provide the best check on the nitrogen losses as recorded by the PCAPS.

On the average the PCAPS under-estimated the loss of nitrogen by 12% with a range of -263 to 100% (Table 24). For the 10 sites used in the analysis, the PCAPS did a good job of estimating the mass loss of nitrogen ($R^2 = 0.58$). This is reassuring for the nitrogen loss estimates on the organic and mint production sites. The fact that the PCAPS do under-estimate the nitrogen loss on the average is most likely linked to the fact that the PCAPS tend to under-sample the actual ground water recharge. However, the recharge estimates obtained through using PCAPS may be more accurate than mass balance estimates in the long run. The fact that the PCAPS are placed directly below the root zone and have proven to estimate flux and matrix concentrations suggest that quantities which the PCAPS sample may be actual quantities leaching to the subsurface.

The importance of the mass loss of nitrogen estimates is demonstrated in Figure 33. The loss of nitrogen from a management system is a loss of production to the crop and an economic loss to the farmer. Figure 33 is a histogram of the number of farms and which farms lost what range of money with respect to the loss of nitrogen through leaching based on a nitrogen cost of 30¢/lb. Possibly, given information on economic losses as well as environmental impacts, the farmers may understand the results of lack of proper management procedures.

Table 24. Comparison between PCAPS estimated and mass balance estimated losses of nitrogen from each study site.

Site	N rate	Harvested Product	Harvested N	Nitrogen Loss: Mass Balance	Nitrogen Loss: PCAPS
	[kg ha^{-1}]				
Blueberry #1	102	3360	4	98	27
Blueberry #2	76	8100	10	66	40
Grass Seed #1	156	1850	30	126	79
Grass Seed #2	156	2000	31	125	237
Orchard #1	0	50000	32	0	18
Orchard #2	45	4500	6	39	42
Row Crop #1	200	19000	82	118	96
Row Crop #2	170	50000	130	40	86
Row Crop #3	135	2700	12	123	175
Row Crop #4	180	22400	96	84	195

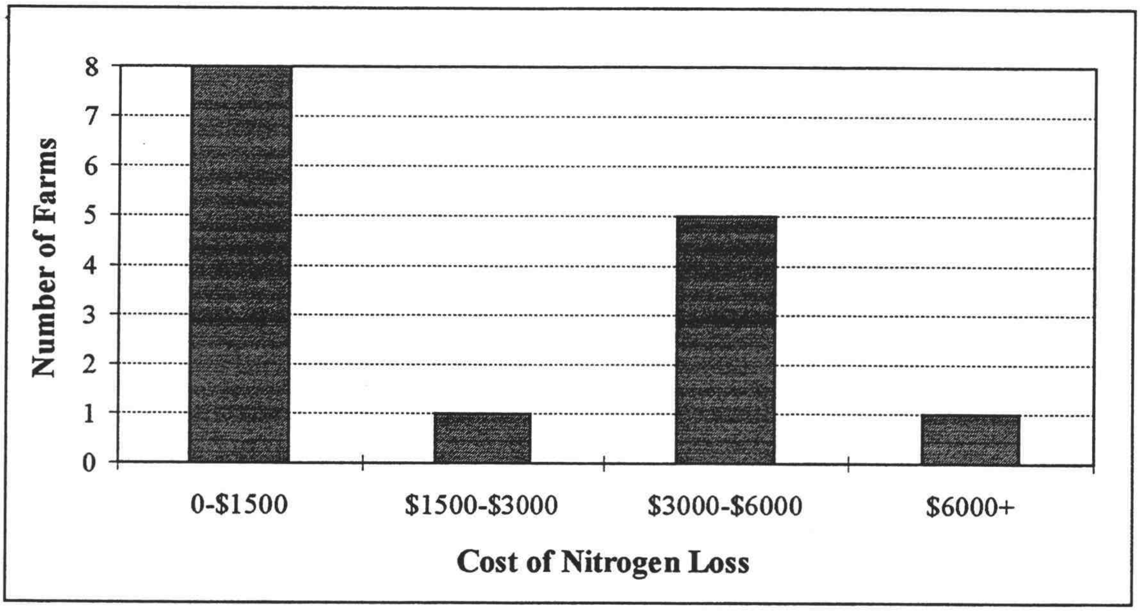

Figure 33. Estimation of the number of farms and the range of costs related to the loss of nitrogen by leaching (based on nitrogen costs of 30 cents/lb on a 100 acre farm).

Best Management Practice Suggestions

The next phase of the Lane County project deals with instituting best management practice (BMP) strategies for reducing the environmental impact of the row crop and mint production systems. Some suggestions as to the types of management strategies which would most benefit these sites were considered. For the most part, the main goal is to reduce the potential for loss of N through management strategies without establishing any undesirable quotas or regulations. If the suggestions are employed by the farmers, the potential for reductions in loss of nutrients is good. Tillage, fertilization, use of cover crops, soil and plant testing, and irrigation are all practices which can be manipulated by farmers to great advantage to minimize the possibility of nitrate leaching into the ground water. This section will focus on BMP suggestions for row crop and mint producers which may allow them to decrease the environmental impact of their management system.

The suggestions deal with three main areas: 1) irrigation, 2) fertilization, and 3) cover cropping.

The goal in application of nitrogen fertilizers is to apply a sufficient quantity to satisfy crop needs above that which the soil cannot provide. Yield is the main variable of concern when considering nitrogen applications. As shown, when calculating mass losses of nitrogen from a cropping system, the major component which removes nitrogen from the soil/plant system is the harvested product. Agricultural extension services and literature information can provide adequate knowledge and support for those needing information on crop yields. Yield goals should be based upon the previous crop history of the farmer. Typically, the farmer is more aware of the capability of a crop on their lands and can provide themselves with accurate estimates of a "good" years harvested product. If previous cropping history is not available, Wiese et al. (1987) suggest using average production over a 5 year period, and add no more than 5% to that average. The Soil Conservation Service county soil surveys also provide yield estimates for popular crop types on all soils found within the county. These estimates are based upon information acquired from experiment stations and extension services.

Combining good water management with proper N management can successfully reduce the NO_3 leaching potential. Generally, good water management is much the same as good N management. The water management objective is to control soil water to produce the highest yield while minimizing leaching. Proper irrigation management may require technical support for farmers not familiar with keeping track of the soil water balances. Good irrigation practices includes knowledge about the soil-water holding capacity and allowable level of water depletion prior to irrigation. The acceptable soil moisture range is defined by the available soil water concept. Field capacity, water content at which drainage is negligible, is considered the maximum acceptable water content for irrigation. The minimum acceptable soil water content is the point at which plants begin to permanently wilt known as the wilting point. Thus, the simplest method of monitoring irrigation requirements relies on monitoring the soil water content. Table 25 gives estimates for the available water capacity on 10 common soil types. The knowledge of a soil water balance (Equation 16) is the best means for farmers to accomplish this strategy. ET is the most important variable for the farmers. The rate of upward water flow (or the depletion of available soil water) depends upon the type of soil and the depth to the water table. Experiment stations such as Hyslop in Corvallis monitors daily ET or ET can be calculated from evaporation pans located in the area. This information can be made available to farmers. In addition, easy to use TDR equipment can be installed to allow farmers to monitor their soil water content throughout the week. Mint farmers must realize that irrigation applications does not mean saturating the soil. The soil should not be over-burdened with water such that leaching occurs. Although it is convenient to turn on irrigation systems and leave them running on timers, water is being applied in excess of that which is required. Employing the concept of irrigating only the available water capacity and only when this has been depleted by 50-60% through ET, mint producers should be able to reduce their economics of irrigation as well as their leaching losses.

Cover cropping is an important concept which should be considered by all row crop producers using a rotation system and may even have a role to play in mint production. Cover crops are important with respect to potential NO_3-N leaching because they use residual or mineralized NO_3^- in soils during non-crop periods. Growing cover crops also utilizes soil water allowing the soil to dry, reducing the amount of water moving through the soil profile and thus decreasing the potential for NO_3-N leaching. Non-leguminous cover crop uptake ranges from 12 to 117 kg N ha^{-1} yr^{-1}, but in most cases from 25 to 45 kg N ha^{-1} yr^{-1}. Brandi-Dohrn et al. (1994) were the first to test the impact of a cereal rye cover crop upon ground water recharge under soils in the humid Willamette Valley of Oregon. They found the cover crop was able to reduce leaching by increasing evapotranspiration on an average of 12%. In addition, the cover crop was able to decrease the mean seasonal recharge NO_3-N concentration by 40% and reduce total mass losses by 16.5 kg N ha^{-1} yr^{-1}.

Table 25. Representative values for soil water capacity of several soil types (adapted from Martin et al., 1991).

Soil Texture	Available water capacity, in. of water/ft. of soil	Volumetric Water Content	
		Field Capacity	Permanent Wilting Point
Coarse Sand	0.6	0.10	0.05
Sand	1.0	0.15	0.07
Loamy sand	1.3	0.18	0.07
Sandy loam	1.5	0.20	0.08
Loam	1.8	0.25	0.10
Silt loam	2.2	0.30	0.12
Silty clay loam	2.0	0.38	0.22
Clay loam	1.8	0.40	0.25
Silty clay	1.6	0.40	0.27
Clay	1.4	0.40	0.28

Cover crops are typically plowed under in the spring, thus increasing the pool of organic matter. More nitrogen is available for mineralization leading to substantial losses of nitrogen during this period if fertilizer applications are not adjusted. For this reason, soil testing for nitrogen should be done in conjunction with using cover crops. If the soil nitrogen content is known, fertilizer applications can be adjusted accordingly to avoid similar leaching patterns under fields having cover crops. It is apparent that a nonleguminous cover crop can decrease the potential for NO_3^- leaching by utilizing residual soil nitrogen during the winter period and by reducing soil water content during its active growth stage.

The most important aspect for all participants willing to work towards reducing the leaching of nitrogen is providing the technical support needed by the farmers. In many cases, research fails to be provided to those who most need the knowledge. Informing and teaching the farmers about wise management of resources requires their knowledge of their system and researchers knowledge of important soil and water processes. Without these two participants working together, the wide-scale problem cannot be solved. Instead of existing as separate entities, the cooperation between groups can provide positive responses where they are most needed. Information on irrigation and fertilizer management and cover cropping are examples of areas which have received much technical focus but may fail to be shared with those who most need the information.

Conclusions

PCAPS can provide information on both matrix and flux concentrations leaching under agricultural production. The ability of the PCAPS to sample ground water recharge can allow for the determination of those management systems which contribute to the largest adverse environmental impacts. Soil variability is unavoidable requiring 7 or more PCAPS per site to obtain accurate measurements of recharge. However, the PCAPS performance was superior to that of suction cup samplers because of their ability to sample flux. Without data on flux, true annual NO_3-N concentrations cannot be calculated. Recharge volumes are also important for estimating mass losses of nitrogen from each management system. The mint production system was found to have significantly higher amounts of percolate due to over-irrigation of the crop during the growing season. As a result, the mint systems lost a majority of their nitrogen during the growing season. NO_3-N recharge concentrations were highest for mint during the growing season while for other crop systems high concentrations were observed during the fall and winter. Mint and row crop sites were found to have the largest adverse environmental impacts of the practices monitored. NO_3-N recharge concentrations under mint and row crop management systems averaged 24 mg L^{-1} and 28 mg L^{-1}, respectively. Orchard and blueberry systems were determined to have little environmental impact with their seasonal NO_3-N concentrations averaging 6 and 4 ppm respectively which is below the EPA water quality standard. Nitrate concentrations in recharge were positively correlated with well nitrate concentrations at all study sites. This result suggests the need for immediate procedures to reduce the impact of high losses of nitrogen from mint and row crop systems.

Seasonal effects were significant for mass losses of nitrogen under the mint and row crop production systems. The effects of the season varied for the mint system. Typically, more nitrogen will be lost during the growing season under mint due to high levels of irrigation and nitrogen application with irrigation water. However, Mint #3 was able to control both irrigation and nitrogen sufficiently to have no difference in nitrogen loss between the winter and summer seasons. For non-organic crop production systems, there was significantly higher losses of nitrogen during the winter season than the growing season. The flush period characterized by high losses of nitrogen due to high residual soil nitrogen and large percolate volumes is extremely important for Oregon farmers. Although irrigation was controlled during the summer to reduce the loss of nutrients to the crop, excess nutrients were leached once ample quantities of percolate were provided.

The PCAPS estimate of the mass loss of nitrogen from the study sites was similar to that obtained using a simplified nitrogen mass balance. On the average, the PCAPS estimated the mass loss of nitrogen to be 12% less than the mass balance. The fact that the mass loss estimates are accurate for the study sites points to the need for technical assistance for the farmers. Not only are the farmers creating adverse environmental conditions, but they are using their resources inefficiently. This inefficiency results in losses of profits for the farmers, who typically lose mare than $1500/yr in nitrogen. The economics of the problem are key to convincing farmers to alter their management practices.

Three areas of focus were presented to assist farmers in developing new management schemes to reduce the impact of their production systems. These BMP suggestions were made with respect to observations made under row crop and mint production systems. Increased fertilizer and water use efficiency go hand in hand. Crop yields are the ultimate goal of proper fertilizer and irrigation application. Fertilizer should only be applied in the amounts based upon what quantities of nitrogen will be removed in the harvested product. If most nitrogen is removed during the harvest, residual soil nitrogen levels will be low thus reducing the chance for high NO_3-N recharge concentrations. Irrigation should only be applied in the amounts that are necessary to sustain the crop. This level is called the available soil water capacity. Applying larger amounts of water than necessary will cause drainage to the subsurface. Fields need not be saturated to supply the needs of the crop. For row crop producers, the use of cover crops during the flush period can significantly reduce residual nitrogen levels and drainage volumes. The use of these concepts in conjunction with technical support from researchers can provide immediate decreases in both the loss of nutrients and elevated NO_3-N concentrations.

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APPENDICES

Appendix A: Soil Properties

Soil	Depth	Particle Size Distribution			Bulk Density		K_{sat}	
		Clay	Silt	Sand	Mean	n	Mean	n
	[cm]				[g cm^{-3}]		[cm day^{-1}]	
Awbrig silty clay loam	17	30	55	15	---	---		
	70	53	27	20	1.49	3	19	3
	114	55	20	25	---	---		
Chehalis silty clay loam	16	35	50	15	---	---		
	86	30	55	15	1.36	3	72	3
	120	20	30	40	---	---		
Cloquato silt loam	17	5	75	20	---	---		
	65	10	65	25	1.49	3	46	3
	115	3	17	70	---	---		
Coburg silty clay loam	20	38	20	42	---	---		
	62	53	17	30	1.49	3	8	3
	118	20	30	50	---	---		
Fluvents	15	2	12	41				
(Dredging soils, significant gravel content)	68	3	15	40	THIS IS FILL MATERIAL WHICH DOES NOT HAVE CONSISTENT HYDRAULIC PROPERTIES			
Malabon silty clay loam	15	33	42	25	---	---		
	67	37	43	20	1.46	3	46	3
	118	11	29	60	---	---		
Newberg fine sandy loam	18	10	12	78	---	---		
	46	5	34	61	1.42	3	57	4
	110	10	24	66	---	---		
Newberg loam	18	15	40	45	---	---		
	65	10	19	71	1.32	3	51	3
	108	13	6	81	---	---		

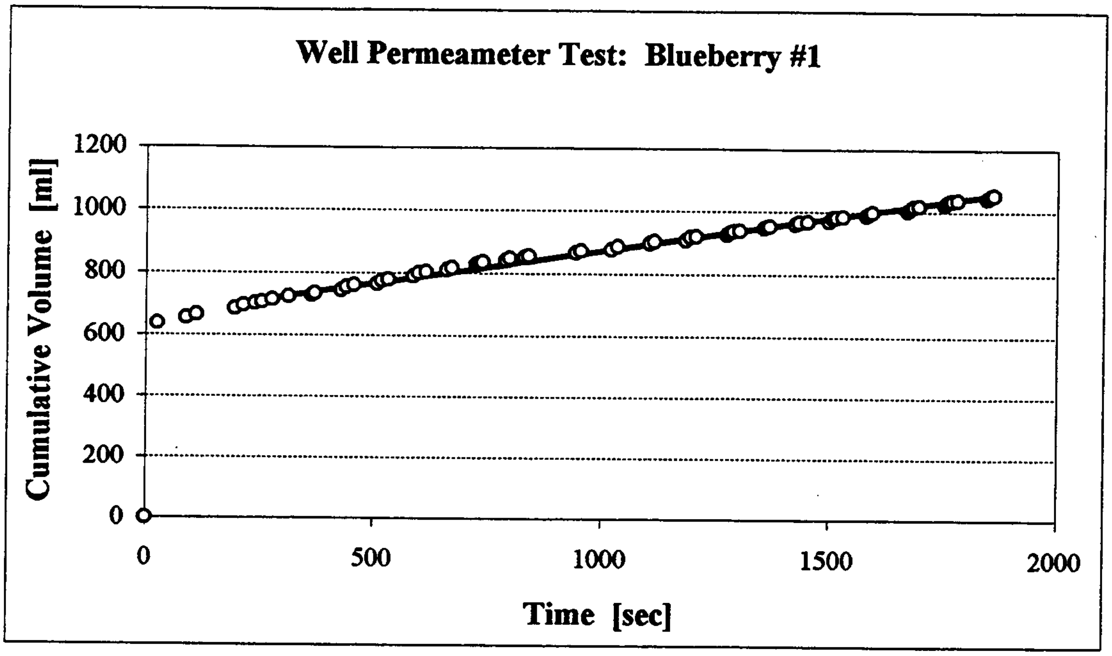
		Water Retention					
		Experimental (Klute)		Empirical (Arya & Paris)		Empirical (RETC)	
Soil	Depth	Pressure	Volumetric Water Content	Pressure	Volumetric Water Content	Pressure	Volumetric Water Content
	[cm]	[kPa]	[cm3 cm-3]	[kPa]	[cm3 cm-3]	[kPa]	[cm3 cm-3]
Awbrig silty clay loam	70	0	0.47	0	0.48	0	0.45
		1.45E+00	0.44	1.51E+00	0.32	4.83E+00	0.43
		4.83E+00	0.42	1.18E+01	0.32	7.25E+00	0.41
		9.66E+00	0.41	2.21E+01	0.31	2.01E+01	0.34
		2.90E+01	0.29	3.43E+01	0.29	3.18E+01	0.29
		7.73E+01	0.21	4.65E+01	0.28	5.22E+01	0.24
				8.76E+03	0.24	7.00E+01	0.21
				3.94E+05	0.18	1.01E+02	0.19
				1.13E+07	0.03	2.03E+03	0.11
Chehalis silty clay loam	86	0	0.52	0	0.52	0	0.52
		1.45E+00	0.45	1.46E+00	0.34	1.93E+00	0.44
		4.83E+00	0.40	1.01E+01	0.33	4.54E+00	0.40
		9.66E+00	0.36	1.88E+01	0.32	1.16E+01	0.36
		2.90E+01	0.33	3.39E+01	0.31	3.09E+01	0.32
		7.73E+01	0.30	4.90E+01	0.31	9.37E+01	0.28
				9.74E+03	0.26	3.38E+02	0.24
				3.42E+05	0.12	1.01E+05	0.12
				1.10E+07	0.03	2.90E+06	0.08
Cloquato silt loam	65	0	0.48	0	0.48	0	0.45
		1.45E+00	0.43	1.51E+00	0.32	2.42E+00	0.42
		4.83E+00	0.39	1.04E+01	0.32	5.02E+00	0.38
		9.66E+00	0.33	1.93E+01	0.31	9.66E+00	0.34
		2.90E+01	0.3	3.29E+01	0.30	1.93E+01	0.31
		7.73E+01	0.27	4.65E+01	0.29	5.80E+01	0.27
				1.12E+04	0.26	1.16E+02	0.25
				2.94E+05	0.03	1.74E+03	0.23
Coburg silty clay loam	62	0	0.50	0	0.50	0	0.46
		1.45E+00	0.41	2.75E-02	0.32	9.66E-01	0.42
		4.83E+00	0.35	1.34E-01	0.31	5.60E+00	0.33
		9.66E+00	0.29	1.72E+00	0.28	1.45E+01	0.27
		2.90E+01	0.24	2.21E+01	0.24	4.64E+01	0.21
		7.73E+01	0.20	5.04E+01	0.21	9.66E+01	0.18
				9.48E+03	0.16	9.66E+02	0.12
				3.37E+05	0.06	9.66E+04	0.08

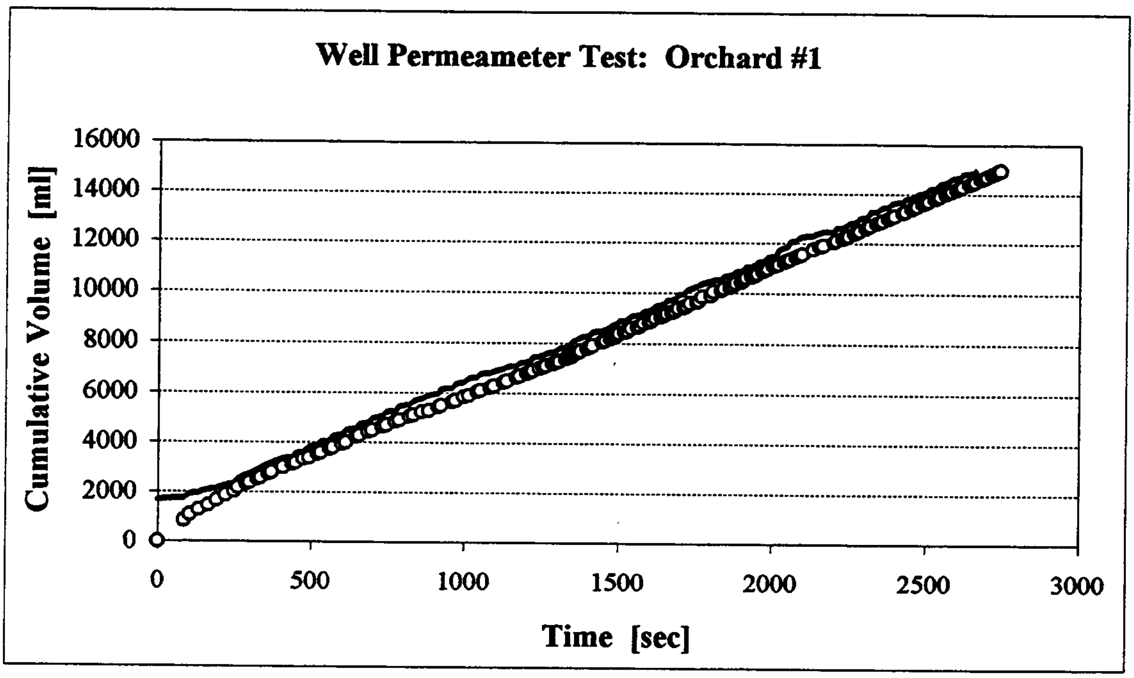
		Water Retention					
		Experimental (Klute)		Empirical (Arya & Paris)		Empirical (RETC)	
Soil	Depth	Pressure	Volumetric Water Content	Pressure	Volumetric Water Content	Pressure	Volumetric Water Content
	[cm]	[kPa]	[cm3 cm-3]	[kPa]	[cm3 cm-3]	[kPa]	[cm3 cm-3]
Malabon silty clay loam	67	0	0.48	0	0.48	0	0.47
		1.45E+00	0.45	2.67E-02	0.35	1.35E+00	0.45
		4.83E+00	0.38	1.14E-01	0.33	3.38E+00	0.40
		9.66E+00	0.32	1.46E+00	0.31	5.12E+00	0.37
		2.90E+01	0.29	1.87E+01	0.29	1.01E+01	0.32
		7.73E+01	0.28	3.95E+01	0.28	2.12E+01	0.29
				9.73E+03	0.26	4.35E+01	0.28
				3.65E+05	0.12	1.74E+02	0.26
Newberg fine sandy loam	46	0	0.48	0	0.48	0	0.48
		1.45E+00	0.44	2.02E+00	0.34	2.90E+00	0.39
		4.83E+00	0.35	1.45E+01	0.30	4.83E+00	0.34
		9.66E+00	0.28	2.70E+01	0.27	8.70E+00	0.30
		2.90E+01	0.24	4.07E+01	0.23	1.64E+01	0.26
		7.73E+01	0.18	5.44E+01	0.18	3.48E+01	0.22
				9.34E+03	0.12	8.70E+01	0.17
				3.01E+05	0.03	1.45E+03	0.09
Newberg loam	65	0	0.49	0	0.49	0	0.45
		1.45E+00	0.44	1.74E+00	0.37	2.42E+00	0.43
		4.83E+00	0.42	1.42E+01	0.35	9.66E+00	0.38
		9.66E+00	0.37	2.67E+01	0.33	2.61E+01	0.31
		2.90E+01	0.31	3.69E+01	0.29	4.15E+01	0.28
		7.73E+01	0.24	4.71E+01	0.24	6.86E+01	0.24
				1.01E+04	0.20	1.19E+02	0.21
				3.22E+05	0.06	4.83E+02	0.14

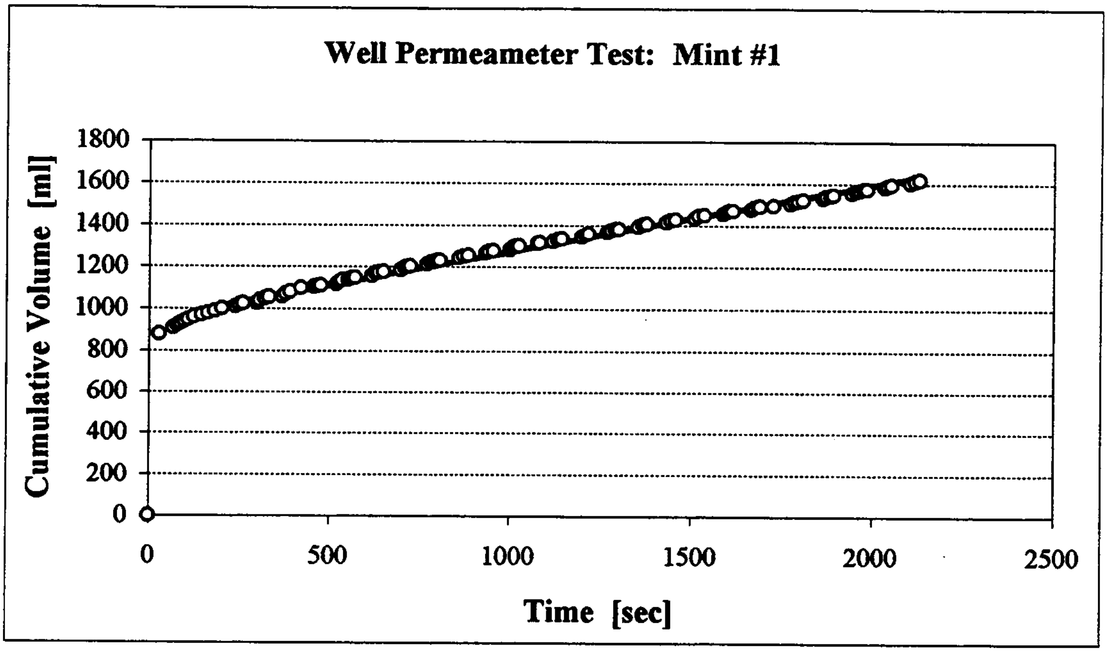
Appendix B: Fertilizer Applications

Farm	Fertilizer Type	Nitrogen Application Amount [kg ha^{-1}]	Type of Application	Time of Application
Blueberry #1	Ammonium sulfate - $(NH_4)_2SO_4$	34 (3 times/year)	banded	April, May and June
Blueberry #2	$(NH_4)_2SO_4$	38 (2 times/year)	banded	April and May
Grass Seed #1	Urea - $CO(NH_2)_2$ $(NH_4)_2SO_4$ Monoammonium phos. - $NH_4H_2PO_4$	22 in November 67 - 2 times/year	banded	March and April
Grass Seed #2	$CO(NH_2)_2$ $(NH_4)_2SO_4$ $NH_4H_2PO_4$	22 in November 67 - 2 times/year	banded	March and April
Mint #1	Urea - ammonium nitrate (Solution 32) $CO(NH_2)_2 + NH_4NO_3$	250	broadcast and thru irrigation	Growing Season
Mint #3	$CO(NH_2)_2$ $CO(NH_2)_2 + NH_4NO_3$ $(NH_4)_2SO_4$	280	broadcast and thru irrigation	Growing Season
Mint #4	$CO(NH_2)_2$ $CO(NH_2)_2 + NH_4NO_3$	370	broadcast and thru irrigation	Growing Season
Orchard #1	None	None	N/A	N/A
Orchard #2	Nitrogen	45	broadcast	Growing Season
Organic #1	leaf composts and manures	≈ 180	intermixed	Growing Season
Organic #2	leaf composts and manures	≈ 180	intermixed	Growing Season
Row Crop #1	Nitrogen - N	200	broadcast and banded	June
Row Crop #2	Nitrogen - N	170	banded and sidedressed	Spring or growing season
Row Crop #3	$CO(NH_2)_2$	135	broadcast	Spring and Fall
Row Crop #4	$CO(NH_2)_2$	180	broadcast	Spring

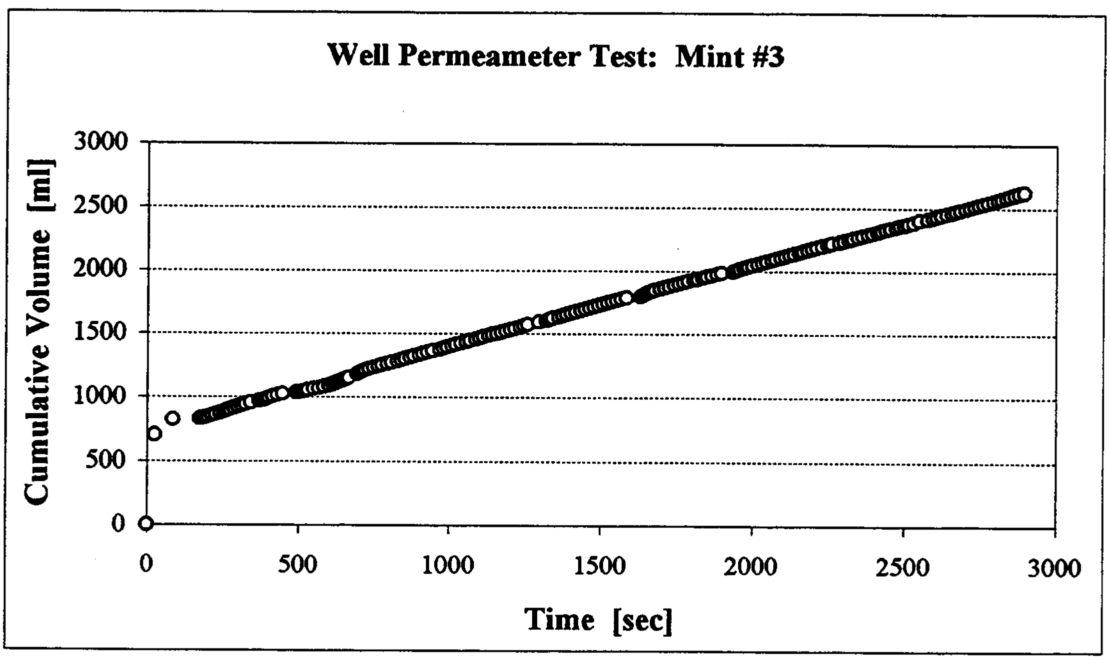
Appendix C: Field Saturated Conductivity Measurements


Well Permeameter Test: Blueberry #1
Cumulative Volume [ml]
1200
1000
800
600
400
200
0
0
500
1000
1500
2000
Time [sec]

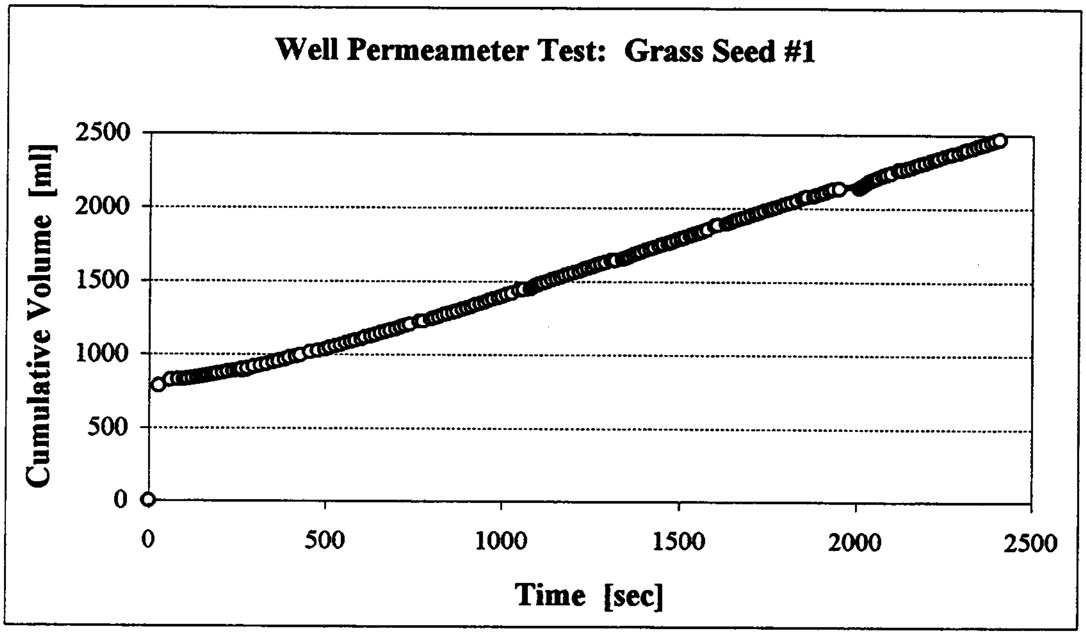

Well Permeameter Test: Orchard #1
Cumulative Volume [ml]
16000
14000
12000
10000
8000
6000
4000
2000
0
0
500
1000
1500
2000
2500
3000
Time [sec]



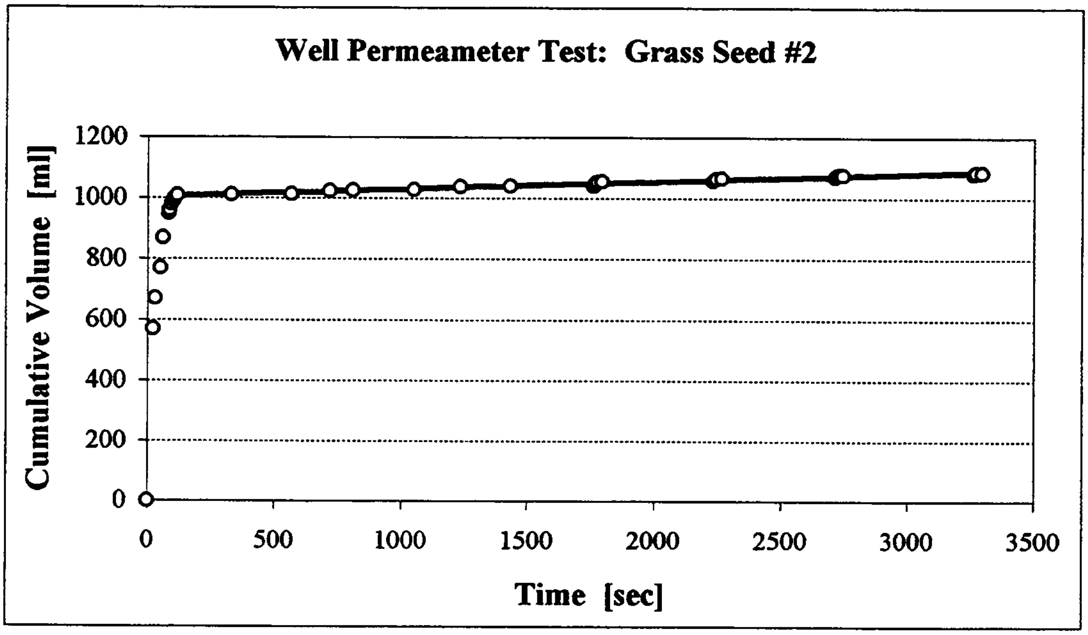
Well Permeameter Test: Mint #1
Cumulative Volume [ml]
0
200
400
600
800
1000
1200
1400
1600
1800
0
500
1000
1500
2000
2500
Time [sec]



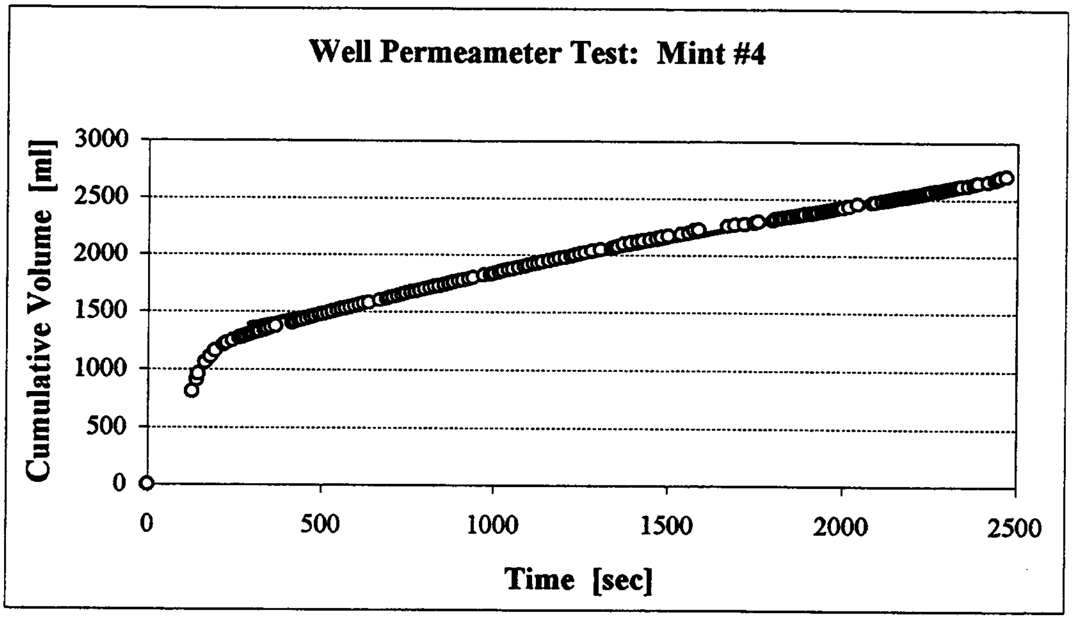
Well Permeameter Test: Mint #3
Cumulative Volume [ml]
0
500
1000
1500
2000
2500
3000
0
500
1000
1500
2000
2500
3000
Time [sec]

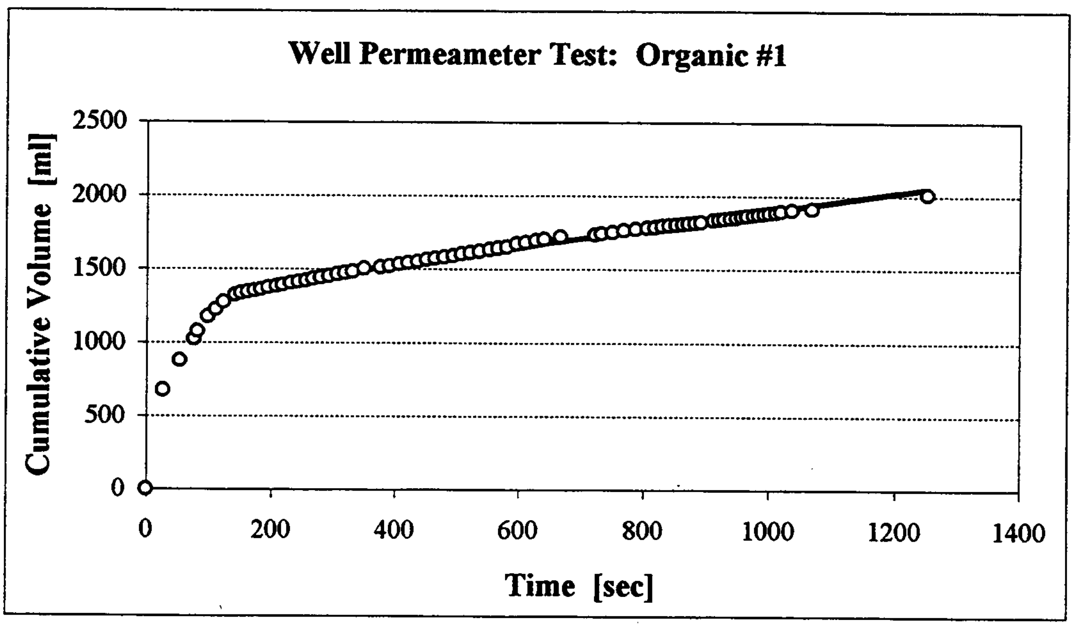


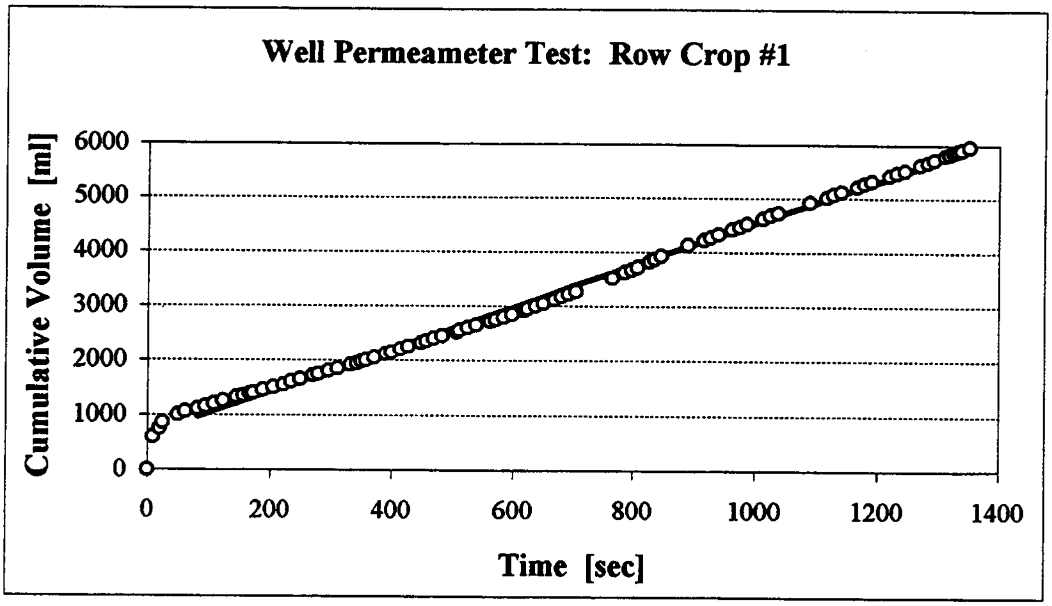
Well Permeameter Test: Grass Seed #1
Cumulative Volume [ml]
2500
2000
1500
1000
500
0
0
500
1000
1500
2000
2500
Time [sec]

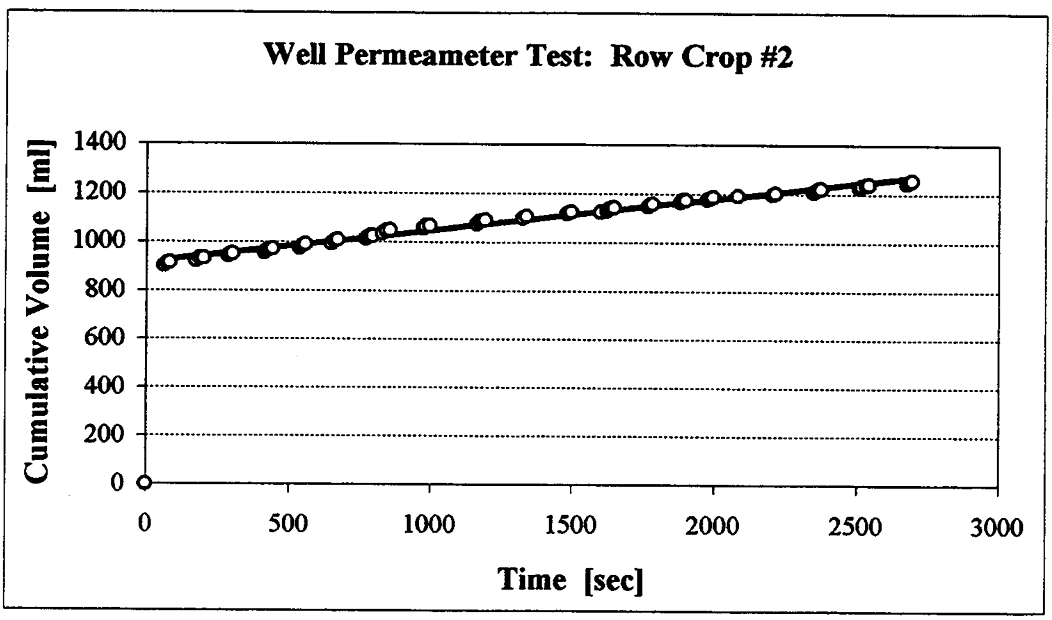


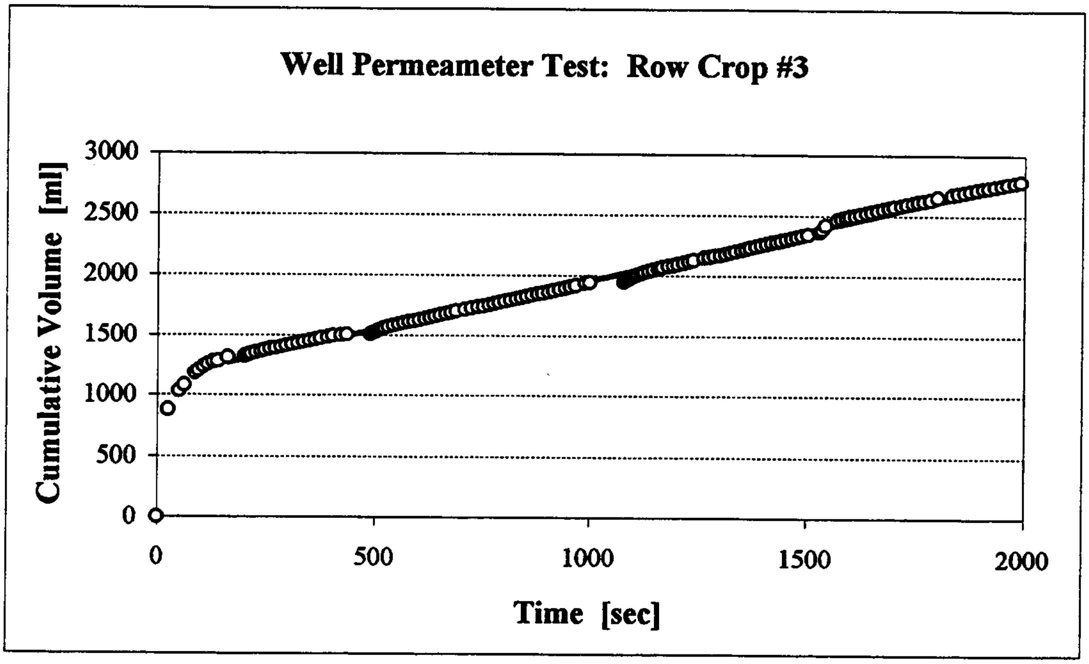
Well Permeameter Test: Grass Seed #2
Cumulative Volume [ml]
1200
1000
800
600
400
200
0
0
500
1000
1500
2000
2500
3000
3500
Time [sec]


Well Permeameter Test: Mint #4
Cumulative Volume [ml]
3000
2500
2000
1500
1000
500
0
0
500
1000
1500
2000
2500
Time [sec]


Well Permeameter Test: Organic #1
Cumulative Volume [ml]
2500
2000
1500
1000
500
0
0
200
400
600
800
1000
1200
1400
Time [sec]


Well Permeameter Test: Row Crop #1
Cumulative Volume [ml]
6000
5000
4000
3000
2000
1000
0
0
200
400
600
800
1000
1200
1400
Time [sec]


Well Permeameter Test: Row Crop #2
Cumulative Volume [ml]
1400
1200
1000
800
600
400
200
0
0
500
1000
1500
2000
2500
3000
Time [sec]


Well Permeameter Test: Row Crop #3
Cumulative Volume [ml]
3000
2500
2000
1500
1000
500
0
0
500
1000
1500
2000
Time [sec]

Appendix D: Monthly Recharge Data

Site	Mass Balance Components [cm]	Sample Collection Period												Annual Total
		Jan-94	Feb-94	Mar-94	May-94	Jun-94	Jul-94	Aug-94	Sep-94	Oct-94	Nov-94	Dec-94	Jan-95	
Blueberry #1	Precip.	10.3	13	13.3	20	10	10.8	5	8	7.7	16	23.9	18.6	156.6
	ET	0.8	1.5	3.3	8	8.4	10.5	10.5	10	6.7	1	1	1	62.7
	Storage loss	0	0	0	0	0	0	-5.5	-2.5	0	8.5	0	0	0.5
	Recharge	9.5	11.5	10	12	1.6	0.3	0	0.5	1	6.5	22.9	17.6	**93.4**
Grass Seed #1	Precip.	22	12	-	12	15	3	2	2.4	2	19	14.2	13.2	116.8
	ET	2	0.5	-	1	7	5	4	2	2	1	1	1	26.5
	Storage loss	0	0	0	0	0	-2	-2	0	0	4	0	0	0
	Recharge	20	11.5		11	8	0	0	0.4	0	14	13.2	12.2	**90.3**
Orchard #1	Precip.	16.5	12.6	14.6	7.3	11.9	5.6	5.9	13.2	6.9	21	16.7	22.2	154.4
	ET	1	2.7	4.2	1.9	8.9	10.6	11.9	7.6	2	1.2	1.2	0.9	54.1
	Storage loss	0	0	0	0	0	-5	-6	0	0	11	0	0	0
	Recharge	15.5	9.9	10.4	5.4	3	0	0	5.6	4.9	8.8	15.5	21.3	**100.3**
Organic #1	Precip.	7	12.5	-	25	10	14	14	10	8.8	18.1	15	19	153.4
	ET	1	0.5	-	9.2	12	11.5	13	8	2	0.8	0.6	0.5	59.1
	Storage loss	0	0	0	0	-2	0	0	0	0	5.5	0	0	3.5
	Recharge	6	12		15.8	0	2.5	1	2	6.8	11.8	14.4	18.5	**90.8**
Organic #2	Precip.	7	14	-	19	9.5	2	0	0	5.5	19	13	12.9	101.9
	ET	0.7	0.5	-	2	9	9	2	1	5	2	1.4	0.9	33.5
	Storage loss	0	0	0	0	0	-7	-2	-1	0	10.5	0	0	0.5
	Recharge	6.3	13.5		17	0.5	0	0	0	0.5	6.5	11.6	12	**67.9**
Mint #3	Precip.	5.4	22.5	-	20	12	12	10	11.5	10.2	13	16	18.9	151.5
	ET	0.4	2.5	-	9.5	7	7	8.4	4	2.7	0.6	0.6	0.4	43.1
	Storage loss	0	0	0	0	0	0	0	0	0	2	0	0	2
	Recharge	5	20		10.5	5	5	1.6	7.5	7.5	10.4	15.4	18.5	**106.4**
Mint #4	Precip.	21.6	10.8	11	22.5	15	8.2	13.1	11.2	10	20	13.8	13.4	170.6
	ET	0.6	0.8	1	5	6.8	6	5.1	5	1	0.5	0.5	0.4	32.7
	Storage loss	0	0	0	0	0	0	0	0	0	4.9	0	0	4.9
	Recharge	21	10	10	17.5	8.2	2.2	8	6.2	9	14.6	13.3	13	**133**
Row Crop #4	Precip.	14	15	-	20	10	8.7	10	10	5.8	17.3	16	20.6	147.4
	ET	0.5	2.5	-	1.3	7.4	10.7	14	8	5.8	1.5	1.5	0.9	54.1
	Storage loss	0	0	0	0	0	-2	-4	0	0	6	0	0	0
	Recharge	13.5	12.5		18.7	2.6	0	0	2	0	9.8	14.5	19.7	**93.3**

Site	Mass Balance Components [cm]	Sample Collection Period												Annual Total
		Jan-94	Feb-94	Mar-94	May-94	Jun-94	Jul-94	Aug-94	Sep-94	Oct-94	Nov-94	Dec-94	Jan-95	
Blueberry #2	Precip.	16.5	17.5	14.6	20	11.9	12	5.9	13.2	7.3	22.5	13.9	22.4	177.7
	ET	1	1.5	3	1.9	8.9	10	11	7.6	4.5	2	1.9	0.9	54.2
	Storage loss	0	0	0	0	0	0	-5.1	0	0	5.8	0	0	0.7
	Recharge	15.5	16	11.6	18.1	3	2	0	5.6	2.8	14.7	12	21.5	**122.8**
Grass Seed #2	Precip.	22	12	-	12	15	3	2	2.4	2	21	14.2	13.2	118.8
	ET	2	0.5	-	1	7	2.5	1.1	4.4	3	2	1	1	25.5
	Storage loss	0	0	0	0	0	0	0	-2	-1	3	0	0	0
	Recharge	20	11.5		11	8	0.5	0.9	0	0	16	13.2	12.2	**93.3**
Orchard #2	Precip.	5.4	22.5	21	25	12.5	10	9	5	16.4	14.5	18.8	22.2	182.3
	ET	0.4	2.5	3	4.2	5.8	9	11	7	2	2	0.8	0.9	48.6
	Storage loss	0	0	0	0	0	0	-2	-2	0	5.8	0	0	1.8
	Recharge	5	20	18	20.8	6.7	1	0	0	14.4	6.7	18	21.3	**131.9**
Row Crop #1	Precip.	16.5	12.6	14.6	7.3	11.9	5	5	5	2.4	23	13.6	21.5	138.4
	ET	1	2.7	3.8	8	8.9	5	5	7	4.4	3.5	2	0.9	52.2
	Storage loss	0	0	0	-0.7	0	0	0	-2	-2	5.5	0	0	0.7
	Recharge	15.5	9.9	10.8	0	3	0	0	0	0	14	11.6	20.6	**85.5**
Row Crop #2	Precip.	16.5	17.5	-	-	12.5	10	10	10	5	22.5	-	-	104
	ET	1	2.7	-	-	8	12	13	12	4.8	3	-	-	56.5
	Storage loss	0	0	-	-	0	-2	-3	-2	0	9	-	-	2
	Recharge	15.5	14.8	0	0	4.5	0	0	0	0.2	10.5	0	0	**45.5**
Row Crop #3	Precip.	8.7	16.5	11.5	27.5	8.7	0	0	0	5.8	17.3	12.6	22	130.6
	ET	0.9	0.9	1.3	7.4	11.7	2.5	0	0	7	1.5	0.5	0.4	34.1
	Storage loss	0	0	0	0	-3	-2.5	0	0	-1.2	7.2	0	0	0.5
	Recharge	7.8	15.6	10.2	20.1	0	0	0	0	0	8.6	12.1	21.6	**96**