A diagram of the soil infiltrator system is included as Figure 5. A french drain was installed down-gradient (downslope) and parallel to the laterals which drained through a drainpipe into the road ditch beside Route 11 North. The french drain was employed to shed excess surface and groundwater from the absorption area on the site.

B. SOIL SAMPLING

Soil samples were taken with a Giddings drilling rig. Particle size analysis (PSA) was performed on representative samples from each horizon within the profile from the site. The procedure was performed following the procedure of Day (1965). Hydraulic conductivities and bulk densities were also determined on samples from these profiles (Table 6) using the procedures by Klute (1965).

C. ROUTINE MONITORING EQUIPMENT

Monitoring equipment installed at the site included solution samplers, sampling wells, tensiometers, and soil access wells. A map showing all monitoring equipment as located at the site is included as Figure 6.

Soil solution samples were obtained during unsaturated soil conditions by using ceramic cup samplers or by PVC sampling wells if the soil was saturated. The solution samplers were constructed by fitting a 100 kPa high flow ceramic cup to a 5.1 cm diameter length of Fasco tubing. To install the sampler, a 6.3 cm hole was augered to the desired depth. A slurry of silica flour was poured into the hole so that the ceramic cup would be covered when the sampler was inserted. The flour ensured good contact with the surrounding soil. After the sampler was in place, soil was placed above the ceramic cup and filled to the original soil surface with a mixture of sand and bentonite. Bentonite was placed around the sampler at the surface. To obtain a sample, any residual fluid in

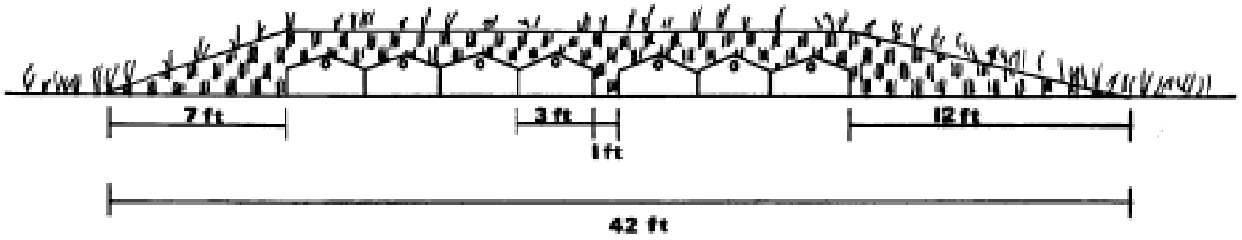

Figure 5. Cross-sectional sketch of installed soil infiltrator low-pressure distribution system.

Table 6. Particle size analyses, bulk densities, and hydraulic conductivities determined for the Blairton silt loam.

Horizon	Depth (cm)	Particle Size Analysis Sand	Silt	Clay	Bulk Density (g/cm^3)	Hydraulic Conductivity Rating (cm/hr)	Class
		%					
Subsystem 1 (design loading rate 5.1 Lpd/m^2)							
Ap	0-23	4	37	60	1.25	13.70	rapid
Bt1	23-58	ND	46	54	1.60	0.04	very slow
Bt2	58-84	ND	28	72	1.57	0.11	very slow
Bt3	84-107	ND	46	54	1.60	0.04	very slow
C	107-140	0	47	53	1.61	0.61	mod slow
Cr	140+	0	50	50	-----	-----	----------
Subsystem 2 (design loading rate 10.2 Lpd/m^2)							
Ap	0-17	20	62	18	1.43	25.60	very rapid
Bt1	17-33	11	60	29	1.47	7.92	mod rapid
Bt2	33-58	3	60	27	1.66	4.32	moderate
Bt3	58-74	3	60	37	1.69	0.94	mod slow
C	74-89	ND	56	44	1.55	3.60	moderate
Cr	89+	8	58	34	1.52	0.28	slow

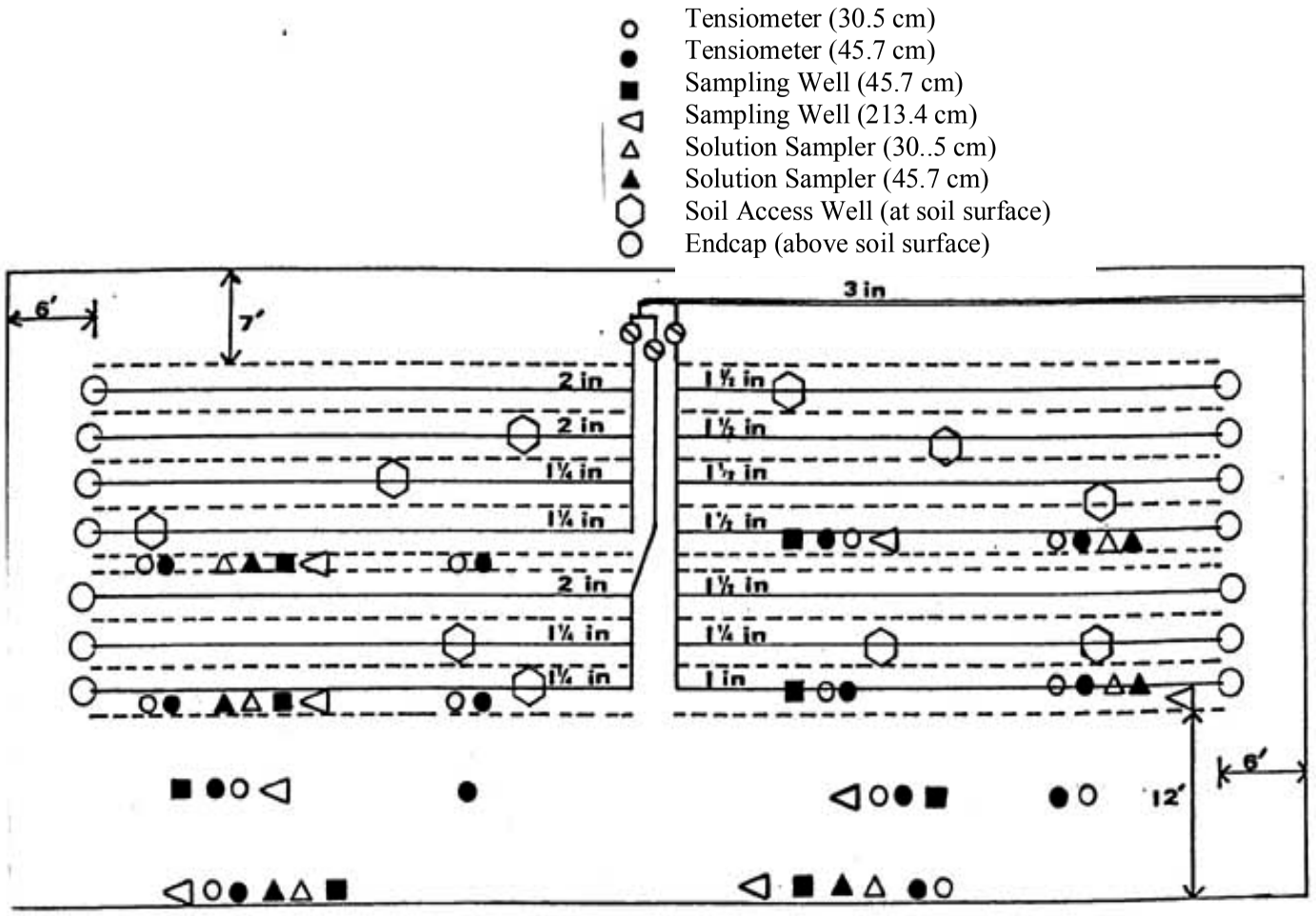

Figure 6. Layout of routine monitoring equipment installed within and surrounding the soil infiltrator low-pressure distribution system.

the sampler was discarded by placing a vacuum on the sampler; fresh solution could be extracted from the surrounding soil under unsaturated conditions where the matric tension in the soil did not exceed 100 kPa.

Although samples could be obtained from the solution samplers during saturated conditions, well samples were preferred as the ceramic cup filters the sample and may alter the composition of the soil solution to some degree depending on the component in question. Sampling wells were constructed of 3.8 to 5.1 cm Schedule 40 PVC pipe with 0.6 cm holes perforating two opposite sides of the lower 15.2 cm of the pipe. The holes were covered with fiberglass screening and a solid PVC cap was attached to the bottom to create a reservoir. A hole was augered to the desired depth and the well was placed in it. Coarse sand was added around the bottom of the sampler to cover the holes and 2.5 to 5.1 cm of soil was then packed around the well. Sand and bentonite was then filled around the well to the surface to seal the sampler. To sample, any residual liquid was first discarded. The well was allowed to refill and a sample was removed using a vacuum pump. This sampling method was employed in the OSWTDS field studies of Ijzerman *et al.* (1993) and Monnett *et al.* (1996) as well.

The solution samplers and wells were placed at either 30.5 or 45.7 cm below the trench bottom, depending on soil conditions. Deep wells were installed to a maximum depth of 213.4 cm. Control wells were placed upslope and downslope of the subsystems to a depth directly above the restrictive layer.

Soil moisture status was monitored using tensiometers. The tensiometers were constructed from lengths of 1.3 cm electrical conduit (PVC) fitted with a 100 kPa standard ceramic cup on one end and with an approximately 7.6 cm length of clear acrylic tubing on the other end. The tensiometers were filled with a 50/50 mix of water and ethylene glycol. A rubber septum was inserted into the acrylic tube to create a sealed system. A screw auger was used to open a hole to the desired depth. A slurry of silica flour was added to the hole in an amount sufficient to cover the ceramic cup. Soil was

used to backfill the hole to the surface and the surface was sealed with bentonite. The moisture was read with a Troxler model 2601 neutron moisture meter. The tensiometers were placed at depths of 30.5 and 45.7 cm, below trench and in the areas directly below the absorption field.

Soil access wells were made of 15.2-cm thin-walled PVC tubing. The ends were left open, with one end placed directly on the trench bottom surface. The surface end was covered with 15.2-cm PVC cap to restrict odors and limit inflow of precipitation to the trench bottom. A similar monitoring scheme was utilized to evaluate two Virginia spray-irrigation systems by Monnett *et al.* (1996). A counter on the dosing pump allowed household water use to be monitored.

D. ROUTINE CHEMICAL MONITORING AND ANALYSIS

The site was visited monthly except when weather conditions did not permit travel or sampling. Water samples from the pump chamber and soil solution samples from the experimental systems were placed on ice and transported to Virginia Tech for analysis. Effluent samples were filtered (0.45 um pore size) with the use of a prefilter. Nitrate-N, ammonium-N, ortho-phosphorus, and chloride concentrations, and pH and electrical conductivity determinations were performed on all regularly collected samples. Chemical oxygen demand (COD) and total kjeldahl nitrogen (TKN) were determined periodically on the pump chamber and soil solution samples. Metals (P, K, Ca, Mg, Fe, and Na) were run on all samples periodically. Similar chemical monitoring and analyses were performed by Hagedorn and Reneau (1994) in shallow-placed LPD systems and by Monnett *et al.* (1996) in spray irrigation field studies.

Once in the laboratory at Virginia Tech, each sample was allowed to equilibrate to room temperature. An aliquot was removed to determine the pH using a combination electrode with a Beckman Expandomatic IV pH Meter and to read the electrical conductivity with a Sybron/Barnstead Conductivity Bridge Model PM-70CB. The

remaining sample was millipore-filtered with a prefilter and a 47-mm 0.45 um membrane filter. A three-mL aliquot of the filtrate was analyzed on an Orion Scientific AS 140 AutoAnalyzer equipped with two Orion Scientific AC100 detectors and an Orion Scientific AR 200 strip chart recorder. Nitrate-N and ammonium-N concentrations were generated in ug/mL from the strip chart recorder. Nitrate-N concentrations were determined colorimetrically using a copper coated cadmium reduction technique (USEPA, 1979). Ammonium-N were determined colorimetrically using an indophenol method (USEPA, 1979).

Ortho-phosphorus was determined using a single reagent-ascorbic acid method (APHA, 1980). Necessary dilutions were performed with a Hamilton MicroLab M Diluter. Chloride concentrations in ug/mL were determined using a colorimetric titration procedure (Adriano and Doner, 1982) on a Hakkebuchler Digital Chloridometer. Each sample was run two times, and the concentration values were averaged.

A closed reflux colorimetric COD procedure was used to generate COD values (APHA, 1986). Digestion of samples for TKN followed standard procedures (APHA, 1986) and were analyzed using the indophenol method (USEPA, 1979). Metals analyses were performed on an inductively coupled plasma (ICP) instrument by the Soil Testing Laboratory at Virginia Tech.

E. TENSIOMETER VALUES AND ANALYSIS

Tensiometer readings were reported over a span of time bracketing the microbial tracer and denitrification studies. A Microsoft Excel program analyzed the monthly measurements and converted them to soil matric potential means in the unit kilopascals (kPa). The program took into account the length and depth of each instrument, and adjusted the pressure head from the rubber stopper (where the reading is taken) to that at the ceramic cup placed in the soil at varying depths.

F. MICROBIAL TRACER STUDIES

Microbial tracer studies were performed in the summer (July 31-August 3, 1989) and in the winter (January 29-February 1, 1990). Routine microbiological monitoring at this site was not performed because of the addition of tracer organisms to the soil environment during the tracer studies.

F. 1. PREPARATION--SUMMER 1989

F. 1. a. BACTERIA

An *Escherichia coli* strain, ATCC Number 25922, was selected for resistance to 100 ug/mL spectinomycin (Sigma Chemical Co., St. Louis, MO) and 100 ug/mL sodium azide (Sigma Chemical Co.) with the gradient plate procedure (Melnick et al., 1978). Flasks of tryptic soy broth (TSB; Difco Laboratories, Detroit, MI) were inoculated with the antibiotic-resistant strain and were incubated at 37^{o}C for 24 h. Three L of nutrient broth (NB, Baltimore Biological Laboratories Microbiology Systems, Baltimore, MD) were inoculated with 100 mL of starter culture. Six L of culture were prepared in this manner for each subsystem; and the twelve L were incubated at 30^{o}C for 12 h, then transferred into large polypropylene bottles and placed on ice in coolers for ten to 12 h until poured into the pump chamber on-site. All inoculations and transfers were performed in a laminar flow transfer hood. Selection of antibiotic resistant strains followed the methodology reported by Ijzerman *et al.* (1992) and Ijzerman *et al.* (1993) regarding similar OSWTDS tracer studies.

F. 1. b. VIRUS

A 100 mL flask of TSB was inoculated with 1.0 mL of host *Escherichia coli* (ATCC Number 12435) and incubated at 37^{o}C. Once the bacteria initiated growth phase (approximately two h), 0.1 mL of bacteriophage f2 (ATCC Number 15766-B1) was

added, and the flask was incubated at 37°C for 18 h. This served as the starter flask of phage.

Four 100 mL flasks of TSB were inoculated with the host *Escherichia coli* (ATCC Number 12435) and incubated at 37°C for 12 h. One 100 mL flask of starter culture was prepared for each three L of NB, and 12 L total was added to the pump chamber. After addition of host to each four L flask, two h were allowed for the bacteria to enter the growth phase, and fifteen mL of the previously-prepared bacteriophage f2 (described above) was added to each four L flask. All four L flasks were incubated at 30°C for 12 h. The prepared 12 L were transferred to several large polypropylene bottles and placed on ice in coolers for ten to 12 h until poured in the pump chamber on-site.

F. 2. PREPARATION--WINTER 1990

F. 2. a. BACTERIA

A second *E. coli* strain was selected for resistance to 100 ug/mL nalidixic acid (Sigma Chemical Co.) and 100 ug/mL sodium azide. The strain was isolated from the pump chamber at another field site and included in this project to provide a different tracer from that used in the summer. Preparation of this tracer was performed in the same manner as the Summer 1989 bacterial tracer.

F. 2. b. VIRUS

A 50 mL flask of TSB was inoculated with 0.1 mL of host strain *E. coli* B and incubated overnight at 37°C. Twelve h after inoculation, 1.5 mL of this stock culture was inoculated into a 150 mL flask of TSB. This culture was placed on a Lab-line rotary shaker and shaken at 250 oscillations/min for two h at 37°C. The flask was removed from the shaker and placed back in the laminar flow hood, where the 1.5 mL inoculation

of the bacteriophage T1 (Biology Department, VPI&SU) was performed. The culture was once more placed on the shaker at 150 oscillations/min for 24 h.

Starter flasks of host *E. coli* B were prepared by inoculating 30 mL flasks of TSB with the bacterium and incubating the flasks at 37^{o}C for 12 h. The 30 mL cultures were transferred into four L Erlenmeyer flasks (one starter per one large flask) and incubated at 30^{o}C for 2.5 h. Twenty-five mL of the prepared bacteriophage T1 stock culture (described above) was added to each four L flask and placed for 12 h at 30^{o}C. All 12 L were transferred to large polypropylene bottles and left on ice in coolers for ten to 12 h, until arriving on-site.

F. 3. FIELD RELEASE

The cultures of the *E. coli* and viral tracers were transported to the field on ice in coolers. The homeowner was asked to switch the dosing pump off one to two d before arrival to ensure a large enough volume of effluent to adequately mix with the tracers.

Upon arrival at the site, the pump chamber lid was removed and all 12 L of tracer bacteria and virus were added. Thirty min were allowed for both cultures to mix with the effluent. During that time, all sampling wells were pumped dry. This water was disposed of. After 30 min, the dosing pump was manually turned on and allowed to pump down to a level slightly below the low-water (pump-off) float (Figure 7). A sample from the pump chamber was taken to estimate colony-forming units (CFU) and plaque-forming units (PFU) of the starting effluent. The pump was then turned off and remained off until after the 72 h collection had occurred.

After the dosing event, the wells were sampled at approximately 24, 48, and 72 h. A vacuum pump was used to place all water in the sampling wells into 250 mL acid-washed Nalgene bottles. The bottles were placed on ice in coolers. If possible, more

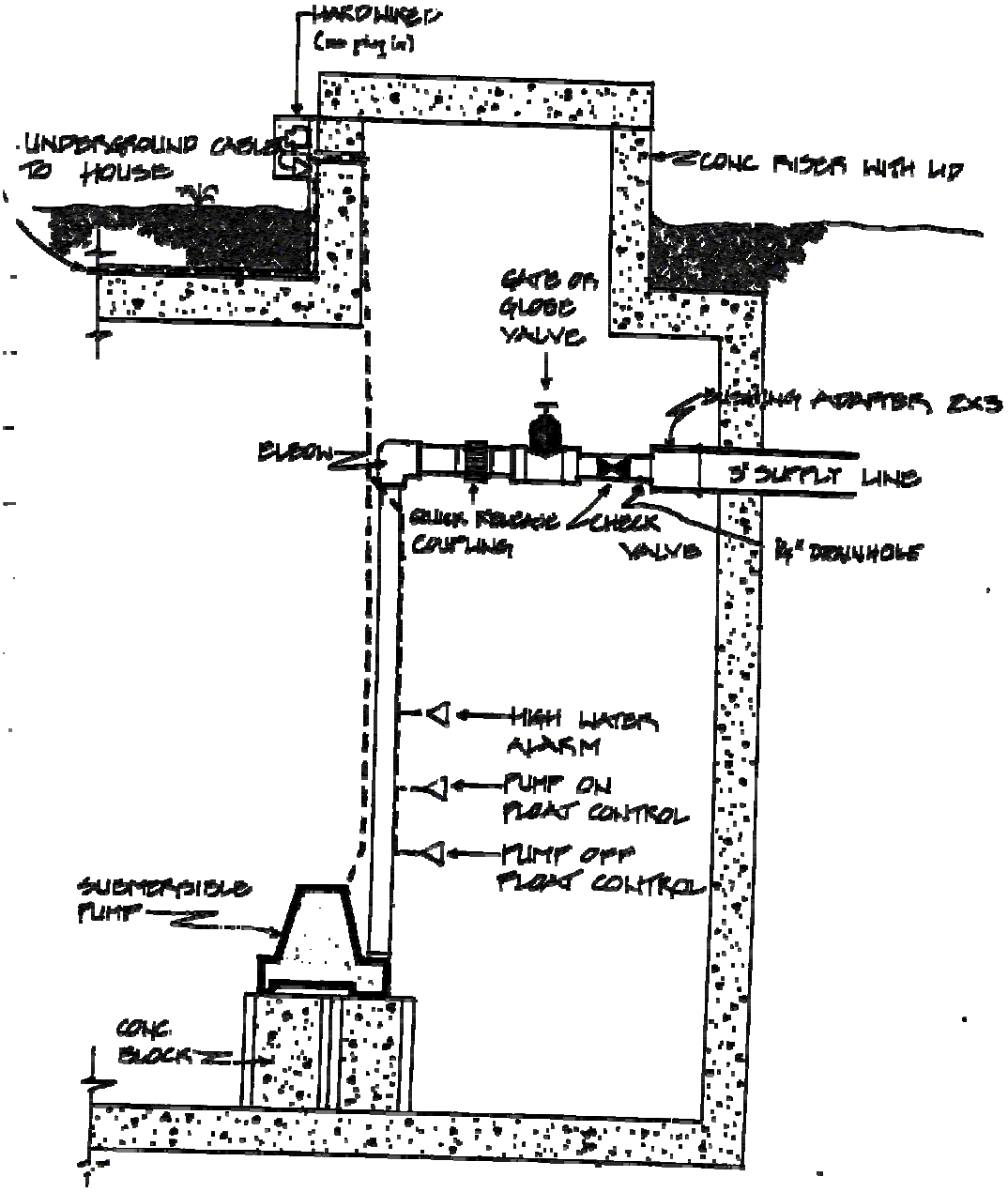

Figure 7. Cross-section sketch of installed pump chamber including pump and piping locations.

than 50 mL of water were pumped from the wells. If more water was left in the well after collecting each 50 mL sample, the well was pumped until empty. A solution of disinfectant and tap water was flushed through the tubing system followed by a second flush with tap water between samples from different wells.

Samples were not taken from the solution samplers equipped with porous ceramic cups due to studies indicating that porous cups do not yield valid water samples for fecal coliform analyses (Wang *et al.*, 1980). DuBois *et al.* (1979) also noted a much higher passage rate through the ceramic material with deionized water than with secondary effluent. Since the present experiment was to recover bacterial and viral tracers from septic tank effluent, it was believed that samples from the sampling wells would be a better measure of indicator organism density than samples collected from porous cup solution samplers.

Previous experiments had demonstrated that phage f2 did not plaque the summer bacterial tracer and phage T1 did not plaque the winter bacterial tracer, so these could be added to the field site simultaneously. Ijzerman *et al.* (1992) and Ijzerman *et al.* (1993) utilized the same procedures in following fate and transport of tracer microorganisms through shallow-placed LPD systems.

F. 4. LABORATORY RECOVERY

In the laboratory, water samples were Millipore filtered with the use of a pre-filter and a 47 mm 0.45 um gridded membrane filter (Cellulosic, White Grid). If 50 mL of sample were not available, distilled water was added to bring the filtrate to a 50 mL total volume.

For the bacterial tracer, the membrane filter was placed on a 100 mm Petri plate (50 x 9 mm, Falcon HAWG-04750) of Eosin Methylene Blue agar (EMB; Difco Laboratories, Detroit, MI) containing the appropriate antibiotics. Plates were incubated at 37^{o}C in a plastic box lined with moistened paper towels for 24-26 h.

For the viral tracer, the sterilized filtrate was added to a warm solution of 50 mL double-strength Trypticase Soy Agar (TSA; Difco Laboratories, Detroit, MI). A 250 mL flask of TSB was inoculated with the host *E. coli* two to four h before filtration of samples began. This culture was placed at 37^{o}C for two h. Three mL of this fresh *E. coli* host was added to the TSA and swirled to mix. Equal portions of the filtrate and agar solution were poured into three Petri plates (100 x 15 mm, Falcon) and allowed to harden at room temperature, and then incubated at 37 C until plaques appeared. These techniques were employed in similar field studies by Ijzerman *et al.* (1992) and Ijzerman *et al.* (1993).

The bacterial tracer was enumerated by counting colonies that grew on the membrane filter and the viral tracer by the number of plaques formed in the agar. Bacterial counts were established for plates having less than 100 colonies. Counts over 100 were said to be greater than 100. Plaque numbers, if under 100, were actually counted. Otherwise, magnitude determinations based on serial dilutions of 10^2, 10^3, and 10^4 were used.

The data was found to reflect a non-normal count distribution, so a SAS program was written to organize the counts into observational categories (Keith Selander, personal communication, 1990). Categories established were as follows:

Bacterial Tracer:
- 0 colony-forming units (CFU)
- 1 - 50 CFU
- 51 - 100 CFU
- >100 CFU

Viral Tracer:
- 0 plaque-forming units (PFU)
- 1 - 50 PFU
- 51 - 100 PFU
- 10^2 - 10^3 PFU
- 10^3 - 10^4 PFU
- >10^4 PFU

For the water samples that were too turbid to filter successfully, a microtechnique for isolating fecal coliforms from soils described previously (Hartel and Hagedorn, 1983) was utilized. Ninety-six well microtiter plates (Falcon 3072) with 12 rows and eight wells per row were filled with a solution of 2X lactose broth (LB, Difco Laboratories, Detroit, MI) and 0.04 percent Bromothylmol Blue indicator dye (Sigma Chemical Co.) amended with 20 ug/mL of each antibiotic.

A 1:1 dilution with a 50-200 uL, eight channel micropipette (VWR Scientific, Bridgeport, NJ) was made in the first row. After that a 1:5 dilution was made (50 uL into 250 uL) with the last row on the plate always used as a control. The plates were incubated 48 h at 37^{o}C. All wells that turned from green to yellow were counted as positive. The number of wells positive were converted to Most Probable Number (MPN) values in colony-forming units (Rowe *et al.*, 1977).

G. DENITRIFICATION STUDIES

Two 15.2 cm PVC soil access wells per subsystem were chosen randomly. Two 30.5 cm soil cores were taken per well with a 30.5 cm soil probe equipped with a 30.5 cm length 2.5 cm diameter acrylic tube. These soil samples were taken during two seasons--Summer/Fall 1989 and Winter 1990. The soil-filled acrylic tube was removed from the probe, capped at both ends, placed on ice in coolers, transported to the lab, and assayed within 24 h.

In the lab, the samples were cut into two parts, 0-15.2 cm and 15.2-30.5 cm. The two halves from the same depths were combined. The soil was removed from the acrylic tube and mixed in a Whirlpak sterile plastic bag. A five g subsample was weighed out in an aluminum weighing pan and placed at 100^{o}C for at least 24 h. The sample was removed and weighed to determine percent soil. This calculation was:

percent soil = weight of dry soil/weight of wet soil.

Approximately ten g of soil was weighed out in 125 mL Erlenmeyer flasks. (Six of these per bag of soil were weighed). Two different treatments were set up in triplicate. The treatments consisted of a 2 mL glucose amendment and a treatment of no amendment. The glucose solution was prepared by dissolving 250 g of glucose in one L of water. This two mL amendment then equaled a 0.5 g glucose addition to the soil. A rubber septum was placed on the flask, and the flask was flushed with greater than ten L/min of helium gas for at least one min. A second needle, inserted in the septum, allowed for release of oxygen gas in the flask. This allowed equilibration to atmospheric pressure. Ten mL of helium gas was removed with a ten cc syringe and replaced with ten mL of acetylene gas in a separate syringe. Two additional controls consisting of only the amendments and no soil were put through the same procedure.

All samples were incubated at 20^{o}C for 48 h. A 0.5 cc (0.5 mL) amount of each gas sample was removed with a 3 cc syringe. The sample was injected into a gas chromatograph (GC) to analyze nitrous oxide production. The Varian 3700 GC was equipped with a 0.6 cm 2 m glass Porapak-Q column and an electron capture detector. The carrier gas used was argon with ten percent methane. An inlet temperature of 60^{o}C, a column/oven temperature of 50^{o}C, and a detector temperature of 350^{o}C allowed good separation of the nitrous oxide peak with a retention time of approximately 1.2 min. Retention times and peak areas were read directly off a Hewlett-Packard integrator set up with the GC. Standards of nitrous oxide were also run to establish a linear regression curve for the N_2O concentrations.

These concentrations were then adjusted for water and gas phases according to procedures described by Tiedje (1982). All calculations were performed with the use of a SAS (SAS Institute Staff, 1982) program. The data were corrected for unequal variances using a Variance Stabilizing Model (Statistics Department, VPI&SU) which determined that a 0.2 log data transformation was required. Analysis of Variance (ANOVA) was performed on nitrous oxide emissions using the General Linear Model (GLM) procedure

of SAS. Mean separations between treatments were performed by Duncan's Multiple Range Test when the overall F-test was significant at $p<0.05$. Similar denitrification studies from other alternative OSWTDS field sites were performed by Hagedorn and Reneau (1994).

IV. RESULTS AND DISCUSSION

A. STUDIES OF AN ALTERNATIVE OSWTDS

The purpose of conventional OSWTDS is to dispose of and treat human wastewater in a safe and sanitary manner. Removal of harmful microorganisms is expected to occur as the effluent percolates through the soil profile. The site chosen for this experiment had soil and site conditions that were not suitable for a conventional OSWTDS as determined by the Commonwealth of Virginia Sewage Handling and Disposal Regulations (1982). Tables 1, 2, and 6 describe the soil characteristics. A shallow groundwater table, as evidenced by grey mottles at 45.0 cm, and low hydraulic conductivity, estimated with a percolation rate of 47.2 min/cm or slower, were the noted restrictions for the soil at the site. However, specially designed low-pressure distribution subsystems with design loading rates of 5.1 and 10.2 Lpd/m^2 were placed in this soil to determine the performance of a shallow-placed low pressure infiltrator system (Table 3). Dosing amounts were monitored and averaged to determine actual flow rates over the period of study (Table 4).

Soil matric potentials were determined with depth both within the absorption area (subsystems 1 and 2) and down-gradient from the absorption area (down-gradient from subsystem 1 and down-gradient from subsystem 2) over the months that tracer and denitrification studies were conducted. Up-gradient and down-gradient background water chemistry from the absorption area (subsystems 1 and 2) was determined prior to and following the research period. Water chemistry from each subsystem in the OSWTDS was determined with regular monitoring over the entire research period that began in Sep 1988 and continued through Mar 1991.

After the soil infiltrator OSWTDS had functioned for approximately 12 months, tracer bacteria and bacteriophage strains were passed through the two subsystems to monitor the fate and transport of microorganisms of public health concern. Ultimately, all treated effluent reaches groundwater and may recharge groundwater that could be used as potable water. It is very important that the soil filters and treats the effluent to an acceptable level of safety before reaching groundwater. Tracer experiments were conducted in the summer and winter seasons since any subsurface absorption system must function under both dry and wet climatic extremes of the year. Laboratory experiments to simulate field denitrification were performed to estimate the soil's ability to change potentially harmful nitrates in the effluent to a less hazardous nitrogen form, nitrous oxide (N_2O), that is released into the atmosphere.

All data collected was analyzed to determine whether the soil infiltrator LPD functioned satisfactorily in the Blairton silt loam, that was characterized by low hydraulic conductivity and a shallow groundwater table (Tables 1, 2, 3, and 6), and to determine if this OSWTDS is a potential alternative system for use in marginal/unsuitable soils.

B. HYDRAULIC PERFORMANCE OF ALTERNATIVE OSWDS

B. 1. FLOW RATES

The soil infiltrator LPD was designed for a daily flow of 1135.6 Lpd based on current Commonwealth of Virginia Sewage Handling and Disposal Regulations (1982) for a two-bedroom house (Table 4). A set dosing volume of 851.7 Lpd, or 75 percent of the designed daily flow, provided a dosing interval of 1.98 d. Monitoring flow rates over time showed an actual flow of 709.1 Lpd, approximately 62.3 percent of the designed daily flow rate of 1135.6 Lpd.

Actual flow rates for the soil infiltrator LPD remained fairly constant over the entire research period (Table 5). Rates ranging from Sep 1988 through Mar 1991

indicate an average flow rate of 709.1 Lpd, or 62.3 percent of the designed flow. Averages were determined for three-month periods during the first year of system operation and then collectively averaged for the remaining study period of 18 months. The last 18 months of the study averaged the lowest actual flow at 571.5 Lpd, or 50.3 percent of design flow. Greatest flow rates of 840.3 and 874.3 Lpd occurred during Apr-Jul 1989 and Jul-Sep 1989, respectively. These were equivalent to 74 and 77 percent of the designed flow, respectively. Homeowner use determined flow through the system so only differences in personal routine can be offered as explanation for the high and low flows experienced over the study period. Overall, however, the system was never put at risk for overloading as the actual flow was always near or below 75 percent of design flow.

The designed loading rate for subsystem 1 was 5.1 Lpd/m^2 to approximate one-half the current Virginia regulation (Table 4). The actual loading rate was determined to be 2.4 Lpd/m^2, approximately 47 percent of the design rate. Subsystem 2 was designed for a loading rate of 10.2 Lpd/m^2, to approximate the current state regulation. The actual loading rate was determined to be 4.9 Lpd/m^2, or 48 percent of the designed rate. Each subsystem was actually loaded at less than 50 percent of the design flow, therefore, no risk of overloading occurred within either subsystem.

The soil infiltrator LPD was designed to meet as many current regulations (State Board of Health, 1982) as possible; the design rates were based on current standards, however, homeowner use determined the actual flow rate, or actual loading rate. For the duration of the research study the house was occupied by only two individuals who most likely away from the home quite a bit (probably worked away from the home). The system was designed for a maximum occupancy of four persons, with the possibility of each utilizing the plumbing for a large part of each day, everyday. In the case of this research, the occupancy limit was never exceeded and actual wastewater, or effluent, generated was quite below maximum allowance, therefore, the soil infiltrator LPD

system was never tested at true design flow. Results and conclusions can only be determined from performance at about one-half design flow for each subsystem (2.4 Lpd/m^2 for subsystem 1 and 4.9 Lpd/m^2 for subsystem 2).

B. 2. SEEPAGE AND PONDING

Both subsystems 1 and 2 were operational in the summer of 1988. During the first year of the research study (1988-89), no evidence of seepage occurred at the base of the soil infiltrator LPD. Water was detected at the drain outlet only once. Visual observations inside the infiltrators through 15.2 cm access ports showed no ponded water in either subsystem in the first six months of operation. In March 1989, 2.5 cm of ponded water was observed in the lower end of subsystem 1. During the summer of 1989 more ponded water was observed in subsystem 1 while subsystem 2 remained dry. As much as 7.6 cm of water was ponded in the lower lateral of subsystem 1 in July 1989. During November 1989 subsystem 1 was dry, however, ponded water was again observed in the winter months.

Between January and March 1991 as much as 7.6 cm of ponded effluent was present in both subsystems. The standing effluent in subsystem 1 was attributed to a soil layer with near fragic characteristics (a discontinuous indurated layer) that extended under subsystem 1, but not under subsystem 2 (Tables 1 and 2). In January of 1991, after heavy rainfall both subsystems were saturated. Samples were collected from surface ponding 22.9 m downslope of the subsystem and from the french drain which collects water approximately 7.6 m downslope of the subsystems. The fecal coliform analysis indicated the surface water contained no fecal coliforms and the french drain sample contained 2 CFU/100 mL. Overall, both subsystems appeared to function well under the designed effluent loads in spite of occasional ponding.

B. 3. HYDRAULIC PERFORMANCE DURING TRACER AND DENITRIFICATION STUDIES

Soil matric potentials, determined with tensiometers, are presented by season during the tracer and denitrification studies in Tables 7 (summer, 1989) and 8 (winter, 1990). Data is categorized as either within absorption area or down-gradient from absorption area. Within the absorption area includes subsystem 1, with a design loading rate of 5.1 Lpd/m^2 (actual loading rate 2.4 Lpd/m^2) and subsystem 2, with a design loading rate of 10.2 Lpd/m^2 (actual loading rate 4.9 Lpd/m^2). Down-gradient from the absorption area includes areas down-gradient from both subsystems (i.e. down-gradient from subsystem 1 and down-gradient from subsystem 2). Pressure head readings were taken from tensiometers installed directly under the absorption area (within absorption area) and from tensiometers placed 365.8 and 762.0 cm down-gradient from the absorption area (down-gradient from subsystems 1 and 2). These readings were converted to soil matric potentials and averaged to determine moisture conditions both within the absorption area and down-gradient from the absorption area.

B. 3. a. SUMMER 1989

Summer 1989 soil matric potentials included those generated from tensiometer readings recorded from tensiometer readings recorded during routine monitoring in the months of May, Jun, Jul, and Sep 1989. Within the absorption area, subsystem 1 was slightly wetter than subsystem 2, at both 30.5 and 45.7 cm, respectively compared. The area down-gradient from subsystem 2 was wetter than the area down-gradient from subsystem 1. Both subsystem 1 (within the absorption area) and the area down-gradient from subsystem 1 appeared wetter with depth during the summer season. Both subsystem 2 (within the absorption area) and the area down-gradient from subsystem 2 appeared drier with depth in the summer season.

Table 7. Soil matric potentials determined for both of the subsystems installed in Blairton silt loam, with depth within the absorption area and down-gradient from the absorption area in the summer 1989*.

Subsystem	Design Loading Rate (Lpd/m^2)	Actual Loading Rate (Lpd/m^2)	Soil Depth (cm)	Mean Soil Matric Potential (kPa) Within Absorption Area	Mean Soil Matric Potential (kPa) Down-gradient Absorption Area
1	5.1	2.4	30.5	-8.24	-16.91
			45.7	-7.58	-15.91
2	10.2	4.9	30.5	-12.22	-9.06
			45.7	-12.48	-9.78

*May, June, July, and September.

Table 8. Soil matric potentials determined for both of the subsystems installed in Blairton silt loam, with depth within the absorption area and down-gradient from the absorption area in the winter 1990*.

Subsystem	Design Loading Rate (Lpd/m^2)	Actual Loading Rate (Lpd/m^2)	Soil Depth (cm)	Mean Soil Matric Potential (kPa) Within Absorption Area	Mean Soil Matric Potential (kPa) Down-gradient Absorption Area
1	5.1	2.4	30.5	-15.82	-43.97
			45.7	-12.21	-44.90
2	10.2	4.9	30.5	-23.42	-32.16
			45.7	-26.18	-47.40

*November, January, February, and March.

Once the soil infiltrator LPD system was operational, every 1.98 d 851.7 L of effluent were dosed into the absorption area which consisted of subsystems 1 and 2. A discontinuous indurated layer below subsystem 1 most likely acted as a restriction under much of that subsystem, slowing the effluent percolation rate. As water will travel the path of least resistance, the french drain, installed parallel to the laterals and down-gradient to shed excess precipitation and soil water from the site, would offer an easy path in saturated conditions. Matric potentials for subsystem 1 (within the absorption area) and the area down-gradient from subsystem 2 have similar matric potentials, possibly indicating saturated flow moving diagonally toward the french drain.

B. 3. b. WINTER 1990

Winter 1990 soil matric potentials included those generated from tensiometer readings recorded during routine monitoring in the months of Nov 1989, Jan, Feb, and Mar 1990. Subsystem 1 (within the absorption area) was slightly wetter than subsystem 2 (within the absorption area). Areas down-gradient from each subsystem, however, maintained similar moisture conditions. Subsystem 2 (within the absorption area) and the area down-gradient from subsystem 2 were drier with depth, as in the summer study.

B. 4. SUMMARY

Both subsystems functioned very similarly in regard to hydraulic performance over both seasons. On several occasions during the three year research study ponding was noted in both subsystems but especially in subsystem 1 which had the indurated layer beneath it. Most ponding however occurred just after a dosing event.

The data indicates that winter moisture conditions were drier than summer conditions, opposite to the anticipated result. Normally, weather events cause the groundwater table to rise over the winter months, saturating the lower soil horizons. In the case of this site grey water mottling at 45.0 cm (Tables 1, 2, and 3) indicates a

recurring high groundwater table in this soil. The moisture data for this site was collected to enhance the microbial tracer and denitrification studies and generated for only a short period in both summer and winter seasons. Over these short intervals normal seasonal moisture patterns may not be apparent, rather only a set of conditions determined within a short window of time at that site.

As neither subsystem was tested at its design flow rate, potential use at those design rates can not be predicted or estimated. At the actual loading rates of 2.4 and 4.9 Lpd/m^2, both subsystems 1 and 2, respectively, appeared to perform adequately in terms of hydraulics. It is possible that the two subsystems, installed side by side with only the manifold trench between them, did communicate effluent between them. To test for the possibility of this occurrence, a chemical tracer or dye could be added to the effluent prior to dosing and soil water samples collected a determined interval after the tracer dye dosing cycle. The soil infiltrator LPD system as installed, did not have a mechanism to test for this possibility, however, further research with this type of OSWTDS might include a mechanism such as a chemical tracer to test the potential of soil water communicability between two closely installed infiltrator systems.

C. WATER QUALITY IN ALTERNATIVE OSWTDS

C. 1. BACKGROUND WATER QUALITY

At the beginning of operation of the soil infiltrator LPD, background water quality was tested by collecting soil water samples up-gradient of subsystems 1 and 2 (Table 9). Near the end of the research study additional background water quality tests were conducted from sampling wells and solution samplers within both subsystems 1 and 2 and down-gradient 365.8 and 762.0 cm of both subsystems 1 and 2. These measurements are also included in Table 9 for comparison.

Hydrogen ion concentrations (pH) remained near neutral, ranging only between

6.50 and 7.50 in all locations sampled. Electrical conductivity (EC) and chloride ion concentrations appear to have been diluted by effluent dosings and soil water. Nitrogen in the forms of ammonium-N and nitrate-N appear quite low for those expected beneath and within eight m of an OSWTDS. Phosphate-phosphorus concentrations are also quite low in these soil water samples as well.

As a reference, the current potable drinking water standard for nitrate-N set as a United States Environmental Protection Agency (USEPA) Maximum Contaminant Level (MCL) is 10.0 mg/L. Only the nitrate-N averages for subsystem 1 in 1991 at both depths tested (30.5 and 45.7 cm) actually exceeded that standard. These concentrations indicate effluent application of N directly from the septic tank/pump chamber with little opportunity yet for purification by the soil environment. Most likely a bit of travel through the soil environment would further reduce those concentrations prior to joining either ground and/or surface waters that could be utilized as a potable drinking water source.

Water quality parameters from areas around subsystems 1 and 2 were comparatively similar with only a few exceptions. Overall, up-gradient concentrations/values were greater for most parameters than down-gradient concentrations/values. Travel through the absorption area appears to have enhanced background water quality most likely by dilution.

Table 9. Background water quality parameters with depth surrounding Subsystems 1 and 2 prior to and following microbiological tracer and denitrification studies.

Subsystem	Soil Depth	pH	EC (umhos/cm)	Cl^-	NO_3^-	NH_4^+	PO_4^{2-}
				(mg/L)			
1989 Up-gradient of Subsystems							
1	*	6.80	748	42.4	7.16	2.09	0.05
2	*	6.50	401	43.9	9.26	2.17	0.09
1991 Within Subsystems (Within Absorption Area)							
1	30.5	6.99	393	44.2	13.04	2.15	0.07
	45.7	6.70	437	44.1	11.24	1.54	0.16
2	30.5	7.35	1122	47.6	4.68	3.19	0.03
	45.7	6.86	659	40.2	6.86	1.76	0.09
1991 Down-gradient of Subsystems							
Lateral distance of 365.8 cm from west side of absorption area							
1	30.5	#	#	#	#	#	#
	45.7	6.85	242	13.4	2.26	0.21	0.04
2	30.5	#	#	#	#	#	#
	45.7	6.82	237	4.8	1.01	0.27	0.23
Lateral distance of 762.0 cm from west side of absorption area							
1	30.5	7.05	230	15.0	0.12	0.19	0.01
	45.7	6.83	214	11.6	1.19	0.14	0.08
2	30.5	7.47	210	5.3	9.65	0.39	0.03
	45.7	7.24	291	9.1	1.04	0.30	0.04

*Values were averaged per subsystem, therefore, no soil depth measurement is given.
#Water quality testing was not performed on these samples.

C. 2. EFFLUENT AND ABSORPTION AREA SOIL WATER QUALITY

C. 2. a. EFFLUENT WATER QUALITY DURING THE RESEARCH STUDY

During the three year research period, effluent quality from the pump chamber registered pH values of 6.80 to 7.59, EC values of 527 to 807 umhos/cm, chloride ion concentrations of 30.6 to 49.2 mg/L, nitrate-N concentrations of 0.07 to 0.87 mg/L, ammonium-N concentrations of 7.42 to 31.00 mg/L and phosphate-P concentrations of 1.70 to 4.73 mg/L (Table 10). Total Kjeldahl Nitrogen (TKN), determined only for the 1991 effluent samples, averaged 37.2 mg/L.

Values fluctuated during 1989 and 1990, however, the highest value/concentration for each parameter occurred in 1991, the third and last year of the research study. Not only were the 1991 values/concentrations high, but they were much increased over those determined in 1990. Most likely this occurred due to aging of the system, with build-up of waste materials and therefore, nutrient load, in the septic tank. Additional nutrients would then be expected to move with the aqueous phase into the pump chamber. Changes in household uses and activities may also have influenced the effluent quality, with time. Overall, the effluent produced from this household was very weak, containing little organic matter and very small amounts of detergents. Homeowner use determined not only the amount, but also the strength of the effluent dosed into the soil infiltrator LPD system. The performance of this system as a test OSWTDS must be evaluated with that concept in mind. A stronger effluent, or higher strength wastewater, might have stressed the soil environment to a greater extent, producing a lesser quality soil water and a greater potential threat to ground and/or surface waters.

C. 2. b. ABSORPTION AREA WATER QUALITY DURING THE RESEARCH STUDY

Hydrogen ion concentration (pH), chloride ion concentration, ammonium-N concentration, and phosphate-P concentrations stayed constant within the absorption area (subsystems 1 and 2, collectively) over the three year study period (Table 10). EC values were also constant except for a high value of 1017 umhos/cm in subsystem 1 at 30.5 cm depth in 1990, and 1122 umhos/cm in subsystem 2 at 30.5 cm below trench in 1991. Both of these values exceed the 1990 and 1991 effluent EC values of 652 and 807 umhos/cm, respectively.

Nitrate-N concentrations were fairly constant over time with a greater concentration of 12.8 mg/L from subsystem 2 at 30.5 cm depth in 1989, and greater concentrations of 13.0 and 11.2 mg/L from 30.5 and 45.7 cm depths, respectively, in subsystem 1 in 1991. These greater concentrations correspond with higher ammonium-N concentrations of 21.7 mg/L in 1989 and 31.00 mg/L in 1991. Overall, the effluent was very weak and contained a much smaller concentration of nitrogen than would have been expected for an OSWTDS.

C. 2. b. 1. SUBSYSTEM 1 WATER QUALITY DURING THE RESEARCH STUDY

Over the research period, hydrogen ion concentrations were consistently reduced from 30.5 cm to 45.7 cm with the 45.7 cm values less than the corresponding effluent pH. EC was not determined in the 1989 subsystem samples. In 1990 at 30.5 cm EC was quite high at 1017 umhos/cm. At 45.7 cm EC had been reduced to 599 umhos/cm, less than the effluent average for the year. In 1991 effluent EC was reduced to one-half at 30.5 cm. EC at 45.7 cm was slightly increased to 437 umhos/cm.

Table 10. Effluent and soil water quality means over the three-year study period for the soil infiltrator LPD designed for a daily flow of 1135.6 Lpd.

Year	Sampling Location	Soil Depth (cm)	pH	EC (umhos/cm)	Cl^-	NO_3^-	NH_4^+	PO_4^{2-}	TKN
					(mg/L)				
1989	Pump Chamber	*	7.10	527	30.6	0.07	21.70	1.70	#
	Subsystem 1	30.5	7.20	#	51.2	6.02	3.07	0.07	#
		45.7	6.60	#	37.1	7.73	1.57	0.04	#
	Subsystem 2	30.5	6.40	#	36.7	12.80	4.50	0.12	#
		45.7	6.50	#	47.7	7.08	0.74	0.08	#
1990	Pump Chamber	*	6.80	652	31.2	0.55	7.42	1.72	#
	Subsystem 1	30.5	7.25	1017	42.0	4.80	2.40	0.05	#
		45.7	6.70	599	35.0	7.30	1.45	0.03	#
	Subsystem 2	30.5	6.60	351	35.0	8.50	2.40	0.08	#
		45.7	6.50	374	39.0	6.60	0.46	0.09	#
1991	Pump Chamber	*	7.59	807	49.2	0.87	31.00	4.73	37.2
	Subsystem 1	30.5	6.99	394	44.2	13.00	2.15	0.06	#
		45.7	6.70	437	44.1	11.20	1.54	0.16	#
	Subsystem 2	30.5	7.35	1122	47.6	4.68	3.19	0.03	#
		45.7	6.86	660	40.5	6.86	1.76	0.09	#

*The soil depth measurement does not apply to the pump chamber sample collection.
#Water quality testing was not performed on these samples.

Chloride ion concentrations in subsystem 1 were more concentrated than corresponding effluent concentrations in both 1989 and 1990. In 1991 effluent chloride concentrations exceeded subsystem chloride concentrations. In all three years concentrations at 45.7 cm were reduced from those determined at 30.5 cm. The chloride ion is an important species to consider in soil water movement as it is very stable and does not undergo any transformations within the soil environment. Any changes noted in the chloride ion concentration are due then to dilution or concentration of the soil water.

As expected subsystem nitrate-N concentrations greatly exceeded the corresponding effluent concentration. In 1989 and 1990 nitrate-N concentrations increased slightly from 30.5 to 45.7 cm deep. In 1991, 13.00 mg/L of nitrate-N was documented at 30.5 cm and 11.20 mg/L of nitrate-N was documented at 45.7 cm. Pump chamber effluent was also increased in 1991 to 0.87 mg/L.

Ammonium-N concentrations decreased significantly from effluent to subsystem, as expected. Concentrations were consistently higher at 30.5 cm than at 45.7 cm. Phosphate-P was greatly reduced between effluent and subsystem determinations. In 1989 and 1990 slightly greater concentrations occurred at 30.5 cm as compared to 45.7 cm. In 1991 the 30.5 cm determination was consistent with the previous concentrations, however, a slight peak of 0.16 mg/L appeared at 45.7 cm. The effluent phosphate-P concentration was higher that year (4.73 mg/L) as compared to the two previous years.

C. 2. b. 2. SUBSYSTEM 2 WATER QUALITY DURING THE RESEARCH STUDY

Subsystem pH values were slightly decreased from corresponding effluent pH values. In 1989 and 1990 average pH values at both depths ranged only between 6.40 and 6.60. In 1991 pH values rose at both depths to 7.35 at 30.5 cm and 6.89 at 45.7 cm. Effluent pH was also highest in 1991 as compared to the first two years of the research.

EC was not determined on subsystem samples in 1989. In 1990 subsystem values were much reduced from the effluent determination. The EC at 45.7 cm was slightly

higher than the EC at 30.5 cm. In 1991 a greater EC value of 1122 umhos/cm was determined at 30.5 cm. At 45.7 cm EC was reduced from the effluent value of 807 umhos/cm.

Chloride ion concentrations followed the same pattern as subsystem 1 with subsystem concentrations greater than effluent concentrations in 1989 and 1990. Concentrations at 45.7 cm were greater than those at 30.5 cm in both years. In 1991 effluent concentration exceeded those in subsystem 2. In this case the concentrations decreased with depth, indicating dilution of the effluent had occurred.

Nitrate-N concentrations were greatly increased between pump chamber and subsystem. In 1989 and 1990 nitrate-N concentrations decreased between 30.5 and 45.7 cm, however, in 1991 a higher concentration was determined at 45.7 cm as compared to the determination at 30.5 cm.

Ammonium-N concentrations between effluent and subsystem were greatly reduced. In all three years average concentrations were decreased from 30.5 to 45.7 cm. Phosphate-P concentrations were greatly reduced between effluent and subsystem. In 1989, the 30.5 cm depth registered slightly higher than the 45.7 cm depth. In 1990 and 1991 the concentrations at 45.7 cm were higher than those at 30.5 cm.

C. 2. b. 3. SUBSYSTEM COMPARISON

There are no notable differences between the overall performance of either subsystem 1 or 2 with regard to water quality. The effluent strength tended to be weaker than expected, due to homeowner use. Each subsystem performed well, however, due to the weakness of the effluent neither was as fully tested as would have been if a stronger household effluent had been generated.