National Decentralized Water Resources Capacity Development Project

Performance of Engineered Treatment Units and Their Effects on Biozone Formation in Soil and System Purification Efficiency

Environmental Science and Engineering
Colorado School of Mines
Golden, Colorado

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Golden, Colorado

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DISCLAIMER

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ABSTRACT

The research described in this report investigated the field performance of onsite wastewater systems (OWSs) using engineered treatment units followed by soil treatment. The goal of this type of OWS is to enable higher—or equivalent—performance at higher hydraulic loading rates (HLRs) and/or less unsaturated soil depth.

The primary objectives of the research were:

- Delineate the effluent quality over time with respect to chemicals and pathogens from treatment units which produce effluents of differing qualities
- Determine the effects of higher effluent quality on soil clogging and biozone development during effluent infiltration and percolation in soil
- Determine the treatment efficiency achieved by tank-based treatment units and soil treatment unit operations for selected chemicals and pathogens

The research was completed through field experimentation at the Mines Park Test Site located on the Colorado School of Mines (CSM) campus in Golden, Colorado.

In this project, the effluent qualities produced by three different treatment units—septic tank, septic tank with textile filter unit (TFU), and septic tank with a membrane bioreactor (MBR)—were characterized in detail. The effects of these effluent qualities on the hydraulic and purification performance achieved during soil treatment in an Ascalon sandy loam soil were studied. Full-scale treatment units and pilot-scale soil test cells were established and started up during spring 2004. Operation of the test cells and engineered treatment units continues.

The effluents generated by the septic tank, TFU, and MBR units, after a period of start-up operations, were consistent in quality for each unit. As expected, the three treatment units achieved different treatment efficiencies for organic matter, solids, nutrients, and bacteria. The relative efficiency ranking shows: septic tank effluent (STE) << TFU << MBR. The relative ranking for operational complexity, operation and maintenance requirements, energy use, and cost, followed a similar pattern: STE << TFU << MBR. Due to the short duration of the performance evaluation completed, it is difficult to estimate the service life, long-term operation and maintenance requirements, or life-cycle costs of the OWSs using a TFU or MBR.

Adding an engineered treatment unit to produce higher-quality effluent than typical STE, can retard soil-clogging development and enable application of higher HLRs to soil and, concomitantly, smaller soil treatment units (assuming purification is reliably achieved over the service life of the system). However, the magnitude of the increases in HLRs enabled by higher effluent quality is likely limited by the hydraulic properties of the natural soil. It may be

reasonable to limit the daily design HLR for a given soil treatment unit regardless of effluent quality, to a small percentage of the soil's saturated hydraulic conductivity (K_{sat}) (for example, design daily HLR = 3 to 5% of the K_{sat}).

The treatment train purification for chemicals such as organic matter and nitrogen was extremely high (more than 99%), and it follows a trend of higher performance: septic tank + MBR + soil treatment is greater than septic tank + TFU + soil treatment, which is greater than septic tank + soil treatment.

The treatment trains including a TFU or MBR, generally perform better with respect to purification. They are less affected by HLR than the treatment train based on only STE and soil treatment. The overall performance of the treatment trains with a TFU or MBR is relatively better with 60 cm of soil. Increasing the vadose zone soil depth (for example, from 60 cm to 120 cm) tends to shrink the differences in performance between the three treatment trains. The ability of an Ascalon sandy loam soil to remove virus was quite high by 60 cm. At that depth it was insensitive to whether the natural soil had received STE, TFU effluent, or MBR effluent at experimental design HLRs of either two or eight centimeters per day (cm/d).

The results of bromide tracer tests and infiltration rate measurements and modeling reveal that some degree of soil clogging and biozone formation is occurring in the Ascalon sandy loam soil, even with higher-quality effluents applied. Also, viruses are being effectively removed (removal in soil of about 6-logs).

During this project, a major field experiment was established and operations were initiated, yielding an array of treatment unit operations and performance data over a period of approximately six months (April to October 2004). This research duration has provided valuable insight concerning the startup and early operation and performance of an OWS, but a longer period of monitoring and assessment is needed to develop long-term data and provide greater insight relevant to full-scale system operation.

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

EXECUTIVE SUMMARY

More than 25% of the US population and 37% of all new developments use OWSs [United States Environmental Protection Agency (US EPA) 2002]. Traditionally, OWSs are comprised of septic tanks for preliminary treatment of raw wastewater followed by percolation through soil, which acts as a porous media biofilter to achieve purification before groundwater recharge.

In many areas, engineered treatment units (for example, sand filters or textile media biofilters) may be implemented. These units may be used in situations where the standard specifications cannot be met with a conventional system or in sensitive areas (such as those with nitrogen loading concerns). With the installation of these engineered treatment units, effluent discharged to soil can be of secondary or higher quality, thereby mitigating the need for stringent siting specifications (Wren 2003). For example, the application of higher-quality effluent to soil may enable use of higher HLRs or system installation at sites where shallower depths of unsaturated soil exist.

The research described in this report investigated the field performance of OWSs using engineered treatment units followed by soil treatment. The goal of this type of OWS is to enable higher—or equivalent—performance at higher HLRs and/or with less unsaturated soil depth. The primary objectives of the research were:

- Delineate the effluent quality over time with respect to chemicals and pathogens from treatment units which produce effluents of differing quality
- Determine the effects of higher effluent quality on soil clogging and biozone development during effluent infiltration and percolation in soil
- Determine the treatment efficiency achieved by tank-based treatment units and soil treatment unit operations for selected chemicals and pathogens

To accomplish these objectives, a major field experiment was established, and operations were initiated. An array of treatment unit operation and performance data was yielded over six months (April to October 2004).

The effluent qualities produced by three treatment unit operations installed at the Mines Park Test Site were assessed to provide insight into the median values and temporal fluctuations achieved by the contrasting unit operations.

The three pilot-scale units include:

- A typical septic tank
- A septic tank unit with an Orenco Systems, Inc. TFU with recycle for in-tank denitrification
- A septic tank followed by aeration with a Zenon Environmental Corp. MBR

These three units were selected because they represent distinctly different types of unit operations, regarding:

- Stage of development and use in OWSs
- Treatment processes employed
- Effluent quality produced
- Operational robustness
- Conduciveness to monitoring and performance assurance

Septic tanks are widely employed in OWSs with TFU applications rapidly growing. While MBRs have not been evaluated for use in an OWS, they represent a potentially promising new method of treatment for such applications, particularly for cluster systems.

The septic tank used for this study is located near an apartment building at Mines Park on the Colorado School of Mines (CSM) campus and has been in operation since September 1998. This system is comprised of two 5,700-liter (L) (1,500-gallon) concrete septic tanks followed by a 3,800-L (1,000-gallon) concrete chamber used for septic tank effluent (STE) collection and sampling in previous CSM laboratory studies.

An Orenco Systems, Inc. (Sutherlin, Oregon) TFU was selected for this study with the core of the treatment unit—the AdvanTex Filter—comprised of a fiberglass basin filled with an engineered textile material. The unit selected for this study—an AdvanTex AX20—provides 20 square feet (horizontal cross-section) filled with hanging sheets of textile media designed to treat domestic STE for removal of biochemical oxygen demand (BOD), total suspended solids (TSS), and total nitrogen (N). Due to limited effluent volume available for this research, the AX20 system was scaled down in size. Only 25% of the total textile pod area [approximately 0.46 m^2 (4.3 ft^2)] received STE. The Orenco Systems, Inc. Mode 1(a) configuration, where recirculation occurs through the second compartment of the septic tank, was employed during this project.

The HomeSpring MBR (Zenon Environmental Corporation, Canada, www.zenonenv.com) was selected for study in this project. The ZeeWeed® membrane fiber is commercially available and widely used in municipal water treatment, but it has not been evaluated for small-scale onsite treatment applications. The ZeeWeed® MBR process is a proprietary technology that consists of a suspended-growth biological reactor combined with ultra-filtration membranes. The unit used in this study was an experimental unit comprised of an anoxic tank, an aerobic tank, the ZeeWeed® membrane, a compressor, and a peristaltic pump, located in a temperature controlled building.

Assembly, installation, and startup of the TFU and MBR occurred during the fall of 2003 and were completed by the spring of 2004. Characterization of the operation and performance of the septic tank, TFU, and MBR was conducted continuously during and after system startup. During startup of each TFU and MBR, daily observations were made of operation and effluent quality. With increased time of operation, as with the septic tank unit, weekly and biweekly observations were made to characterize the performance and effluent quality of the treatment units.

After a period of start-up operations, the effluents generated by the septic tank, TFU, and MBR unit were consistent in quality. As expected, the three treatment units achieved different treatment efficiencies for organic matter, solids, nutrients, and bacteria. The relative efficiency ranking is:

STE << TFU << MBR

The relative ranking for operational complexity, operation and maintenance requirements, energy use, and cost followed a similar pattern:

STE << TFU << MBR

Due to the short duration of the performance evaluation, it is difficult to estimate the service life, long-term operation and maintenance requirements, and life-cycle costs of the OWSs employing TFUs or MBRs.

Soil treatment of STE is commonly relied on as an integral component of an OWS due to high purification performance, limited operation and maintenance requirements, relatively low cost, and the soil treatment unit's long service life. However, there is a growing interest in developing engineered tank-based treatment systems that can produce higher-quality effluents than STE. These systems could reduce the reliance on soil for further treatment.

In concept, the use of a reliable and efficient engineered treatment unit can enable OWSs to be used in settings with unsuitable or poorly-suited site conditions (for example, limited lot sizes, limited depth of vadose zones). However, limited field research has been done that demonstrates the treatment efficiency achievable in soil as a function of the effluent quality applied at different HLRs. To enhance understanding on this subject, research was conducted involving controlled field experimentation using effluents of three qualities applied at two loading rates to replicate *in situ* soil test cells.

The field experiment was initiated during fall 2003 when 18 *in situ* test cells were established at the Mines Park Test Site at CSM. The soils at the Mines Park Test Site are primarily fine, loamy soils (mixed, mesic Aridic Argiustolls). These test cells were installed with an open horizontal soil infiltrative surface. They were loaded with three different effluents (septic tank, TFU, and MBR) at two HLRs (2 and 8 cm/d). Each condition was replicated three times (3 effluent qualities $\times$ 2 HLRs $\times$ 3 replicates = 18 test cells). A set of six ancillary test cells was also installed with gravel at the infiltrative surface. Effluent is dosed to each test cell once an hour during a 16-hr period (7 am to 10 pm) each day through a single orifice in the center of the cell.

Test cells were installed within a trench with the infiltrative surface (that is, bottom of the trench) located at approximately 90 cm below ground surface. Galvanized steel culvert sections, each 60 cm in diameter, were placed in the trenches and pressed into the soil surface to approximately 2.5 cm below the infiltrative surface. The culverts were sealed along their outside circumferences with a native soil slurry and bentonite clay pellets. Each test cell provides about 2,920 cm^2 of bottom area infiltrative surface (no sidewall). The infiltrative surface for each test cell was examined, photographed, and prepared in a similar fashion to remove any anomalous features and ensure replicate testing conditions between test cells. Gravel was placed on the infiltrative surface of the ancillary test cells. Access ports were installed for inspection of the infiltrative surface and for collection of intact soil cores. Finally, the test cells were backfilled and compacted, and the site was graded.

To evaluate the purification of effluent during migration through the vadose zone, porous stainless steel suction lysimeters were installed. The lysimeters were installed at 60, 120, and 240 cm (2, 4, and 8 ft) below the infiltrative surface in 12 of the 18 cells. In some cases, an electrical resistance wafer was also installed to enable measurements of soil temperature and moisture levels. The lysimeters have a nominal pore size of 0.2 microns. The small pore size limits sampling for bacteria, but is necessary to inhibit air from entering the lysimeters instead of soil-water solution.

Routine field monitoring of the soil test cells includes:

- Measurement of applied effluent composition
- Applied effluent HLR
- Hydraulic behavior of the soil infiltrative surface
  - Infiltration rate changes
  - Ponding occurrence
  - Magnitude of ponding
- Soil pore water quality

Adding an engineered treatment unit to produce effluent of higher quality than typical STE can retard soil-clogging development and enable application of higher HLRs to soil and, concomitantly, smaller soil treatment units (assuming purification is reliably achieved over the service life of the system).

The magnitude of the increases in HLRs enabled by higher effluent quality is likely limited by the hydraulic properties of the natural soil. It may be reasonable to limit the daily design HLR for a given soil treatment unit, regardless of effluent quality, to a small percentage of the soil's saturated hydraulic conductivity (K_{sat}) (for example, design daily HLR = 3 to 5% of the K_{sat}).

The treatment train purification for chemicals such as organic matter and nitrogen was high (greater than 99%) and follows a trend of higher performance:

Septic tank + MBR + soil treatment > septic tank + TFU + soil treatment > septic tank + soil treatment.

The treatment trains including a TFU or MBR generally perform better with respect to purification. These units are less affected by HLR than the treatment train based on only a septic tank and soil treatment. The overall performance of the treatment trains with a TFU or MBR was relatively better with 60 cm of soil. Increasing the vadose zone soil depth (for example, from 60 cm to 120 cm) tends to shrink the differences in performance between the three treatment trains. The ability of an Ascalon sandy loam soil to remove virus was quite high by 60 cm and insensitive to whether the natural soil had received STE, TFU effluent, or MBR effluent at experimental design HLRs of either 2 or 8 cm/d. The results of bromide tracer tests, infiltration rate measurements, and modeling reveal that some degree of soil clogging and biozone formation is occurring in the Ascalon sandy loam soil, even with higher-quality effluents applied, and viruses are effectively removed (removal in soil of about 6-logs).

During this project, a major field experiment was established and operations were initiated, yielding an array of treatment unit operations and performance data over a period of approximately six months (April to October 2004). This research duration has provided valuable insight concerning the startup and early operation and performance of an OWS. However, a longer period of monitoring and assessment is needed to develop long-term data and provide greater insight relevant to full-scale system operation.

1 PROJECT BACKGROUND

1.1 Background and Motivation

Over 25% of the US population and 37% of all new developments use OWSs (US EPA 2002). Traditionally, OWSs are comprised of septic tanks for preliminary treatment of raw wastewater followed by percolation through soil. The soil acts as a porous media biofilter (PMB) to achieve purification before groundwater recharge. Due to the high demand for land, development in areas traditionally considered unsuitable for such treatment systems (over two-thirds of the land available in the US) has occurred. Additionally, certain pollutants (such as nutrients) are accumulating in water resources, placing increased demands on the quality of treated effluents discharged to the environment. These progressions, and others, have led to the increased use of engineered treatment units that produce effluent of a higher quality than that of a typical septic tank.

Current regulations for OWSs that rely on conventional septic tank treatment followed by soil percolation specify minimum and maximum values for many features of the system. For example, in Jefferson County, Colorado, current regulations specify:

- Minimum depths to groundwater or bedrock
- Minimum distance from drinking water wells
- Minimum and maximum soil percolation rates
- Minimum distances from cut banks or fill areas
- Maximum natural grades
- Minimum size allotments for the soil absorption (Jefferson County 1999; Wren 2003).

In many areas, engineered treatment units (for example, sand filters, textile media biofilters) may be implemented where these regulatory specifications cannot be met using a conventional system or in sensitive areas such as those with nitrogen loading concerns. With the installation of engineered treatment units, effluent discharged to soil can be of secondary or higher quality, thereby mitigating the need for stringent siting specifications (Wren 2003). For example, the application of higher-quality effluent to soil may enable use of higher hydraulic loading rates (HLRs) or system installation at sites where shallower depths of unsaturated soil exist. In many states, regulations for OWSs allow for higher HLRs to soil when higher quality effluents are applied. For example, Rhode Island allows for soil treatment unit size reductions for certain innovative and alternative technologies, and Washington state permits a 50% reduction in size based on biochemical oxygen demand (BOD_5), total suspended solids (TSS), and fecal coliform bacteria levels.

As illustrated in Table 1-1, there is not universal agreement on the values prescribed for various design requirements.

Table 1-1
Examples of Vertical Separation and Drainfield Sizing Reductions Allowed With Use of an Engineered Pretreatment Unit

State	Reduction in Vertical Separation Allowed	Reduction in Drainfield Sizing (or increase in HLR) Allowed
Alabama		Reviewed/approved on a case-by-case basis
Alaska		Dependent of treatment type effectiveness
Arizona		Reviewed/approved on a case-by-case basis
California		Based on treatment type and loading rate (expressed as lb BOD / 1000 ft^2 / day)
Colorado		Requires approval of health officer
Delaware		Reviewed/approved on a case-by-case basis
Florida		25% Reduction for aerobic treatment units
Georgia		Dependent on type of treatment
Hawaii		Reviewed/approved on a case-by-case basis
Idaho		Dependent on type of treatment
Illinois		Dependent on type of treatment
Iowa		Dependent on type of treatment
Kansas		Reviewed/approved on a case by case basis
Kentucky	Up to 30 cm reduction	Dependent on type of treatment
Montana		Up to 50% size reduction dependent on required monitoring data
New Hampshire		Dependent on type of treatment
New Mexico		Reviewed/approved on a case by case basis
North Carolina		Up to 25% increase in HLR
Oklahoma		Dependent on soil type

Table 1-1
Examples of Vertical Separation and Drainfield Sizing Reductions Allowed With Use of an Engineered Pretreatment Unit (Cont.)

State	Reduction in Vertical Separation Allowed	Reduction in Drainfield Sizing (or increase in HLR) Allowed
Oregon	Up to 1 m reduction based on soil type	Dependent on type of treatment
Pennsylvania		One third reduction for aerobic tank
Rhode Island		35 to 50% Reduction dependent on type of treatment
Virginia		Approximately 30 to 67% increased HLR based on soil percolation rate
Washington		50% reduction dependent on type of treatment

Note: Vertical separation defined as the depth of soil from the point of effluent application (that is, infiltrative surface) to the top of the high seasonal groundwater table.

The interaction of wastewater effluent with soil is complex. During long-term wastewater effluent infiltration into soil or similar porous media, there is normally an accumulation of pore-filling agents. The accumulation occurs at and immediately below the infiltrative surface through which wastewater effluent enters the soil pore network. The reduction in pore size yields a loss in permeability, which in turn affects the hydraulics at the infiltrative surface and within the underlying soil profile.

The rate and extent of clogging development is dependent on several factors (Siegrist *et al.* 2001):

- Soil morphology
- Wastewater composition and loading rate
- Application mode and continuity of use

Microorganisms can play an important role in the formation of a clogging zone.

- Frankenberger *et al.* (1979) proposed biological factors that influence the decrease in hydraulic conductivity:
  - Production of gases
  - Microbial destruction of soil structure
  - Accumulation of metabolic products in the soil pores

  They observed that total biological activity and a selective bacterial population were significantly and negatively correlated with hydraulic conductivity.
- Gupta and Swartzendruber (1962) found a decrease in hydraulic conductivity as a function of bacterial numbers.
- Mitchell and Nevo (1964) found a positive correlation between clogging in sand columns and the accumulation of polysaccharides produced by indigenous dune sand bacteria.
- Lindenbach and Cullimore (1989) created clogging in sand columns using Pseudomonas, Acinetobacter, and Bacillus isolated from clogged soils. They attributed this clogging to a slime or biofilm produced by the bacteria.
- Ronner and Wong (1994) studied bacteria and bacterial extracellular polymers associated with clogged soil pores of OWSs; they isolated approximately 160 bacteria and found that 30% of the isolates tested in small columns induced clogging within two weeks of column inoculation.
- Siegrist (1987) found a decrease in the concentration of polysaccharides corresponded with an increase in clogging severity in pilot-scale infiltration cells dosed with domestic STE. Siegrist concluded that polysaccharide materials were not responsible for soil clogging, but that clogging depended on cumulative loading of total BOD and total solids (TS).

Wastewater effluent application to natural soils results in some degree of pore filling and the development of a clogging zone that can be important from a purification perspective. Not only does the clogging zone reduce the rate of infiltration and thereby contribute to unsaturated flow in the underlying soil profile, it can be more biogeochemically reactive than the natural soil.

While an effluent distribution system to a varying degree can yield an unsaturated flow regime similar to that caused by a soil-clogging zone, the absence of a clogging zone may have additional purification consequences. Clogged infiltrative surface zones are characterized by elevated organic matter accumulations and high water contents along with high microbial densities. The genesis of the soil-clogging zone has been described as a humification-like process. The change in soil properties is most pronounced in clean mineral soils (for example, sands and sandy loams) as compared to finer-grained, more organic soils (for example, silty clay loams). Due to its beneficial properties, the clogging zone is now referred to in a more positive perspective as a biozone.

Based on previous work demonstrating a relationship between effluent quality and HLRs with the rate and extent of soil clogging, design approaches have emerged using engineered treatment

units (for example, sand filters, textile media biofilters) to produce higher-quality effluents. These effluents can retard or eliminate wastewater-induced soil clogging. These systems have been shown to produce effluents of secondary or advanced secondary quality, typically low in BOD_5 and TSS, and in some cases low in total nitrogen.

However, the ability of these units to remove bacteria, viruses, emerging organics, and other contaminants is more uncertain. Despite years of study, the interaction of wastewater and soil is not fully understood. Limited knowledge is available relating effluent quality to soil treatment unit performance, particularly with respect to soil removal of pathogens and emerging organic chemicals.

Information on the pathogen removal efficiency of TFUs has recently been acquired (Wren 2003; Wren *et al.* 2004). As part of a collaborative research project between the Colorado School of Mines (CSM) and Jefferson County, Colorado, performance data from 30 full-scale sand filter and TFU systems were evaluated. Eight of the systems with TFUs were selected for additional study. During a 12-week monitoring period, biweekly samples were collected from each of the eight sites and analyses were made for:

- pH
- Alkalinity
- Solids
- BOD_5
- Chemical oxygen demand (COD)
- Total nitrogen
- Nitrate nitrogen
- Ammonium nitrogen
- Total phosphorus
- Fecal coliform bacteria

Additionally, virus removal was assessed at four of the eight sites by spiking the system with two bacteriophages (MS-2 and PRD-1) and a conservative tracer (potassium bromide).

The results of the research revealed generally high-quality effluent from the TFU with respect to BOD_5, TSS, and total nitrogen, with variations at a given site and between sites (Wren 2003). The virus tracer test revealed removal efficiencies ranging from 0.4- to 0.8-log reduction for virus for each single pass through the textile media filter. Total virus removal rates with recirculation were estimated as high as 3.5-log. These virus removal results are consistent with those reported by Higgins *et al.* (1999) who found a 0.3- to 0.6-log reduction for MS-2 in open-cell foam trickling filters and Foss *et al.* (2002) who later reported a 0.5-log reduction in MS-2.

Engineered treatment units are increasingly being used to enable much higher HLRs of effluent to soil treatment units (for example, 10 to 20 cm/d rather than 1 to 5 cm/d) and to reduce the size of the soil infiltration area or the vertical separation distance to the saturated zone (for example, groundwater). In some cases, the soil infiltration area or vertical separation to groundwater may be reduced by as much as 50% or more (Table 1-1). While this may be technically sound based on biozone development and hydraulic performance considerations, it has implications related to purification, particularly with respect to pathogenic bacteria and viruses (Van Cuyk and Siegrist 2001; Van Cuyk *et al.* 2001b; Van Cuyk *et al.* 2002; Van Cuyk 2003).

The research has shown that packed bed biofilters (for example, sand filters and textile media filters) have the ability to produce effluents with low levels of BOD_5 and TSS (for example, less than 10 mg/L) and in some designs, lower total nitrogen (for example, less than 20 mg-N/L). However, the ability of these systems to remove virus appears to be limited to about 1-log or less for single-pass packed bed biofilters and about 3-logs for multiple-pass designs. STE can contain virus on the order of 10^5 plaque-forming units (pfu)/mL, 5-logs). The evolving Ground Water Rule (GWR) (US EPA 2000), which is based on a paradigm of a 12-log removal in virus from toilet to tap, can be interpreted as requiring removals of about 4- to 8-logs. This implies that infiltration and percolation through 60 to 120 cm of unsaturated soil in a soil treatment unit could be required to remove 2- to 4-logs (or more) of virus, assuming that the deeper vadose zone or underlying saturated zone can provide an additional 2- to 4-logs of virus removal.

It is currently unclear whether an OWS incorporating an engineered treatment unit, such as a TFU or an even more advanced treatment unit such as an MBR, can provide equivalent or better removal efficiency for bacteria and virus than a typical septic tank-soil treatment system. Work by Green (1976) suggests that virus removal is inversely related to the quantity of STE applied to soil; that is, an increase in the dose volume results in a decrease in the total log-removal of added viruses. The absence of a classic biozone in systems employing engineered treatment units might impact the treatment capability of the soil, particularly if higher application rates are used and/or if sites with shallower depths of soil are used.

1.2 Project Objectives

The overall goal of the research is to determine the treatment efficiency in an OWS employing engineered treatment units followed by soil treatment, with the goal of enabling higher HLRs and/or less unsaturated soil depth. The primary objectives of the research were:

- Delineate the effluent quality (median values and fluctuations) with respect to chemicals and pathogens over time from three pilot-scale treatment units which produce effluents of differing quality (septic tank, septic tank with TFU, and septic tank with MBR)
- Determine the effects of higher effluent quality on soil clogging and biozone development during effluent infiltration and percolation in soil
- Determine the treatment train efficiency achieved by tank-based treatment units and soil treatment unit operations for selected chemical and pathogen removal

1.3 Project Approach

This research project was completed through controlled field experimentation that built on previous and ongoing research concerning the hydraulic and purification performance of soil treatment units with particular respect to pathogen fate (Van Cuyk 2003; Van Cuyk and Siegrist 2001). The project approach provided the requisite experimental control at the pilot-scale, while being representative of full-scale operations, to enhance the understanding of design and performance relationships of engineered treatment units and OWSs.

The project was carried out by a team of principal investigators with varied expertise (Figure 1-1 and Table 1-2). In addition, numerous students and staff at CSM have made key contributions to this research project (Table 1-3).

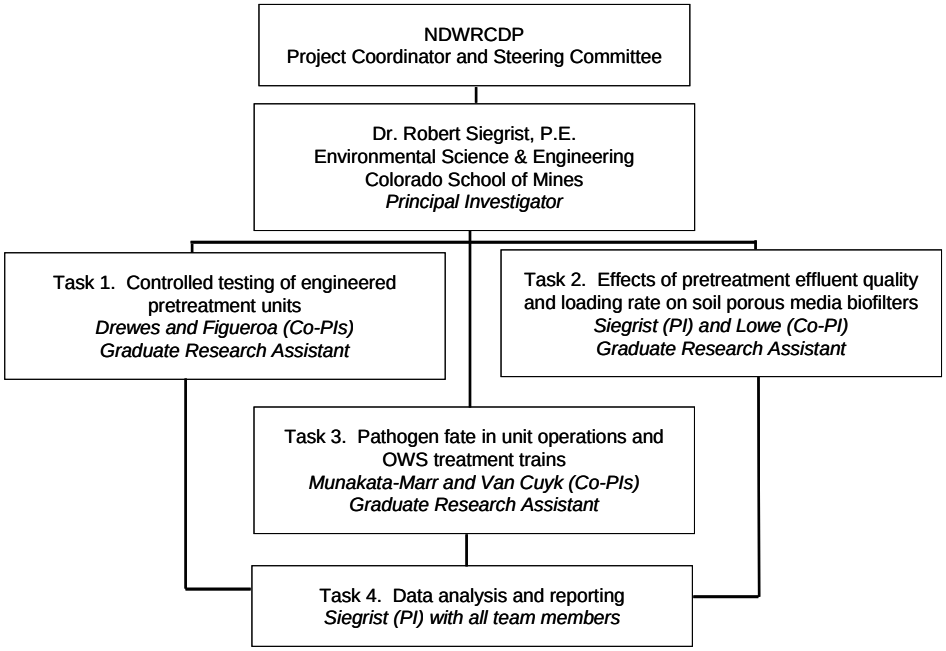


Figure 1-1
Project Team Organizational Chart

Table 1-2
Project Team Members and Their Relevant Expertise and Commitment to the Project

Team Member (*Title*)	Relevant Primary Expertise	Project Role
Robert Siegrist, Ph.D., P.E. (*Professor*)	Wastewater engineering, OWS, PMB, risk assessment, process modeling	PI responsible for project direction and reporting
Sheila Van Cuyk, Ph.D. (*Post-Doctoral Assoc.*)	OWS, soil purification, multicomponent surrogate/tracer methods	Co-PI focused on virus surrogate/tracer testing of treatment units and soil PMBs
Kathryn Lowe, M.S. (*Sen. Research Assoc.*)	OWS, field monitoring and technology evaluation, contaminant hydrology	Co-PI focused on test cell experiments at the Mines Park Test Site
Jörg Drewes, Ph.D. (*Assistant Professor*)	Water and soil aquifer treatment systems, membrane processes, soil bioactivity assays	Co-PI focused on MBR monitoring and evaluation
Junko Munakata-Marr, Ph.D. (*Assistant Professor*)	Environmental microbiology, bacterial source tracking, bioprocesses	Co-PI focused on microbial characterization and assessment
Linda Figueroa, Ph.D., P.E. (*Associate Professor*)	Wastewater engineering, bioprocesses for nutrient removal, system modeling	Co-PI focused on modeling of engineered unit operations

Table 1-3
Student and Staff Research Group Members and Their Involvement in the Project

Group Member	Status	Research Focus
John Albert	Ph.D. Candidate & Research Staff, Environmental Science & Engineering	Microbial characterization, bacterial source tracking, multicomponent surrogate/tracer testing
Kathy DeJong	Ph.D. Candidate, Environmental Science & Engineering	Emerging organic contaminants, multicomponent surrogate/tracer testing
Charlotte Dimick	M.S. Student, Environmental Science & Engineering	Vadose zone sampling, conventional pollutant analysis
Jimmy Kopp	Undergraduate student, Environmental Science & Engineering	Engineered unit installation and monitoring, conventional pollutant analysis
John Luna	Ph.D. Student, Environmental Science & Engineering	MBR operations
Jim McKinley	Ph.D. Student, Environmental Science & Engineering	Biozone characterization
Tanja Rauch	Ph.D. Candidate, Environmental Science & Engineering	Organic carbon characterization, soil bioactivity assays
Kyle Tackett	M.S. Student & Research Staff, Environmental Science & Engineering	Field monitoring and soil coring
Ryan Walsh	M.S. Student, Environmental Science & Engineering	Engineered unit operations monitoring and evaluation at the Mines Park Test Site
Gwen Woods	Undergraduate student, Environmental Science & Engineering, Mathematics and Computer Science	Sample collection, conventional pollutant analysis

This research was completed at the Mines Park Test Site, which was established at CSM to enable controlled field experimentation and improve the understanding of OWS design and performance (see Chapter 2). This test site has been established with funding provided by sources other than the National Decentralized Water Resources Capacity Development Project (NDWRCDP), but provides a key resource for accomplishing the objectives of this project. The test site was established on the CSM campus near the Mines Park student housing complex.

The first phase consisted of the installation of a wastewater interception and pretreatment facility in 1998. This facility has provided a source of STE used for laboratory experimentation over the past several years. The establishment of a field research area was initiated during 2002 to enable controlled, field testing of OWS methods and technologies. A site evaluation of the Mines Park Test Site was completed during spring 2002 (Lowe and Siegrist 2002) and revealed the site to have primarily fine loamy soils (mixed, mesic Aridic Argiustolls) (Appendix A).

Ascalon sandy loam, the dominant soil series present, has a typical soil profile that includes:

- Neutral, sandy loam surface layer (0–18 cm)
- Mildly alkaline, sandy clay loam (18–28 cm)
- Moderately alkaline, sandy loam subsoil layer (28–46 cm)
- Mildly alkaline and moderately alkaline, sandy loam and gravelly sandy loam substratum (46–152 cm).

(See Section 3.2.2 for further site information.)

A major field experiment was initiated at the Mines Park Test Site during fall 2002 to investigate the effects of infiltrative surface architecture and STE HLR on biozone development and soil treatment efficiency. This experiment includes 40 *in situ* test cells representing a pilot-scale soil absorption trench, including three infiltrative surface architectures and two STE HLRs with five replicates of each condition. A set of test cells also receives tap water as a control. System hydraulics are being monitored through infiltration rate measurements and solute tracer tests. Purification for chemicals and pathogens is being studied through sampling of the STE applied and soil solution in the vadose zone at 60 and 120 cm below the infiltrative surface using microporous stainless steel suction lysimeters. Access ports enable inspection of the infiltrative surface and collection of intact soil cores. Due to the relevance of the study objectives and the data being generated, this study is considered a “companion study” for the current NDWRCDP project.

The research described in this report was completed during the period of July 2003 to January 2005 and was supported by funding received from the NDWRCDP, with leveraged funding from other sources (for example, existing test site infrastructure, manufacturer in-kind donations). This research includes a set of four interrelated tasks. Details regarding materials, methods, and procedures are presented in Chapters 2 and 3.

- Task 1. Characterization of the effluent quality produced by engineered treatment units
- Task 2. Field evaluation of the effects of pretreatment and effluent quality on the hydraulic and purification performance of soil treatment
- Task 3. Fate of bacteria and virus in engineered treatment units and soils receiving different effluent qualities
- Task 4. Data analysis, modeling, and reporting

1.4 Report Organization

The report is organized to present background information and motivation for research (Chapter 1) followed by a discussion of the treatment unit operation and performance (Chapter 2). The design and performance of the soil test cells employed in this research are presented in Chapter 3. This chapter includes information generated during the current project and pulls from the ongoing, companion study at the same Mines Park Test Site. Chapter 4 provides discussion that ties results from the treatment units to treatment observed in the soil. Conclusions and recommendations can be found in Chapter 5.

2 WASTEWATER TREATMENT IN ENGINEERED TANK-BASED UNITS

2.1 Introduction

The effluent qualities produced by three treatment unit operations installed at the Mines Park Test Site were assessed to provide insight into the median values and temporal fluctuations achieved by three contrasting unit operations. The three pilot-scale units include:

- Typical septic tank
- Septic tank unit with an Orenco Systems, Inc. TFU including recycle for in-tank denitrification
- Septic tank followed by aeration with a Zenon Environmental Corp. MBR

These three units were selected because they represent distinctly different types of unit operations. The differences include:

- Stage of development and use in OWSs
- Treatment processes employed
- Effluent quality produced
- Operational robustness
- Conduciveness to monitoring and performance assurance

Of these, the septic tanks are widely employed in OWSs, and TFU applications are rapidly growing. MBRs have not been evaluated for use in an OWS, but represent a potentially promising new method of treatment for such applications (particularly for cluster systems).

2.2 Wastewater Source

The domestic STE used in this study was generated by a multifamily apartment complex located at the Colorado School of Mines' Mines Park housing facility located near the corner of Highway 6 and 19th Street in Golden, Colorado. The quality of this STE is described in Section 2.5.2.2 (and Table 2-10).

2.3 Engineered Treatment Units

This section discusses the three treatment unit operations used for this study.

2.3.1 Septic Tank Unit

The septic tank is used as the first (or only) tank-based treatment unit in nearly all OWSs. Septic tanks may be used alone or in combination with other processes to treat raw wastewater before it is discharged to a subsurface soil infiltration system (US EPA 2002). The septic tank is a watertight basin that is buried underground close to the wastewater source. Raw wastewater can flow by means of gravity to the tank. These tanks must have sufficient volume and appropriate geometry to provide adequate hydraulic residence time and quiescent conditions for sedimentation. Septic tanks are anaerobic and have long solids retention times (for example, years) that can enable digestion resulting in a reduction of sludge volume (40%), BOD (60%), suspended solids (70%), and conversion of much of the organic nitrogen to ammonium (Reneau *et al.* 2001). Septic tanks are also important as they attenuate instantaneous peak flows from the dwelling unit or establishment.

Despite the treatment that occurs in a typical septic tank, high concentrations of organic matter, nutrients, and pathogens remain in STE. Further treatment is needed to protect water quality and public health. Table 2-1 presents recent literature values for typical STE and other selected treatment processes.

2.3.2 Textile Filter Unit

In areas with high water tables, shallow or fractured bedrock, or other limiting site features, it may be necessary to provide additional treatment of STE prior to discharge to the soil. Packed bed biofilters (for example, single-pass and re-circulating sand filters) have been established as aerobic treatment technologies for the further purification of STE. They typically yield a high-quality effluent. In the past, sand had been the primary matrix for packed bed filters. However, due to the need for specialized properties (for example, grain size distribution and permeability), the application of sand filters can be difficult and/or costly in many areas. New media, such as non-woven textile fabrics composed of plastic filaments configured in densely packed sheets, provide lightweight material that have a high porosity and surface area. The manufactured media is consistent in quality, readily available, and enable easy serviceability, maintenance, and management of the unit. The effluent generated by these units can be of consistently high quality, allowing for significant reduction in BOD, complete nitrification and partial denitrification, and removal of some pathogens (Leverenz *et al.* 2004).

Table 2-1
Typical Effluent Composition

Constituents of Concern	Measurement (units)	Tank-based Treatment Unit Effluent Concentrations			
		Domestic STE	Re-circulating Sand Filter	Peat	Textile Filter Unit
Oxygen Demand	BOD_5 (mg/L)	120–175[a] 100–140[c]	9–14[a]	3–6[d] 4.8–23[g]	2–5[e] 4–45[b] 5[f]
Particulate Solids	TSS (mg/L)	72–115[a] 20–55[c]	12–15[a]	6–7[d] 2.9–3.4[g]	0–2[e] 1–86[b] 4[f]
Nitrogen	Total nitrogen (mg N/L)	47–51[a] 50–90[c]	24–29[a] 31–24[h]	1–4.1[d] 27–49[g]	10–30[f] 69–83[e] 12–108[b]
Phosphorus	Total phosphorus (mg P/L)	8–9[a] 12–20[c]	Dependent on loading and capacity	Dependent on loading and capacity	Dependent on loading and capacity
Bacteria	Fecal Coliform (cfu/100mL)	$4.9X10^5$ – $6.3X10^5$ [a] 10^6–10^8 [c]	$6.1X10^4$ $6.3X10^5$ [a]	$2.9X10^2$ – $1.6X10^3$ [d] 8–13[g]	1–102[f] $8.0X10^2$ – $3.1X10^4$ [e] 10^2–10^5 [b]
Virus	Specific virus (pfu/mL)	0–10^5 [i]	0–10^5 [i]	0–10^5 [i]	0–10^5 [i]
Organic Chemicals	Specific organics (ng/L) 4-nonylphenol Caffeine Tricloscan 3-β-coprostanol cholesterol Estriol 17β-Estradiol Testosterone	 510[j] 5300[j] 14[j] 1100[j] 2800[j] 15.4–17.6[k] 8.3–10.9[k] 40.5–117.6[k]	Trace levels	Trace levels	Trace levels

References: [a] Christopherson *et al.* 2001; [b] Wren *et al.* 2004; [c] Crites and Tchobanoglous 1998; [d] Lindbo and MacConnell 2001; [e] Loomis *et al.* 2001; [f] Bounds 2002; [g] Geerts *et al.* 2001; [h] Converse 1999; [i] Siegrist *et al.* 2001; [j] DeJong *et al.* 2004; [k] Drewes *et al.* (submitted).

An Orenco Systems, Inc. (Sutherlin, Oregon) TFU was selected for study in this research project (www.orenco.com). The heart of the Orenco Systems, Inc. treatment unit is the AdvanTex Filter that is comprised of a fiberglass basin filled with an engineered textile material. The unit selected for this study, an AdvanTex AX20, provides 20 ft^2 (horizontal cross-section) filled with hanging sheets of textile media designed to treat domestic STE for removal of BOD, TSS, and total nitrogen. Two primary modes of operation are typically employed:

- Effluent from the TFU is directed to a recirculation tank or chamber. This is referred to by Orenco Systems, Inc. as Mode 1.
- Effluent from the TFU is recycled to the anaerobic septic tank for optimized nitrogen reduction (nitrified textile filter effluent combined with STE with high carbon levels in anoxic conditions). This is referred to by Orenco Systems, Inc. as Mode 3.

The Orenco Systems, Inc. Mode 1(a) configuration, where recirculation occurs through the second compartment of the septic tank, was selected for testing at the Mines Park Test Site during this project. Orenco Systems, Inc. estimates that nitrogen reduction in Mode 1(a) of operation will typically exceed 60% with total nitrogen in the filtrate ranging between 25 and 35 mg/L (it is noted that performance will vary with loading rates). The AX20 model is designed to handle larger homes and commercial systems with peak design flows of up to 900 gallons per day. Filter sizing (such as performance) is a function of the expected typical mass loads with periodic weekly highs. A typical design HLR of approximately 100 cm/d/ft^2 (25 gpd/ft^2) with a peak design-loading rate of approximately 200 cm/d/ ft^2 (50 gpd/ft^2) is recommended by Orenco Systems, Inc. based on typical per capita flow rates and average strength wastewater characteristics expected from residential type installations (Table 2-2). Effluent quality is dependent on a number of factors, including influent characteristics and loading rates. Low to moderate loading rates produce BOD and TSS of less than 5 mg/L, while higher loading rates produce BOD and TSS in the range of 15 to 25 mg/L.

Table 2-2
Typical Design-Loading Rates for Orenco Systems, Inc. AdvanTex Filter

	Typical Residential STE Strengths			Recommended Design HLR
	Average mg/L	Weekly Peak mg/L	Rarely Exceed mg/L	
BOD_5	130	200	300	100 cm/d/ft^2 (25 gpd/ft^2) 200 cm/d/ft^2, peak (50 gpd/ft^2)
TSS	40	60	150	
Total Kjeldahl Nitrogen (TKN-N)	65	75	150	
Grease & Oil	20	25	25	

2.3.3 Membrane Bioreactor

MBRs rely on suspended growth activated sludge for biological treatment and membranes for solids separation. MBRs have been used for the treatment of municipal/industrial wastewater since the early 1970s (Hardt *et al.* 1970), and have become increasingly popular for water reuse applications or sensitive discharge environments (Yushina and Hasegawa 1994; Brindle and Stephenson 1996; Van Dijk and Roncken 1997; Trussel *et al.* 2000). While MBRs have not been evaluated for use in an OWS, they do represent a potentially promising new method of treatment for decentralized applications. Due to their potential operational complexity and cost, MBR systems might be more appropriate for cluster applications than for individual homes.

The typical MBR consists of a biological reactor with suspended biomass and solids separation provided by ultra-filtration or micro-filtration membranes. The membranes have nominal pore sizes ranging from 0.04 to 0.4 μm (Tchobanoglous *et al.* 2003). They have the ability to eliminate the solids separation functions of secondary clarification and tertiary filtration. They can operate at much higher mixed liquor suspended solids (MLSS) concentrations (Cote *et al.* 1998; Rosenberger *et al.* 2000). The high MLSS concentrations in MBRs permit higher volumetric loading rates and longer solids retention times that can result in less sludge production (Rosenberger *et al.* 2000). These advantages are accompanied by a small footprint and high effluent quality with respect to turbidity, TSS, BOD, and pathogens (Daigger *et al.* 2001). Typical water quality data from a Zenon MBR unit are presented in Table 2-3.

Table 2-3
Average Water Quality From Zenon ZeeWeed (0.04 micron) MBR

Parameter	Effluent Quality
BOD	< 2 mg/L
TSS	< 2 mg/L
Ammonium	< 1 mg/L
Total nitrogen	< 10 mg/L[a]
Total phosphorus	< 0.1 mg/L[b]
Turbidity	< 0.1 NTU

Adapted from http://www.zenon.com/MBR/membrane_bioreactor_overview.shtml
[a] with anoxic zone
[b] with coagulant addition

2.4 Installation and Startup

Installation and startup of the engineered treatment units occurred between July 1998 and March 2004. Detailed description of the treatment units and the startup operations are presented in this section.

2.4.1 Septic Tank Unit

The interception system located near the apartment building at Mines Park has been in operation since September 1998. This system is comprised of two 5,700-L (1,500-gallon) concrete septic tanks (Front Range Precast Concrete, Inc., Boulder, Colorado) followed by a 3,800-L (1,000-gallon) concrete chamber used for STE collection and sampling in previous CSM laboratory studies.

Figure 2-1 presents a schematic of the septic tanks established near the multifamily apartment complex. Gate valves near the building enable the direction of raw wastewater to the interception system or to the City of Golden sewer.

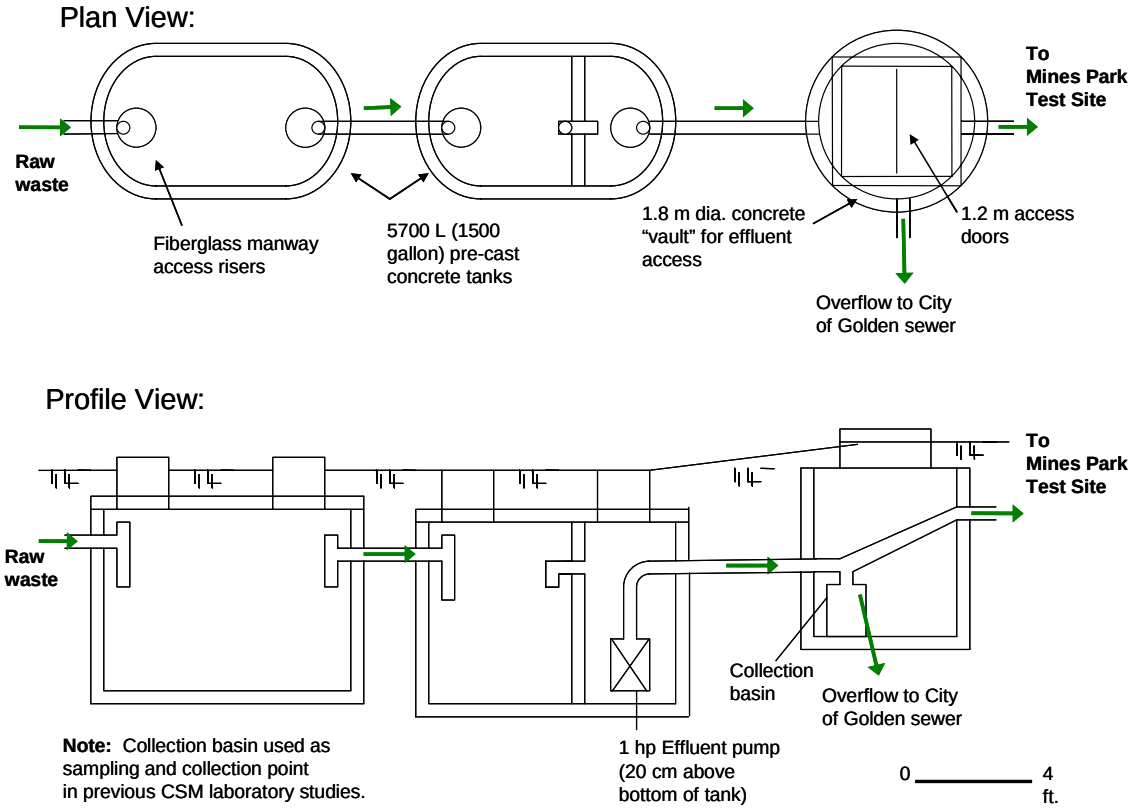


Figure 2-1
Mines Park Test Site Wastewater Interception System

First, raw wastewater from the apartment building passes through the 5,700-L single-chambered tank. Then it goes into the 5,700-L baffled, two-chambered tank. From the second tank's second chamber, a 1-hp effluent pump delivers the effluent through a 3.8 cm (1.5 in.) PVC line approximately 160 m (525 ft) up the hill to the STE holding tank located near the soil test cell area at the Mines Park Test Site. The effluent pump inlet is positioned approximately 20 cm (8 in.) off the bottom of the tank. The effluent delivery line is located 1 m (3 ft) below ground surface (bgs). A check valve in the effluent delivery line at the second chamber of the second tank prevents siphoning of the STE holding tank at the test site. Approximately 180 L (48 gallons) of STE is in the delivery line for 4 to 12 hours.

The STE holding tank located at the Mines Park Test Site began receiving STE in May 2003. Delivery of STE to the STE holding tank is done on an as-needed basis triggered by float switches located approximately 15 cm (6 in.) apart in the STE holding tank. The low level float in the STE holding tank triggers the effluent pump in the interceptor system to turn on and pump STE to the holding tank until the high-level float is triggered and the effluent pump is stopped. The duration for each "effluent pump on" cycle is approximately 15 minutes for 4 to 5 cycles per day with approximately 1,500 L (400 gallons) of STE delivered to the holding tank per day.

Based on the STE holding tank volume and STE use for both this study and the companion study, the average residence time of STE in the STE holding tank is approximately two days. The STE is then delivered, using independent pumps, to the soil test cells for the two studies and the STE serves as the influent for the TFU and MBR. Both the interceptor system and the STE holding tank are equipped with low-level and high-level alarms, as well as redundant off-floats. All overflow from both the interceptor system and the STE holding tank is returned to the City of Golden sewer.

Pumps and floats are controlled by four control panels:

- A custom Orenco Systems, Inc. VCOM control panel for delivery of STE from the interceptor system to the STE holding tank and companion study soil test cells
- An Orenco Systems, Inc. VeriComm control panel for delivery of effluent from the TFU processing tank to the TFU
- A Zenon control panel for delivery of STE to the MBR and aeration intervals
- A control panel for delivery of effluents to the test cells for this NDWRCDP study

Routine operation and maintenance of the interceptor system includes conducting routine field inspections, monitoring operations remotely using the control panels, and pumping both 1,500-gallon interceptor tanks (in April 2001, August 2004, and July 2005). Weekly inspections of the STE holding tank are also conducted, and operations are monitored remotely through the control panel. No maintenance has been required or performed on the STE holding tank.

2.4.2 Textile Filter Unit

The AdvanTex AX20 unit was delivered to the Mines Park Test Site for installation in March 2004. The second chamber of the STE holding tank at the test site serves as the processing tank (recirculation basin) for the TFU. This second chamber of the STE holding tank is isolated from the first chamber, which receives STE from the interceptor system. The baffle in the STE holding tank was sealed to prevent cross contamination or flow between the two compartments. Due to limited effluent volume available for this research, the AX20 system was scaled down in size. Only 25% of the total textile pod area (approximately 0.46 m^2 [4.3 ft^2]) receives STE. Adjacent unused sheets were removed from the pod, and ball valves were placed along the delivery line above the sheets to restrict the area of effluent spray. These ball valves regulate delivery of STE to only that area of the pod that contains textile sheets used for STE treatment.

The pod was buried (with lid exposed to ground surface) in native soil with a vent protruding approximately 45 cm (18 in.) above ground surface. This vent enables passive airflow within the filter pod (no blower is installed). The base of the pod was placed at a higher elevation than the

top of the processing tank to enable gravity return of filtrate from TFU. A 340-L (90-gallon) delivery basin holds TFU filtrate for delivery to the soil test cells. An overflow from the delivery basin is connected to the City of Golden sewer system. Figure 2-2 presents photographs and a schematic of the AX20 TFU employed in this study.

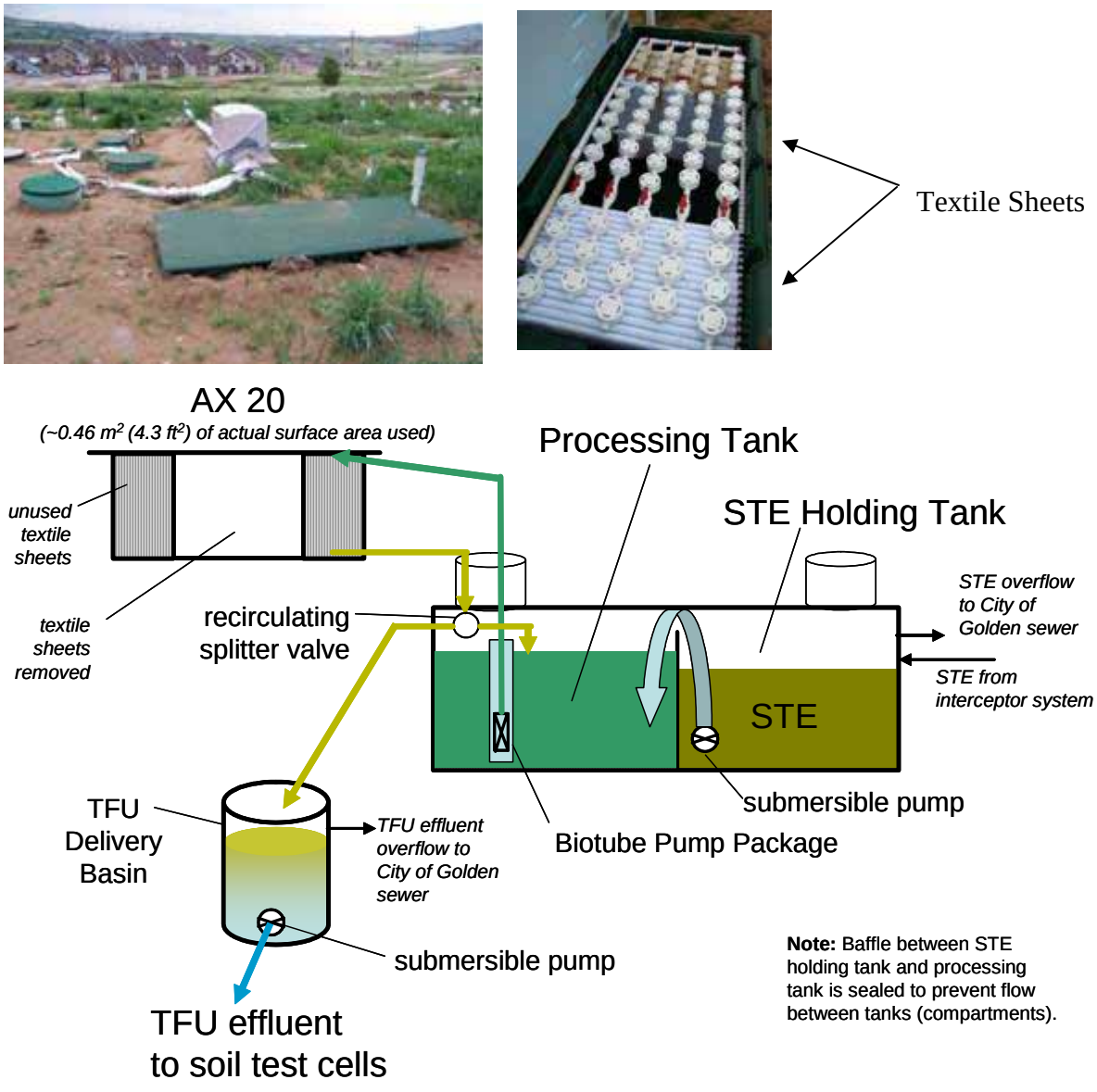


Figure 2-2
Photograph and Schematic of the Orenco Systems, Inc. AdvanTex AX20 TFU Used in This Study

The operation of the TFU at the Mines Park Test Site is as follows:

1. STE from the first chamber of the STE holding tank is delivered by a submersible pump to the second chamber of the holding tank (processing tank or recirculation basin). Approximately 250 L (65 gallons) of STE per day are delivered to the processing tank.

2. A Biotube® pump package (Orenco Systems, Inc.) placed in the processing tank pumps filtered effluent to a distribution manifold located on the top of the AdvanTex filter (only 25% of the AX20 filter unit is used for this study). Delivery of STE to the filter unit is set at a 6-to-1 recirculation ratio; STE is delivered at approximately 26 L/min (7 gal/min) with the pump cycle set at 30 seconds on and 13 minutes off.

3. STE delivered to the filter unit percolates down through the textile media and is collected at the bottom of the filter pod.

4. The STE flows (by gravity) out of the filter pod through a return line that delivers the filtrate (TFU effluent) to the recirculating splitter valve (RSV). The RSV splits the flow between the processing tank and the final discharge tank (TFU delivery basin). The RSV is critical in controlling the liquid level in the processing tank. If the level in the processing tank is low, the RSV directs 100% of the filtrate to the processing tank. If the processing tank is full, the RSV directs 100% of the filtrate to the final discharge tank. Thus, during extended periods of no flow, 100% of the filtrate remains in the processing tank.

5. To regulate delivery of the TFU effluent, the final discharge tank (a 340-L buried tank) stores TFU effluent until delivery to the soil test cells at the design HLR of 2 or 8 cm/d (0.5 or 2 gpd/ft^2). This TFU delivery basin allows for approximately 2.5 days of storage/residence time (Figure 2-2).

2.4.3 Membrane Bioreactor Unit

The HomeSpring MBR (Zenon Environmental Corporation, Canada, www.zenonenv.com) was selected for study in this project. The ZeeWeed® membrane fiber is commercially available and widely used in municipal water treatment, but it has not been evaluated for small-scale onsite treatment applications. The ZeeWeed® MBR process is a proprietary technology that consists of a suspended-growth biological reactor combined with ultra-filtration membranes. The unit used in this study is an experimental unit comprised of:

- Anoxic tank
- Aerobic tank
- ZeeWeed® membrane (0.2 micron)
- Compressor
- Peristaltic pump

All components are operated by a control panel. The MBR is located in an insulated building near the soil test cells at the Mines Park Test Site. A general schematic and photograph of the MBR used in this study is presented in Figure 2-3.

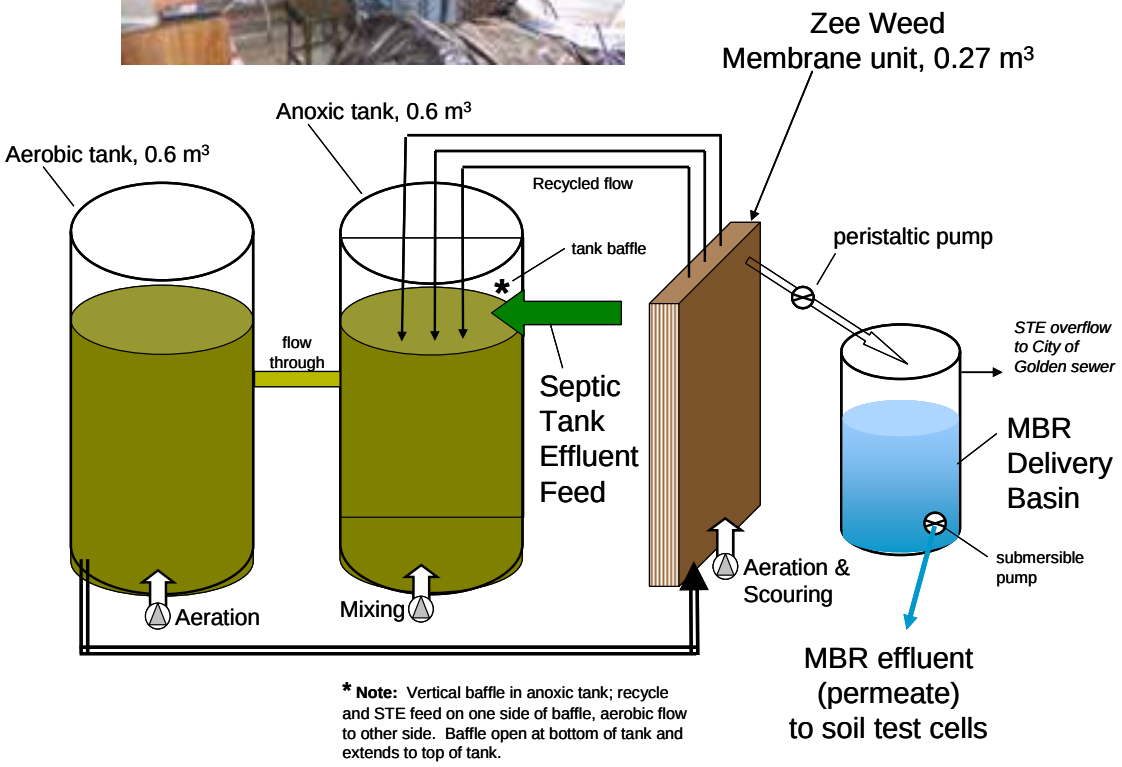


Figure 2-3
Photograph and Schematic of the MBR Unit (Zenon Corp.)

STE is delivered to the MBR by a submersible pump in the STE holding tank. Float switches located in the anoxic tank activate STE feed to the MBR (low-level float turns on submersible pump in STE holding tank and high-level float turns the submersible pump off). Flow through the MBR is controlled by both the effluent (referred to as permeate) production and the recycle of the MBR tank MLSS. The membrane is immersed in a sealed aeration tank and is in direct contact with the MLSS. A peristaltic pump generates vacuum pressure to pull fluid through the membrane and produce effluent.

As effluent is produced, the fluid levels in the aerobic and anoxic tanks are lowered, which activates the float control to the submersible pump in the STE holding tank. The effluent peristaltic pump cycles 8 minutes on and 2 minutes off to allow the membrane to relax. The STE passively flows from the anoxic tank to the aerobic tank with fluid levels in both tanks maintained at 1.38 m (4.5 ft) (above the bottom of the tank) for a system volume of 1.47 m^3

(approximately 1,470 L or 390 gallons). A baffle in the anoxic tank controls fluid movement and residence time through the tank. STE feeds into the first compartment of the anoxic tank; flow through with the aerobic tank is in the second compartment of the anoxic tank. A compressor provides intermittent mixing to the anoxic tank (required to eliminate settling in this basin) and aerobic tank, while the membrane tank is continuously aerated (see Table 2-4). Airflow is introduced at the bottom of the membrane tank allowing for aeration and scouring of the membrane fibers. Operational conditions during operation are presented in Table 2-5.

Table 2-4
Design Considerations for the MBR (Larsson and Persson 2004)

Parameter	Units	Onsite Applications
Flux management	m^3/m^2h	0.37
Aerobic tank aeration cycle	–	10 sec on; 10 sec off
Average flux rate	L/m^2, h	17 – 25
Peak flux rate (≤ 6 hours)	L/m^2, h	< 42
Transmembrane pressure	Psi	5
Maintenance / Cleaning	–	Backpulse and relax hourly
Sludge retention time	Days	10 – 15
MLSS	mg/L	≤ 10,000

Table 2-5
Aeration for the MBR

Operation	Anoxic Tank	Aerobic Tank	Membrane Tank
Function	Mixing	Aeration	Aeration, recycle, and scouring
Air time on (seconds)	6	3	Continuous
Air time off (seconds)	180	10	NA

Typical MLSS for the MBR unit are about 1,700 mg/L, which is within the low range of a conventional activated sludge process (1,500–4,000 mg/L, Metcalf & Eddy 2003). Note that there is no wasting of solids from the MBR unit due to the low BOD loading (observations indicate that the microbial growth rates are roughly equal to the decay rates). Approximately 600 L/d (158 gal/d) of MBR effluent is generated and collected in a 190-L (50-gallon) basin (MBR delivery basin) located adjacent to the MBR, with overflow returned to the City of Golden sewer. MBR effluent is applied (at the design HLRs) to the soil test cells by a submersible pump located in the MBR delivery basin. A detailed description of the MBR effluent produced is provided in Section 2.5.

2.5 Treatment Unit Performance

Assembly, installation, and startup of the TFU and MBR occurred during the fall of 2003 and were completed by the spring of 2004. Characterization of the operation and performance of the septic tank, TFU, and MBR was conducted continuously during and after system startup. During startup of both the TFU and MBR, daily observations were made of operation and effluent quality. With increased time of operation, as with the septic tank unit, weekly and biweekly observations were made to characterize the performance and effluent quality of the treatment units as described below. Table 2-6 presents the timeline for treatment unit startup and operation.

Table 2-6
Timeline for Treatment Unit Startup and Operation

Date	Day of Operation	Treatment Unit Activity
May 2003		Holding tank at Mines Park Test Site receives STE
January 20, 2004		MBR clean water operation
February 2, 2004		MBR receives STE
April 6, 2004	Day 0 (start effluent delivery to soil)	TFU receives STE
May 17–19, 2004	45–46	MBR membrane cleaning
June 11, 2004	66	MBR membrane cleaning
August 2, 2004	118	Interceptor septic tanks pumped

2.5.1 Materials and Methods

Routine operation and monitoring of the three engineered treatment units was conducted for the duration of this study. Materials and methods used for operational monitoring and effluent quality characterization are presented in this section.

2.5.1.1 Operational Monitoring

All three treatment units are monitored for general operational stability, pump and electrical component performance, and characteristic of effluent generated. The interception tanks near the multifamily housing unit where raw wastewater enters the system were pumped in August 2004. Pumping was necessary due to the accumulation of solids in the tanks, but it did not affect the overall performance of the interception system or the quality of the STE.

The TFU operation was continuous throughout this experiment with no observed pump, electrical, or other operational difficulties. Startup for the TFU commenced on April 6, 2004, the same day that effluent delivery to the soil test cells began. Operational conditions were identified that may have impacted system performance. First, the target recirculation ratio was 4-to-1, to

keep the dissolved oxygen levels in the recirculation basin low and to optimize denitrification. However, system monitoring indicates that the recirculation ratio was higher than planned (6-to-1), which may have increased the oxygen in the recirculation basin.

Next, during the first three months of operation, the top 15 to 25 cm of the filter pod were exposed to the atmosphere and may have been more susceptible to ambient temperatures. Furthermore, typical installations in Colorado have an additional 2.5 cm of insulation in the filter pod lid. A soil berm to the top of the TFU was constructed on July 1, 2004 and an additional 1" of insulation will be added to the lid.

Finally, for the duration of the study, the TFU installation did not incorporate venting measures that are typically used when the recirculation tank does not ventilate through the source building (that is, home) plumbing. Lack of those venting measures may have restricted oxygen availability. A retrofit of the system ventilation was conducted in August 2005. Routine maintenance of the TFU, including cleaning of the textile filter sheets and the delivery basin, was conducted in April 2005.

The MBR was more operationally demanding. Initial power supply problems with the STE holding tank feed pump caused discontinuous operation. The problems were eventually resolved. Membrane fouling necessitated periodic cleaning (every four months) and caused temporary system shutdowns. However, based on weekly effluent monitoring, the MBR effluent quality delivered to the field test cells remained relatively consistent throughout the study (see Section 2.5.1.2).

The MBR unit used during this study is an experimental unit (not an off-the-shelf commercially available unit), which may explain additional operational and maintenance requirements. The ZeeWeed® filter membrane is widely used for municipal water applications and development of the MBR unit for domestic wastewater is promising and merits further evaluation.

2.5.1.2 Effluent Quality Characterization

Effluent quality for each of the treatment units has been monitored for the parameters listed in Table 2-7. Weekly grab samples are taken and characterized for these conventional parameters.

2.5.1.2.1. Sample Collection

Samples from all three effluents were collected on a weekly basis to characterize effluent quality. Grab samples were collected in pre-cleaned, acid- and base-washed glass bottles (750 mL) and brought immediately to the CSM pilot lab for analysis. All analyses were performed within the specified holding times (see Table 2-7) with results recorded in lab notebooks and then entered in Microsoft Excel spreadsheets. Ten percent laboratory sample duplicates were analyzed in addition to spike/control samples.

Effluent sample collection points were located in the STE holding tank, the TFU delivery basin, and the MBR delivery basin. Samples were also taken from the TFU processing tank. All effluent quality sample results reported are from grab samples. Results for composite sampling are presented in Section 2.5.2.2.3.

Table 2-7
Water Quality Parameters, Methods Employed, Minimum Detection Limits, and Storage Requirements

Parameter	Method	Estimated Minimum Detection Limits	Preservation/Storage
pH	Electrode Thermo Orion 91–55 Probe, APHA (1998)	0.1	Analyze immediately
Total alkalinity	Titration method 8203, APHA (1998)	0.2 mg/L as $CaCO_3$ [a] 2 mg/L as $CaCO_3$ [b]	4 °C, 24 hrs
TOC/DOC	Shimadzu TOC Analyzer; non-purgeable organic carbon -or- Sievers 800 Portable TOC Analyzer	1 mg/L 0.05 mg/L	4 °C, 28 days
COD	Reactor digestion method 8000, HACH (1998)	0.2 mg/L [a] 3 mg/L [b]	4 °C, ASAP or H_2SO_4 to pH<2, 28 days
BOD_5	DO uptake method 5210, APHA (1998)	0.3 mg/L [a] 30 mg/L [b]	4 °C, 6 hrs
Total phosphorus	Acid persulfate digestion method 8190, HACH (1998) (US EPA approved)	0.06 mg-PO_4/L	4 °C, ASAP or H_2SO_4 to pH<2, 28 days
Total nitrogen	Persulfate digestion method 10071, HACH (1998)[c]	2 mg-N/L	4 °C, ASAP or H_2SO_4 to pH<2, 28 days
Nitrate nitrogen	Chromotropic acid method 10020, HACH (1998)	0.2 mg-N/L	4 °C, ASAP or H_2SO_4 to pH<2, 28 days
Ammonium nitrogen	Salicylate method 10031, HACH (1998)	0.031 mg-N/L [a] 0.6 mg-N/L [b]	4 °C, ASAP or H_2SO_4 to pH<2, 28 days
TS	Oven dried at 103–105° method 2540B, APHA (1998)	5 mg/L	4 °C, 24 hrs
TSS	Oven dried at 103–105° method 2540D, APHA (1998)	5 mg/L	4 °C, 24 hrs
Fecal coliform	Membrane filtration method 9222D, APHA (1998)	1 cfu / 100 mL	4 °C, 24 hrs

[a] Estimated minimum detection limit for lysimeter and MBR effluent samples only. Lower detection limits also applicable to TFU effluent after nitrification began.
[b] Estimated minimum detection limit for STE, TFU processing tank, and TFU effluent samples prior to nitrification.
[c] Total N analysis converts all forms of nitrogen to nitrate and includes organic nitrogen (approximately 95–100% recovered by this method), nitrate, nitrite, and ammonium.

2.5.1.2.2. Sample Analysis

Conventional Parameters. Characterization of effluent quality for all treatment units occurred weekly (with a higher frequency of characterization during treatment unit startup). Grab samples were collected and transported to the CSM pilot laboratory where analysis was completed within 24 hours, or samples were properly preserved until analysis could take place. Results were recorded in a logbook along with additional information such as date, time, and sample identification. Table 2-7 presents a summary of the water quality analyses conducted, the methods employed, minimum detection limits, and necessary preservation and storage.

Non-Conventional Parameters. Non-conventional parameter analyses included size exclusion chromatography (SEC), dissolved organic carbon (DOC), ultraviolet absorbance (UVA), and bulk organic matter fractionation.

For SEC analysis, the apparent molecular weight distribution of organic carbon in aqueous samples was determined using a LC-600 Liquid Chromatograph (Shimadzu) coupled with an SPD-10 A VP UV-VIS Detector (Shimadzu) at 254 nm and a Sievers 800 Turbo Portable total organic carbon (TOC) analyzer (Her *et al.* 2002). Samples filtered (0.45 μm), diluted to approximately 5 mg/L DOC with ultra-pure water (pure through filtration and ion exchange) and adjusted to 5 mS/cm conductivity prior to analysis.

DOC and UVA of aqueous samples were analyzed after 0.45 μm filtration (Whatman) on a Sievers 800 TOC analyzer and a 8740 UV-VIS scanning spectrophotometer (Nicolet) at 254 nm. The specific UV absorbance (SUVA) was determined as the ratio of UVA (1/m) to DOC (mg/L).

The composition of organic matter in the wastewater source was characterized by bulk fractionation. For this test, 300 mL of the filtered sample (0.45 μm cellulose acetate filter, Whatman) were passed through a pre-cleaned, ultra-filtration membrane (Spectrapor, 6,000 Dalton nominal molecular weight cut-off) during constant stirring under a feed water pressure of 40 psi. The membrane was pre-cleaned according to the manufacturer's recommendations and rinsed with ultra-pure water until UV absorbance at 254 nm indicated no further contamination prior to sample application.

After 250–280 mL of the sample volume was processed through the membrane, the feed solution was diluted by adding 100 mL of ultra-pure water. Ultra-filtration was terminated when the feed water was reduced to 20 mL. A mass balance was conducted over the filtration step, taking volumes and DOC concentrations of feed water and permeates into account in order to determine the contribution of colloidal organic matter. Colloidal organic matter is herein operationally defined as the size fraction between 6,000 Dalton and 0.45 μm.

DOC (smaller than 6,000 Dalton) was further characterized as hydrophilic and hydrophobic by processing 100 mL of the ultra-filtration permeate over an XAD-8 resin according to the procedure established by Leenheer *et al.* (2000). For the isolation of hydrophilic organic matter (HPI) and hydrophobic acids (HPO-A), 100 mL of the sample (adjusted to pH 2 using concentrated hydrochloric acid equilibrated to room temperature (22 °C)) were passed through XAD-8 resin (Rohm & Haas, Philadelphia, Pennsylvania) with a capacity factor of $k = 4$.

HPI was collected as the non-absorbable fraction of the sample. HPO-A were subsequently recovered from the resin by 0.1 N sodium hydroxide desorption.

To close the organic carbon mass balance, the fraction of hydrophobic neutrals (HPO-N) was quantified as the portion not recovered by the XAD-8 resin base rinse:

DOC minus HPI minus HPO-A is assumed to equal HPO-N

A mass balance was conducted over the fractionation using DOC and UVA_{254} measurements. The resin was pre-cleaned between sample applications with a 75% acetonitrile rinse (3–4 bed volumes) followed by an ultra-pure water wash until DOC concentrations of the column effluent remained below 0.4 mg/L. The resin was then stored in 0.1 N HCl until the next sample application.

2.5.1.2.3 Data Analysis

Data results were graphed as time series and, when applicable, as percent removal. In some cases, histograms or cumulative frequency distribution (CFD) analysis was conducted using Microsoft Excel software. Percent removal was calculated on a concentration basis comparing the concentration of a constituent in the STE to the concentrations found in the TFU or MBR effluents.

Note that during the tracer test conducted in July 2004, interference with several analysis methods was observed due to bromide. Specifically, elevated bromide concentrations unexpectedly interfered with TS, TSS, COD, DOC, total nitrogen, and nitrate analyses. A series of laboratory tests was completed to establish a correlation between the expected range of bromide concentrations and the expected range of concentrations for the parameter of interest.

Table 2-8 summarizes the results of the laboratory testing. Due to the limited number of affected sampling points, the remaining uncertainty of the corrected data, and the potential for individual data points to suggest a level of accuracy not measured, no attempt was made to “correct” the data based on the laboratory testing. Rather, the data was evaluated within the dataset as measured.

The most significant impact was observed in the TS, TSS, and nitrate analyses. These data were removed from the dataset. Total nitrogen results (day 93 to 128) are expected to be 5 to 10% low, while COD results for the same time are expected to be up to 10% high. While the bias to the total nitrogen and COD results effect the certainty of the individual data point, no significant impact was observed related to the overall effluent quality and estimated removal efficiencies. There was no impact on DOC results, because corrective measures were taken for all samples, including sample dilution to bring bromide concentrations within acceptable ranges and use of the Shimadzu for analysis, which was less sensitive to bromide interference.

Table 2-8
Summary of Bromide Tracer Interferences on Effluent Analyses

Parameter	Bias	Correlation	Comments
TS and TSS	Positive	No correlation established, but measured values are 1+ orders of magnitude high.	Unable to correlate effects, data screened. Number of data points removed: STE (3), TFU (4), MBR (3).
COD	Positive	No interference at <100 mg-Br/L; 110% recovery at 100–1000 mg-Br/L; 200+ % recovery at 1000+ mg-Br/L.	Screened data if Br > 1000. Number of data points removed: STE (1), TFU (0), MBR (2).
DOC	Positive	No interference observed with Shimadzu. No correlation established with Sievers.	All samples re-analyzed with appropriate measures to eliminate interference (sample dilution and selection of instrument for analysis). No data points removed.
Total nitrogen	Negative	95% recovery at 10 mg-Br/L; 90% recovery at 100 mg-Br/L; 83% recovery at 1000 mg-Br/L.	Reported values for July 2004 (day 93 to 128) are low and may suggest lower than expected total nitrogen concentrations. No data points removed.
Nitrate	Negative	75% recovery at 10 mg-Br/L; <25% recovery at 100 mg-Br/L.	Error in recovered concentration is significant and prevents accurate estimation of sample concentration. Number of data points removed: STE (0), TFU (4), MBR (5).

Duplicate field samples were collected with the routine samples at the same location in immediate succession with a regular/routine sample. The identification numbers and locations of the duplicate and regular samples were clearly indicated in the logbook. Duplicate samples undergo the same laboratory analyses as regular samples. Laboratory quality assurance/quality control (QA/QC) procedures included laboratory duplicates and initial and continuing calibration checks following established protocols. Laboratory equipment calibration methods and QA/QC sample frequency were described in the project quality assurance project plan. Percent difference and relative percent difference (RPD) were calculated for each duplicate analysis (APHA 1998).

2.5.2 Results

This section provides results of the routine operational monitoring and effluent quality characterization.

2.5.2.1 Operation and Maintenance

The septic tank, currently the most commonly employed treatment unit for OWSs, provided consistent effluent quality with minimal maintenance demands. The TFU has operated continuously since startup with no demands for maintenance. These two treatment units were inspected weekly. There were no required adjustments of flow rates to the units, cleaning of parts, or replacement of fittings or pumps. The MBR was started up in January 2004 with three

phases of operation to establish biological process and shakedown operations (Larsson and Persson 2004) (Table 2-9).

During Phase 1 (initial startup), 80 L of activated sludge from the City of Golden wastewater treatment plant was added to the MBR unit (on February 4, 2004 and February 16, 2004). The STE feed rate ranged from 1.54 L/h to 24 L/h due to a power outage and STE feed pump failure that limited STE feed to the MBR during startup for about 10 days. During that time, the recirculation rate varied between 108 and 273 L/h. After adding activated sludge on February 16, 2004, it took approximately three weeks for the HLR and MLSS to reach steady state and nitrification was over 95% (Larsson and Persson 2004). During Phase 2 (baseline operation), the STE feed rate remained constant, while the recirculation rate varied. Phase 3 (high load testing), consisted of a high MLSS and COD loading test to evaluate effluent quality changes with respect to performance and removal of nutrients and organic compounds. From the completion of the three start-up phases, the MBR has been running at a constant STE feed of 24 L/h with an MBR recirculation rate of 235 L/h.

Table 2-9
Summary of MBR Operations

	Duration	Ave. STE Feed Rate, L/h (range)	Ave. Flux, L/m^2/h (range)	Ave. MBR Recirc. Rate L/h (range)	Ave. HRT, h (range)	Ave. MLSS, mg/L (range)	Ave. Feed COD, mg/L (range)
Phase 1, Startup and shakedown	01/12/04 to 03/05/04	21.1 (0–203.3)	1.87 (0–3.8)	237.4 (0–488.2)	33 (26–39.8)	not measured	301 (131–408)
Phase 2, Baseline operation	03/05/04 to 05/03/04	25.6 (22.2–28.2)	2.7 (2.34–3.04)	226.7 (181.4–260.0)	33 (31–39)	1700 (1500–1900)	310 (245–409)
Phase 3, High load testing	04/18/04 to 04/24/04	22.7 (21.9–23.2)	2.57 (2.54–2.59)	245.8 (233.3–254.2)	37 (36–38.5)	2700 (2023–3247)	1100 (500–1900)
Study operations	04/07/04 to present	23.2 (20.3–26.1)	2.7 (1.96–2.94)	170 (146–183)	61.6 (54.8–70.4)	2573 (2080–2793)	220 (193–241)

2.5.2.2 Effluent Quality

Characterization of the effluent quality produced from each engineered treatment unit included assessment of water quality, conventional pollutants, organic carbon, and daily and weekly variations. Results from this monitoring are presented in this section.

2.5.2.2.1 Water Quality and Conventional Pollutants

After six months of operation, a large dataset of effluent quality parameters was accrued. A summary of the concentrations of conventional pollutants in each of the three effluents is shown in Table 2-10. Time series data indicate changes in effluent quality during this study with the first day of effluent delivery to the soil test cells on April 6, 2004 (see Table 2-6).

Table 2-10
Septic Tank, TFU, and MBR Effluent Quality Results (April–October 2004)

STE	Average	Median	SD	n	CV	Range
pH				33		6.75–7.61
Alkalinity (mg-$CaCO_3$/L)	272	266	43.5	31	0.16	192–410
TS (mg/L)	367	390	75.2	23	0.21	185–505
TSS (mg/L)	23	20	15.2	25	0.67	n.d.–50
COD (mg/L)	256	257	47.4	32	0.19	176–343
DOC (mg-C/L)	34.2	34.6	11.1	13	0.32	14.1–54.8
$cBOD_5$ (mg/L)	170	180	54.9	13	0.32	99–237
Total N (mg-N/L)	62.4	63.0	10.2	27	0.16	40.0–81.0
Total P (mg-PO_4/L)	20.9	23.5	8.4	27	0.40	0.1–32.1
Ammonium (mg-N/L)	58.0	57.2	8.6	29	0.15	37.2–75.6
NO_3 (mg-N/L)	2.5	2.4	1.3	31	0.50	0.6–7.3
Fecal Coliform (cfu/100mL)	5.1×10^4			27		1.2×10^3–3.7×10^5
TFU (<51 days of delivery)	**Average**	**Median**	**SD**	**n**	**CV**	**Range**
pH				12		6.46–7.77
Alkalinity (mg-$CaCO_3$/L)	228	254	73.2	12	0.32	40–282
TS (mg/L)	337	380	131.2	9	0.39	n.d.–440
TSS (mg/L)	4	n.d.	4.0	10	1.05	n.d.–15
COD (mg/L)	107	94	51.4	12	0.480	40–216
DOC (mg-C/L)	9.7	9.5	2.2	8	0.22	7–14.1
$cBOD_5$ (mg/L)				0		
Total N (mg-N/L)	57.3	56.3	9.0	8	0.16	44–72
Total P (mg- PO_4/L)	18.7	21.2	7.3	12	0.39	0.1–23.7
Ammonium (mg-N/L)	51.8	54	17.6	8	0.34	16.0–75.0
NO_3 (mg-N/L)	1.4	0.8	1.4	11	1.043	0.5–5.3
Fecal Coliform (cfu/100mL)	3.7×10^4			12		n.d.–3.5×10^5

n=number of samples, SD=standard deviation, CV=coefficient of variation, FC ave. is geometric mean

Table 2-10
Septic Tank, TFU, and MBR Effluent Quality Results (April–October 2004) (Cont.)

TFU (>51 days of delivery)	Average	Median	SD	n	CV	Range
pH				20		4.26–7.48
Alkalinity (mg-$CaCO_3$/L)	53.6	26.0	70.9	12	1.32	1.0–250
TS (mg/L)	350	365	153.9	11	0.44	115–725
TSS (mg/L)	6	n.d.	6.6	12	1.05	n.d.–20
COD (mg/L)	56.9	43.3	32.5	20	0.57	20.2–124
DOC (mg-C/L)	11.5	10.0	4.01	4	0.35	8.5–17.4
$cBOD_5$ (mg/L)	4.9	4.1	2.0	9	0.41	1.9–8.6
Total N (mg-N/L)	39.5	37.4	12.9	18	0.33	12.8–66.0
Total P (mg-PO_4/L)	19.0	22.1	8.4	15	0.44	0.15–28.8
Ammonium (mg-N/L)	11.2	9.2	8.6	16	0.77	2.7–27.2
NO_3 (mg-N/L)	16.0	15.3	7.86	15	0.49	4.1–26.8
Fecal Coliform (cfu/100mL)	1.9×10^2			13	n.d.	2.0×10^4
MBR	**Average**	**Median**	**SD**	**n**	**CV**	**Range**
pH				32		3.50–8.99
Alkalinity (mg-$CaCO_3$/L)	30.4	7.9	48.8	27	1.61	n.d.–215
TS (mg/L)	395	378	89.7	22	0.23	290–735
TSS (mg/L)	5	n.d.	7.8	24	1.55	n.d.–40
COD (mg/L)	14	13.2	7.8	29	0.56	0.5–38
DOC (mg-C/L)	6.2	6.2	1.39	10	0.23	4.0–9.1
$cBOD_5$ (mg/L)	1.7	1.3	1.83	9	1.08	n.d.–6.1
Total N (mg-N/L)	26.5	24.0	11.3	19	0.43	10.8–52.8
Total P (mg- PO_4/L)	19.4	21.4	9.9	24	0.51	0.10–38.8
Ammonium (mg-N/L N)	0.67	n.d.	2.1	27	3.16	n.d.–10.7
NO_3 (mg-N/L)	21.3	21.8	6.8	23	0.32	2.3–30.8
Fecal Coliform (cfu/100mL)	n.d.			27		n.d.

n=number of samples, SD=standard deviation, CV=coefficient of variation, FC ave. is geometric mean

Time series data for TSS are shown in Figure 2-4. TSS varied in all effluents, with no apparent reduction, as a result of continued operation for the TFU or MBR compared to STE. Variability in TSS results for STE and the TFU effluent may be attributed to the sample collection method. It was not always possible to catch the in-stream effluent as it flowed into the STE holding tank or TFU delivery basin. This sample collection method may result in higher than expected TS and TSS results as particles at the effluent surface may be collected within the sample.

Time series data for COD are shown in Figure 2-5. Compared to STE, a significant reduction occurred in the TFU and MBR. Average COD concentrations during the last four months of operation (June–October 2004) were 248 mg/L, 56.9 mg/L, and 22.3 mg/L in the STE, TFU, and MBR effluents, respectively.

Figure 2-6 presents average $cBOD_5$ and DOC values for the effluents revealing the significant reduction in $cBOD_5$ and DOC levels that occurred in the TFU and MBR. The low sample number for carbonaceous BOD ($cBOD_5$) analysis is because reliable data for $cBOD_5$ were not collected until after approximately four months of system operation. Problems with the reliability of the DO probe and meter were overcome after the first four months of sample collection.

Figure 2-7 presents nitrate and ammonium values for the effluents generated by the TFU and MBR. Operation of the TFU coincided with the start of effluent delivery to the soil; therefore, a period of operation occurred before nitrification was observed in the TFU and nitrified effluent was applied to the soil test cells. Based on nitrate and ammonium concentrations in the TFU effluent, nitrification began between 44 and 51 days of operation with consistent nitrification observed after approximately three months of operation. The MBR began receiving STE in mid-January 2004. By the time of effluent dosing to the soil test cells, some level of nitrification had already commenced. (Table 2-10; see also Table 2-9). This can be seen in Figure 2-7. The TFU and MBR also demonstrated the ability to denitrify. Average total nitrogen concentrations during the last four months of operation (June–October 2004) were 60.8 mg-N/L, 39.5 mg-N/L, and 24.4 mg-N/L in the STE, TFU, and MBR effluents, respectively. The treatment efficiency of the TFU and MBR is presented in Section 2.5.3.2.

Alkalinity and pH data are shown in Figure 2-8. As expected, the STE pH and alkalinity remained constant at 7 and approximately 270 mg-$CaCO_3$/L, respectively. Alkalinity and pH values in the TFU effluent decreased with operation time as a result of the consumption of alkalinity during the nitrification process. These trends are consistent with nitrogen conversion trends previously discussed. They support initial TFU nitrification within 44 to 51 days and consistent levels of nitrification after approximately three months of operation. Based on the low pH and alkalinity values observed during July and August (roughly days 100 to 150), nitrification was likely limited in the TFU. No alkalinity augmentation was employed.

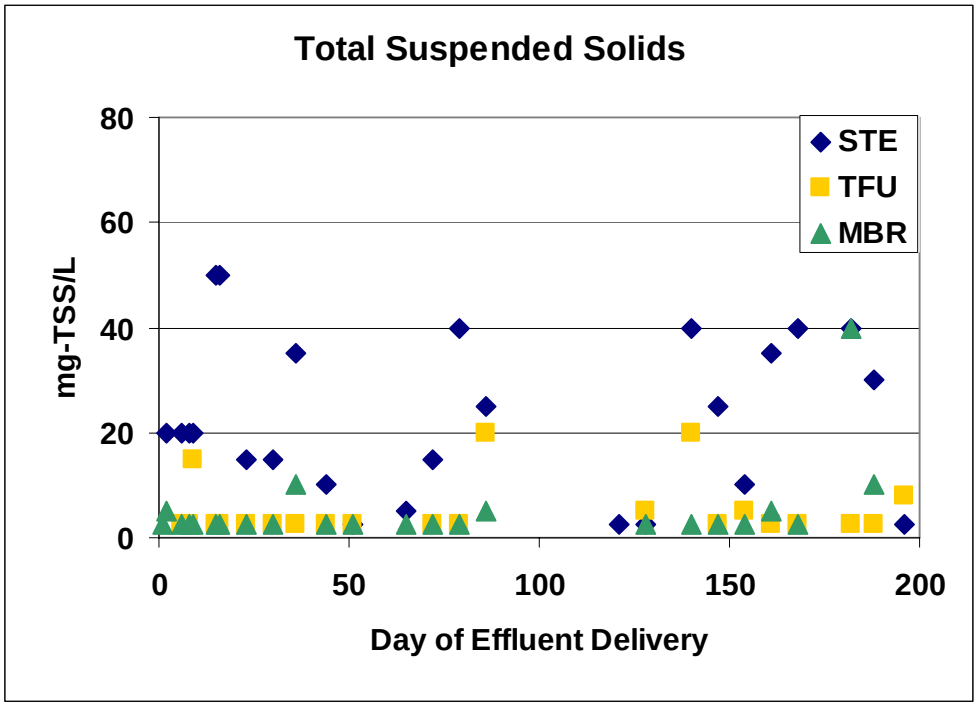


Figure 2-4
TSS Concentrations in Septic Tank, TFU, and MBR Effluents

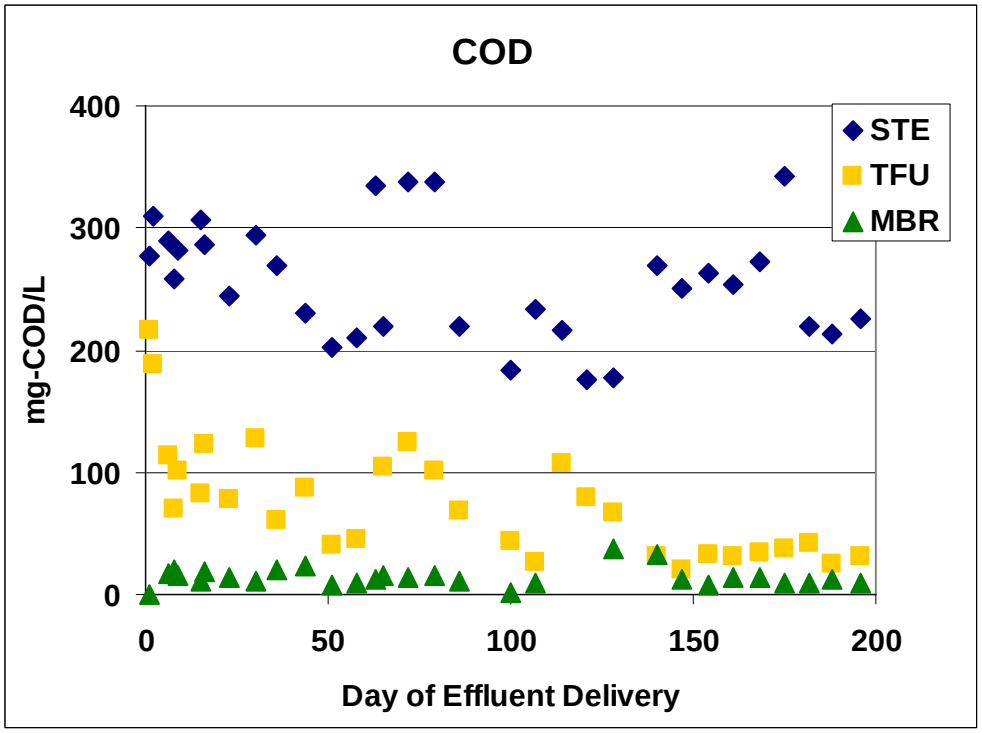


Figure 2-5
COD Concentrations in Septic Tank, TFU, and MBR Effluents

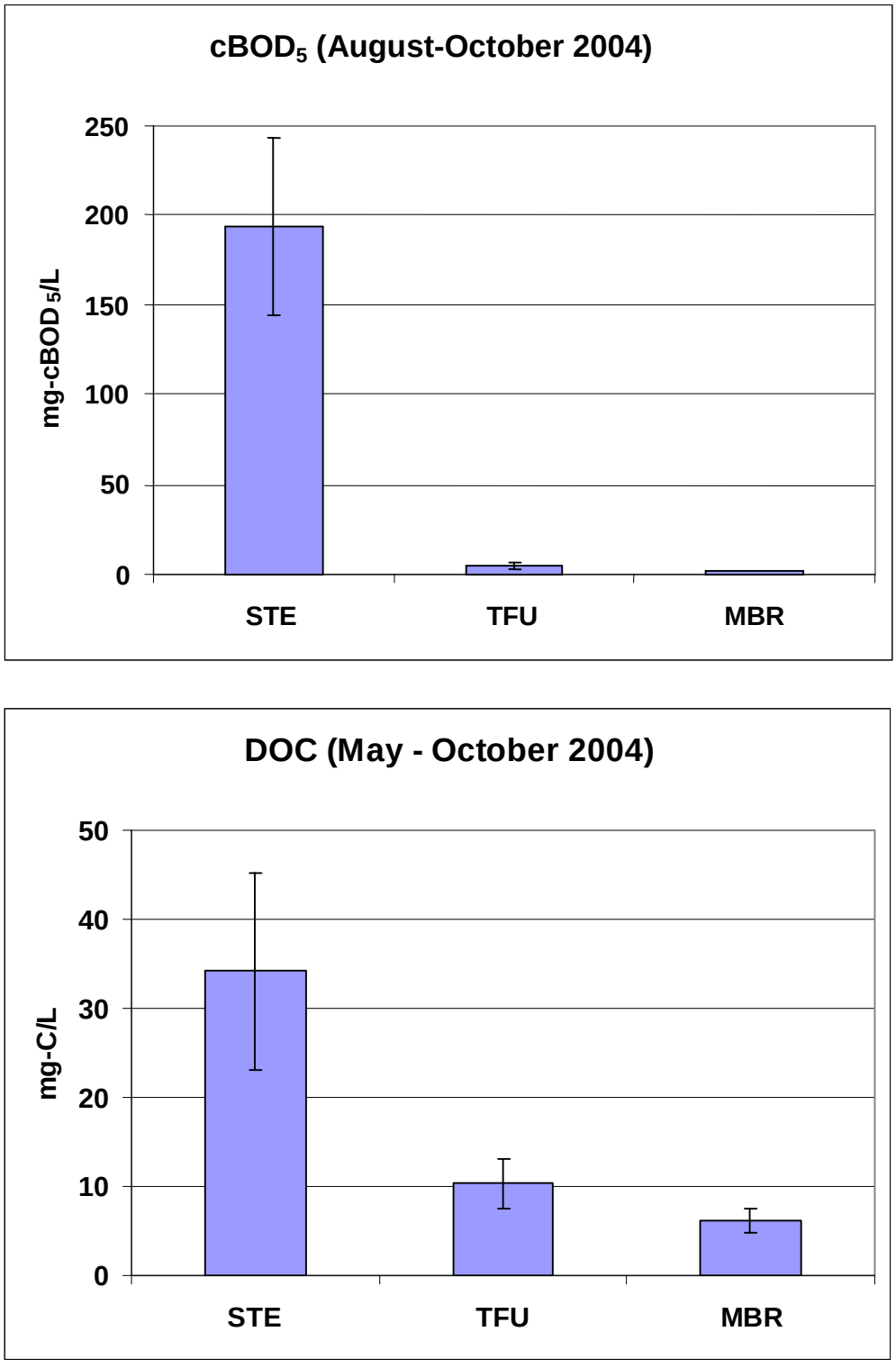


Figure 2-6
Average $cBOD_5$ and DOC Concentrations in Septic Tank, TFU, and MBR Effluents[1]

[1] +/- one standard deviation

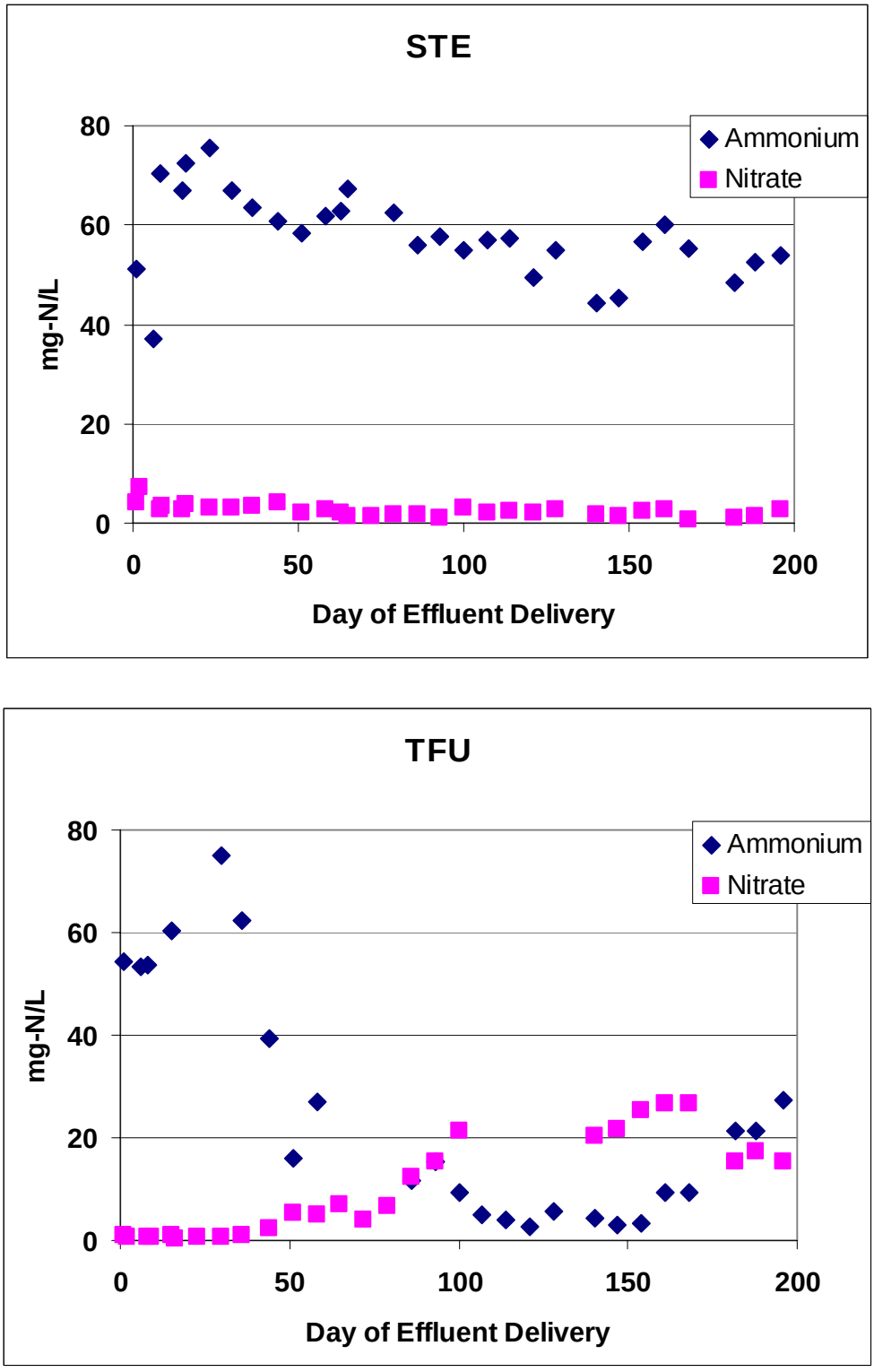


Figure 2-7
Ammonium and Nitrate Concentrations in Septic Tank, TFU, and MBR Effluents

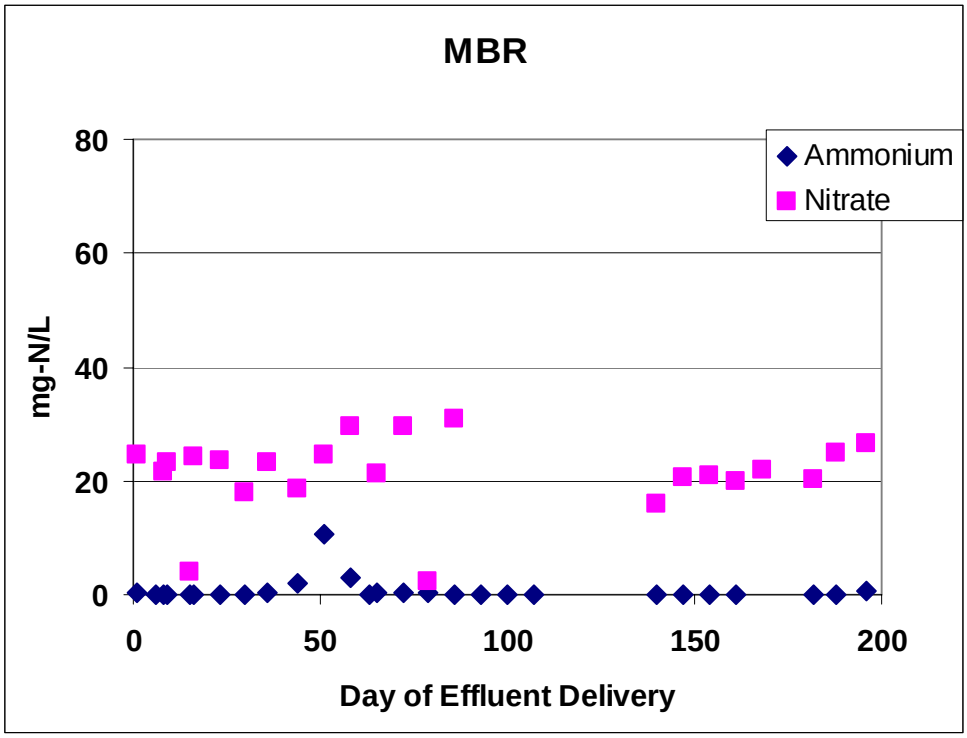
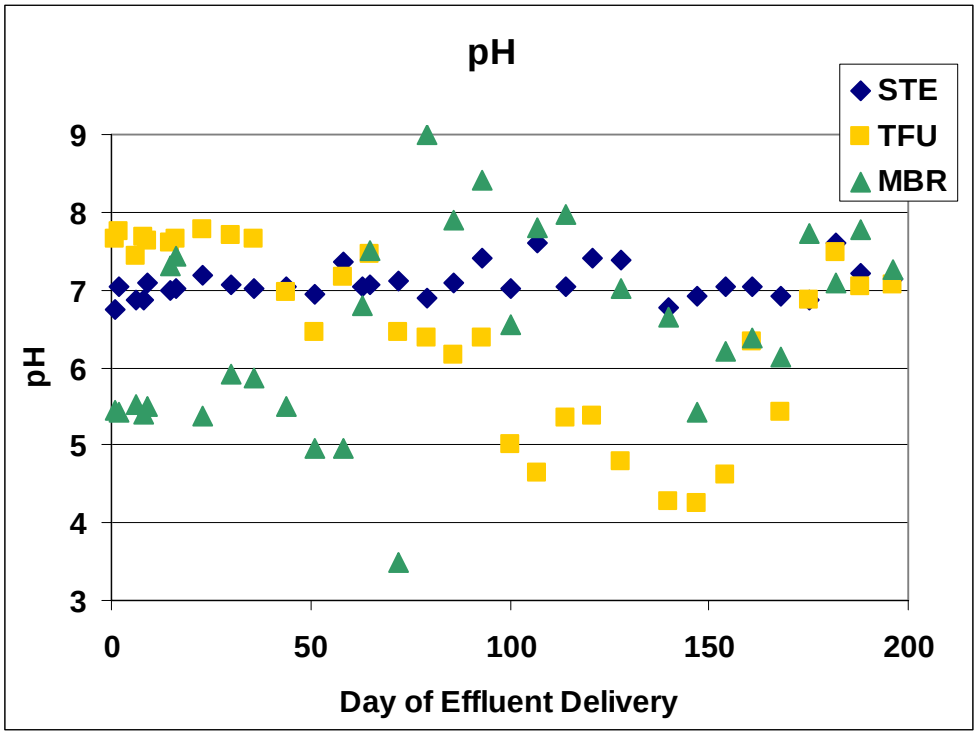


Figure 2-7
Ammonium and Nitrate Concentrations in Septic Tank, TFU, and MBR Effluents (Cont.)

Higher fluctuations in the MBR pH were observed over the operational period and reflect cyclic levels of nitrification during operation and MBR maintenance operations (Figure 2-8). During nitrification, alkalinity is consumed and hydrogen is released, causing the pH to decrease. As the pH continues to decrease, the nitrifying bacteria are inhibited and the ammonium in the system builds up, causing the pH to increase and nitrification is again supported. These cycles are not sufficient to stop nitrification, as shown by constant nitrate concentrations and low alkalinity values in the MBR effluent. Fluctuations in pH and alkalinity in the MBR effluent are also attributed to samples collected immediately after membrane cleaning (which uses a citric acid solution) as well as alkalinity augmentation to the system, which began in mid-June to improve MBR nitrification.



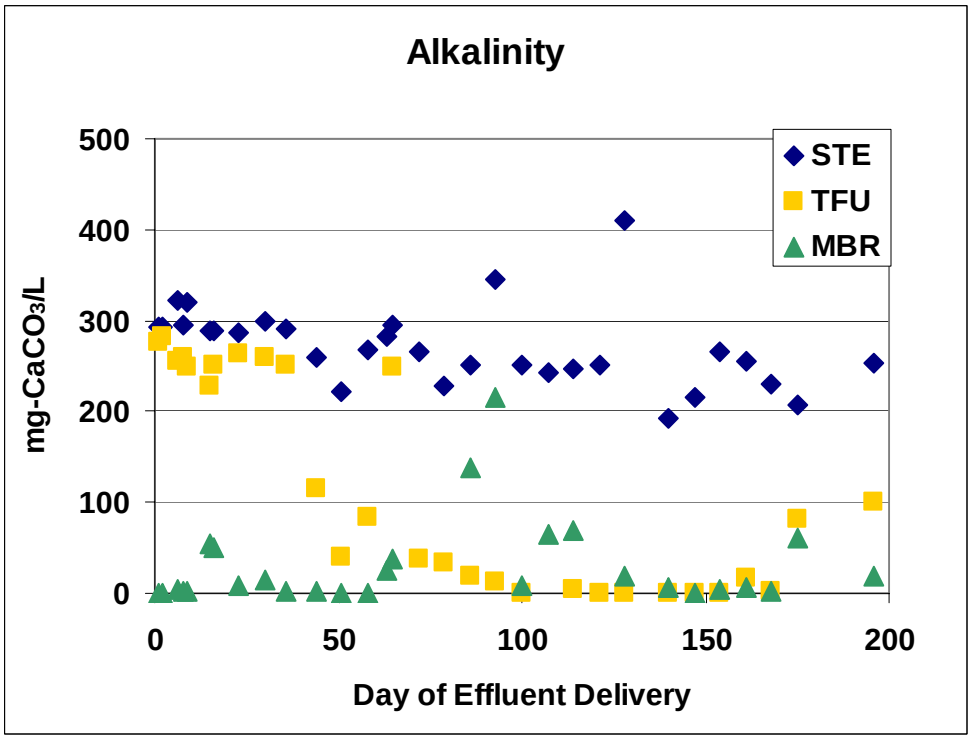


Figure 2-8
pH and Alkalinity Concentrations in Septic Tank, TFU, and MBR Effluents

The TFU and the MBR are not designed for removal of phosphorus, and little phosphorus removal was observed through these treatment units. Figure 2-9 presents the effluent concentration of total phosphorus in each of the three treatment units. While there was some variability of phosphorus concentrations, average values during six months of operation were 20.9 mg-PO_4/L, 18.9 mg-PO_4/L, and 19.5 mg-PO_4/L in the STE, TFU, and MBR systems, respectively.

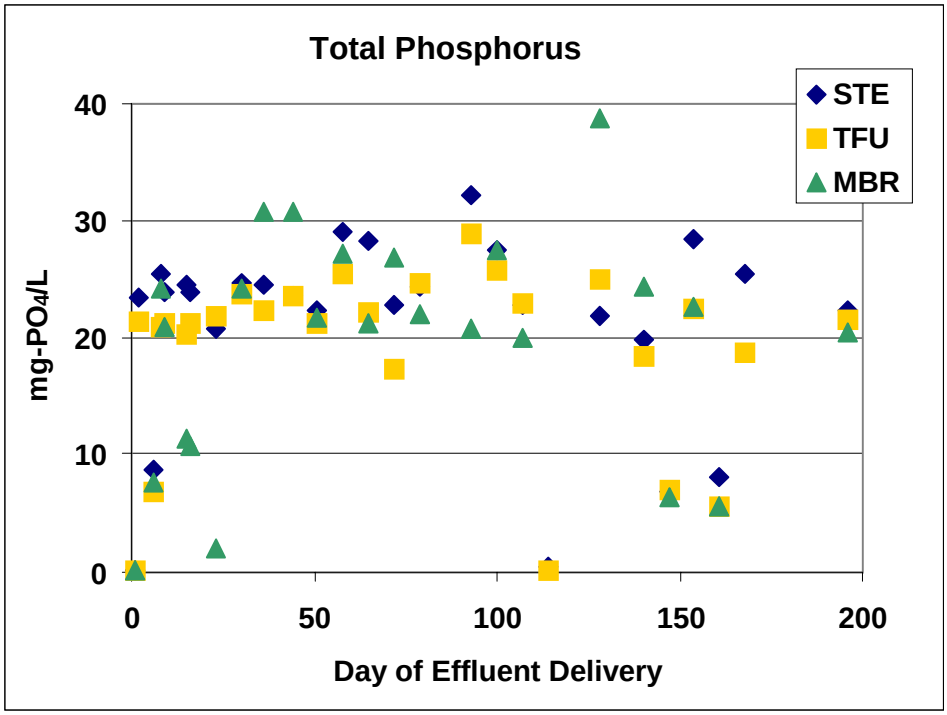


Figure 2-9
Total Phosphorus Concentrations in Septic Tank, TFU, and MBR Effluents

Fecal coliform bacteria were found at consistently high levels in the STE (around 10^3–10^5 cfu/100 mL) as shown in Figure 2-10. Initial high concentrations of fecal coliform bacteria were also observed in the TFU (geometric mean value of approximately 3.7×10^4 cfu/100 mL), but after about 50 days of operation, significant removal of fecal coliform bacteria was observed (96% removal observed as presented in Section 2.5.3.2).

The MBR is based on solids separation/ultra-filtration technologies, and under normal operation, breakthrough of bacteria will not occur. As revealed in Figure 2-10, on two occasions fecal coliform bacteria were detected in the MBR effluent delivery basin. It was speculated that the MBR effluent delivery basin or sample container was contaminated rather than a failure of the membrane. A rigorous cleaning of the delivery basin and the effluent pump tubing was conducted. In addition, the tubing placement was altered and a cover was placed on the basin. Since these changes were implemented, no fecal coliform bacteria have been detected in the MBR effluent.

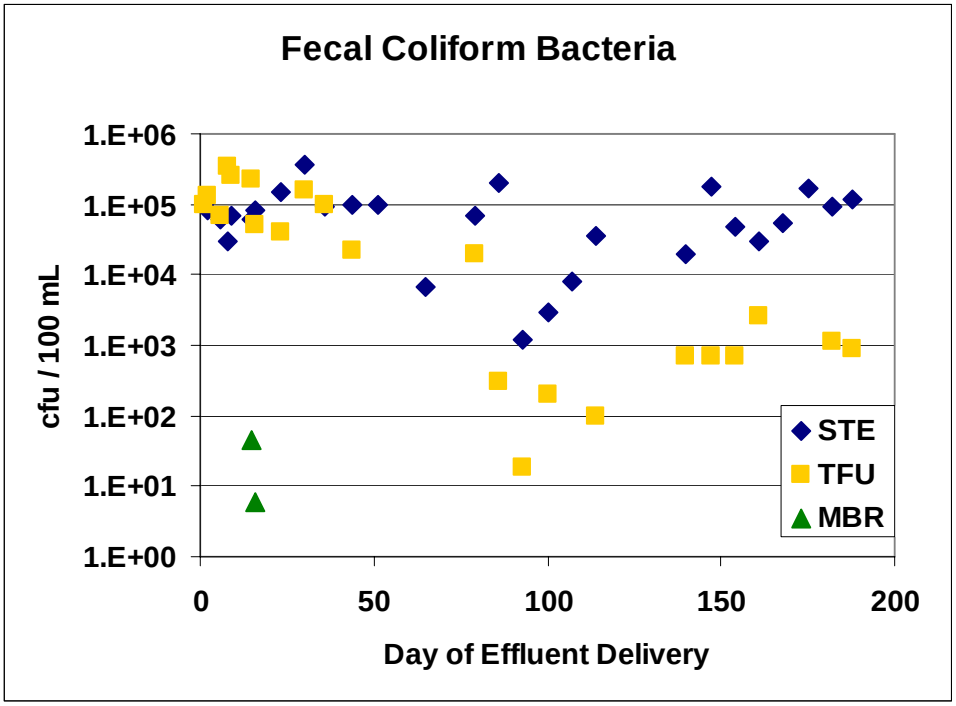


Figure 2-10
Fecal Coliform Bacteria Concentrations in Septic Tank, TFU, and MBR Effluents[2]

2.5.2.2.2 Organic Carbon Characterization

To compare the apparent molecular weight distribution of the organic matter in the different wastewater effluents, SEC analyses were completed with samples of the STE and the TFU and MBR effluents (see Figure 2-11 and Figure 2-12). SEC describes bulk organic carbon in aqueous samples in the form of a fingerprint that is defined by characteristic peaks. For treated wastewater effluents, these peaks were identified as:

- High molecular weight compounds with DOC detection but low UVA (mainly polysaccharides) (1,800 seconds [s])
- Humic substance-like material with DOC response and high UVA (2,950 s)
- Building blocks of humic substances (3,250 s)
- Low molecular weight acids (3,800–4,000 s) (Huber and Frimmel 1996)

Two samples of the STE were analyzed by SEC; one collected in November 2003 and a second collected in May 2004. The results of SEC analysis of the STE collected in November 2003 show a strikingly different molecular weight distribution than typically observed for secondary or tertiary treated municipal domestic effluents (Figure 2-11) (Rauch 2005; Rauch and Drewes 2005). The STE is characterized by saturated (carbon molecules containing only single bonds) high molecular weight compounds (1,800 s) and saturated small molecular weight compounds

[2] No fecal coliform bacteria were detected in the MBR effluent

(4,200 s). A high peak is observed at 3,500 s, which appears between the typical detention times of fulvic acids (building blocks) and small molecular weight acids. Thus, it is not likely to represent either one of them. Also, this peak is not accompanied by high UVA readings typical for fulvic acids.

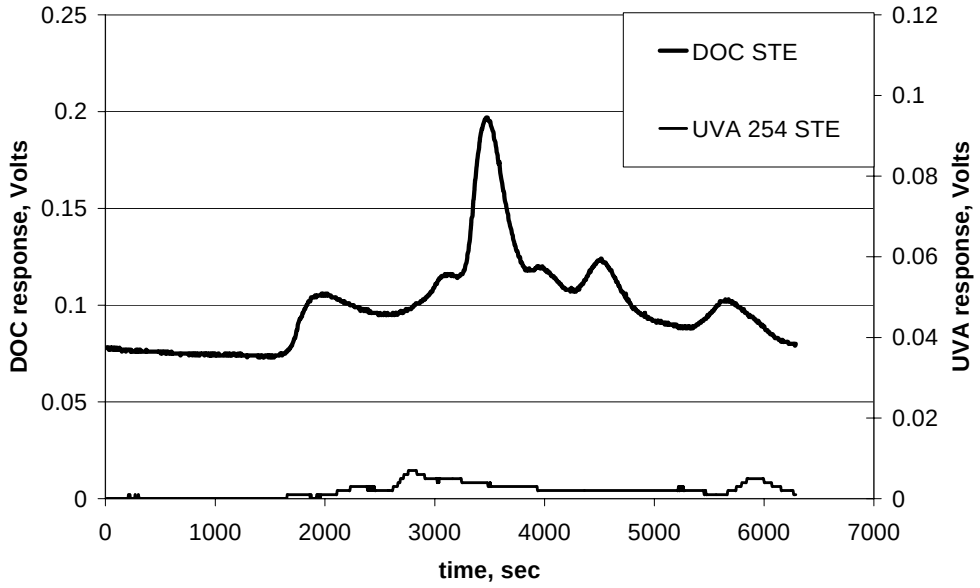


Figure 2-11
SEC Fingerprint of STE Collected in November 2003

Around 5,800 s detention time is another peak not generally observed for secondary or tertiary treated municipal domestic effluents. This peak might be created by salts. The STE sample collected in May 2004 shows significant differences to the sample collected seven months earlier (Figure 2-12). This sample is characterized by a strong bimodal molecular weight distribution including a high portion of polysaccharide-like organic material and an unidentified peak of saturated compounds of medium molecular weight at 3,500 s (approximately 1,000–4,000 Dalton). Small molecular weight acids are not observed and the sample is characterized overall by a lack of unsaturated or aromatic organic material (UVA data not presented here).

Interesting differences were observed between the STE and the other wastewater effluents (Figure 2-12). The TFU effluent is characterized by high molecular weight substances and small molecular weight acids (3,800 s). It also shows a build-up of humic- and fulvic-like material (2,900–3,200 s) that is UV active and thus unsaturated or aromatic in character (unsaturated and aromatic carbon molecules are cyclic and planar). The TFU processing tank effluent is close to the TFU effluent in character regarding molecular weight distribution. The MBR effluent shows a fingerprint that is closest to a secondary or tertiary treated municipal domestic effluent, characterized by the four previously described peaks. In particular, the peaks representing humic and fulvic substances (2,900–3,300 s) are well developed. This indicates that there was further biological degradation of the organic matrix.

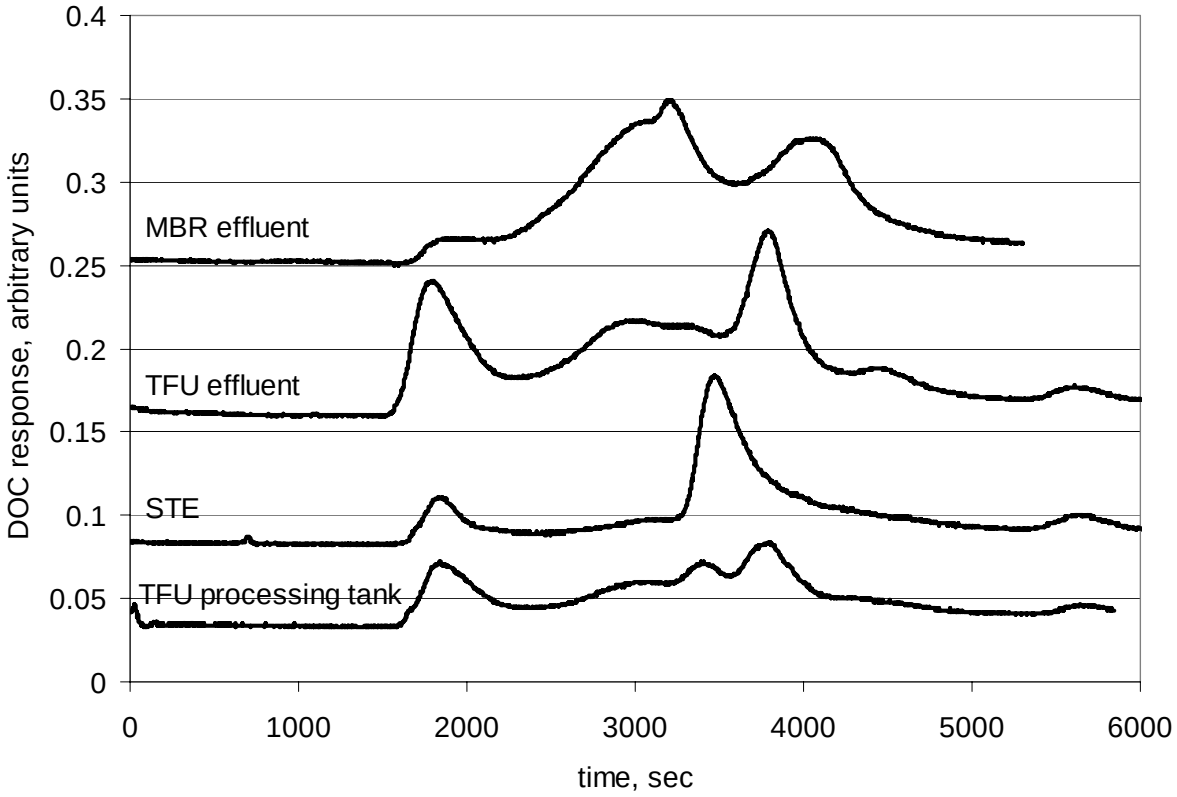


Figure 2-12
SEC Fingerprints of Different Effluent Qualities[3]

To further describe the performance of the different treatment units regarding organic carbon alteration and removal, a bulk organic fractionation approach was applied. This approach differentiated among colloidal, hydrophobic, and hydrophilic organic carbon. Due to limited sample holding times and lengthy sample analysis methods, bulk organic fractionation was conducted on one MBR sample (collected in May 2004), one STE sample (collected in September 2004), and two TFU and TFU processing tank samples (collected in June and October 2004).

The character of the MBR and STE DOC were not expected to change significantly due to the relatively constant effluent quality. However, two TFU samples were analyzed due to potential changes in the DOC character during the TFU nitrification process. The results of these analyses are summarized in Table 2-11 and graphically presented in Figure 2-13 and Figure 2-14. (Note only the June 2004 sample is illustrated.)

[3] Sample collected in May 2004. The y-axis units are irrelevant; this graph presents only the relative shape and location of the peaks in the different samples.

Table 2-11
Results of Organic Carbon Fractionation for Different Wastewater Sources

	STE		TFU Effluent		MBR Effluent	
	DOC %	DOC (mg-C/L)	DOC %	DOC (mg-C/L)	DOC %	DOC (mg-C/L)
Initial feed sample	100.00	33.00	100.00	9.42	100.00	6.31
Colloids	15.00	4.95	32.00	3.01	7.00	0.44
HPI	51.85	17.11	25.63	2.41	31.85	2.01
HPO-A	32.30	10.66	43.22	4.07	54.26	3.42
HPO-N	0.85	0.28	0.00	0.00	6.88	0.43
Sum (colloids + HPI + HPO-A + HPO-N)	100.00	33.00	100.85	9.49	99.99	6.30
DOC_0 (mg-C/L)	33.00		9.42		6.31	
UVA_0 (1/m)	24.00		16.85		10.87	
$SUVA_0$ (L/m × mg)	0.73		1.79		1.72	

HPI=hydrophilic carbon; HPO-A=hydrophobic acids; HPO-N=hydrophobic neutrals; UVA=ultraviolet absorbance; SUVA=specific ultraviolet absorbance

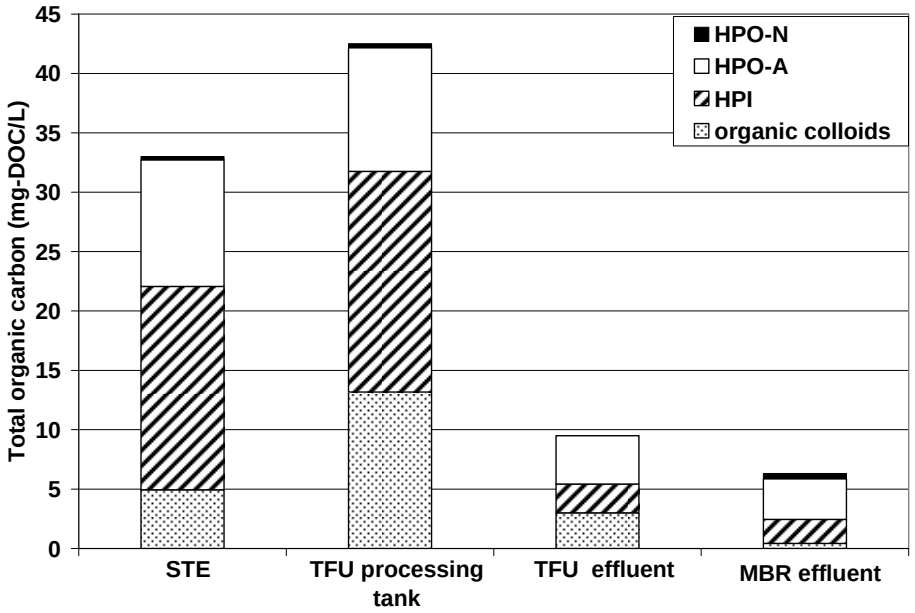


Figure 2-13
Organic Carbon Composition of Different Wastewater Sources

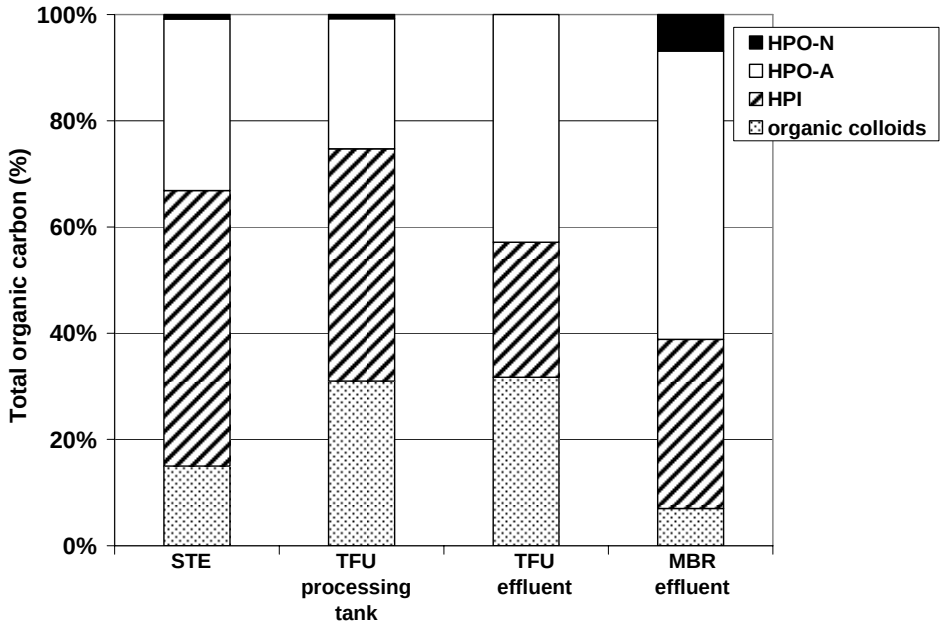


Figure 2-14
Relative Distribution of Bulk Fractions in Different Wastewater Sources

Both the TFU and MBR treatment units were capable of removing organic carbon to approximately 70 and 80% of the initial STE concentration. Mechanisms of removal between both treatment units are different and they each have a different character of residual organic carbon. The increase in SUVA between TFU and MBR effluent—compared to STE and the TFU processing tank effluent—indicates a transformation of the organic carbon matrix to relatively more aromatic and unsaturated compounds, like the ones that occur under the humification of organic carbon. This trend is reflected in the relatively increasing contribution of hydrophobic acids to the TFU and MBR effluent (approximately 45–55%) compared to STE and the TFU processing tank effluent (25 to 30%) (see Table 2-11 and Figure 2-14).

Hydrophobic acids are mainly composed of humic substances. As observed in the SEC analysis, colloidal carbon (polysaccharide-like material) is still a significant contribution to organic carbon in the TFU effluent (32%) but not in the MBR effluent (7%). This does not change after longer operation of the treatment unit (Figure 2-15). Colloidal carbon is typically an easily removed carbon source, which is likely to be quickly degraded during subsequent soil treatment. The better removal of colloidal carbon in the MBR is likely due to size-exclusion and the biological degradation processes that occur in the MBR.

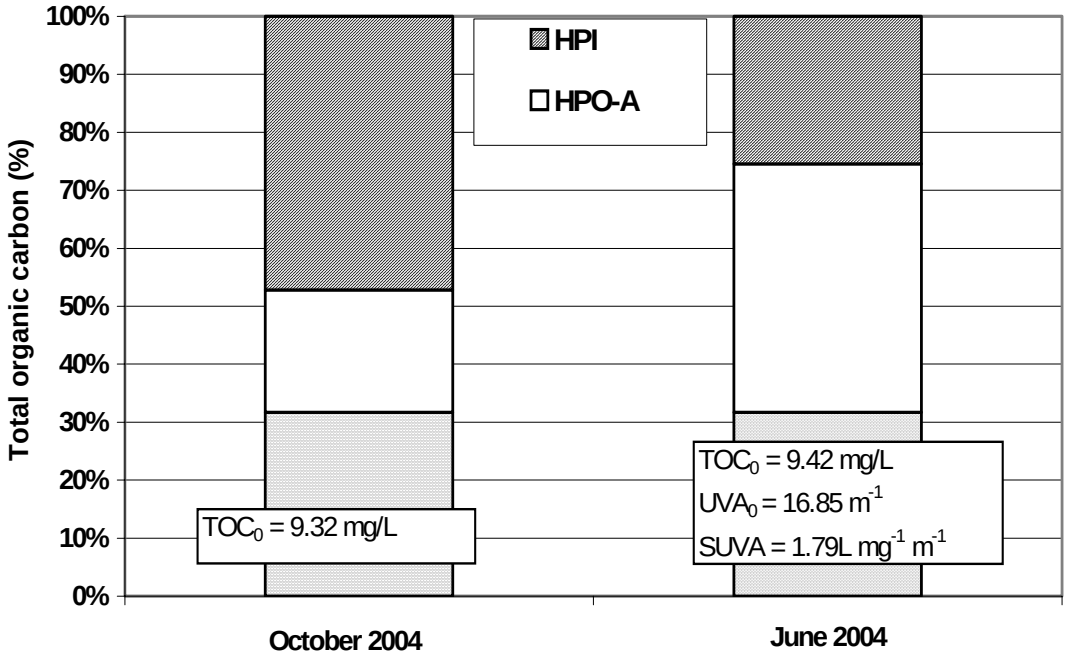


Figure 2-15
Relative Distribution of Organic Carbon in TFU Effluent for Different Sampling Periods

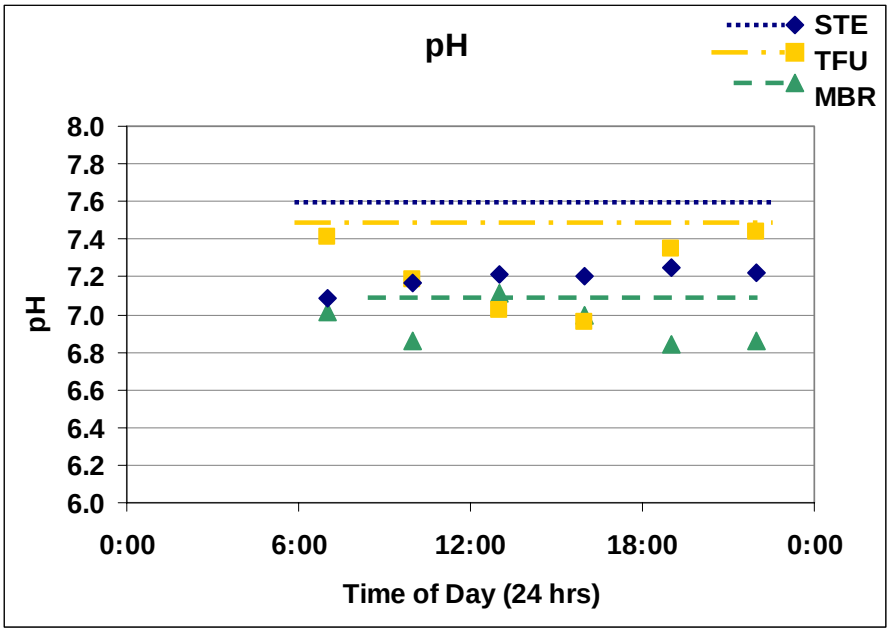
2.5.2.2.3 Short-Term Effluent Quality Variations and Grab Versus Composite Sampling

Effluent samples discussed in this report (aside from this section) were collected as grab samples and were assumed to be representative of the effluent quality. To check this assumption, two separate studies were conducted to compare the effluent quality variability of samples collected as grab samples and those collected as volume-weighted composites. In addition, the sampling studies described in this section provide insight into the shorter-term temporal fluctuations in effluent quality that occur during routine operation.

The first test involved collection of effluent samples every three hours over one day during the period of time when effluent was dosed to the soil (7 a.m. to 10 p.m., see Section 3.2.5). This daily composite sample had individual samples collected at 7 a.m., 10 a.m., 1 p.m., 4 p.m., 7 p.m., and 10 p.m. with analyses conducted on the individual samples as well as the composite sample. The composite sample was created by taking an equal volume of each of the individual samples collected at distinct times and pooling and mixing these samples together. All samples were analyzed for:

- pH
- COD
- TSS
- Nitrate
- Ammonium
- Fecal coliform bacteria

Results for the daily composite versus grab samples are shown in Figure 2-16 and Figure 2-17.



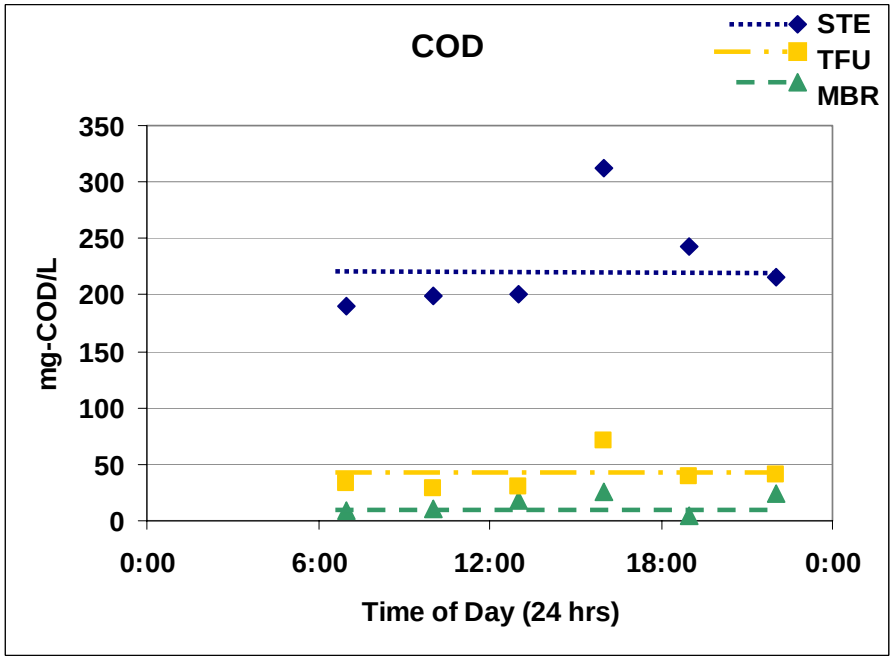


Figure 2-16
pH, COD, and TSS Results for Hourly Grab Samples (Symbols) and Volume-Weighted Composite (Dashed Lines) of Each of the Three Effluents

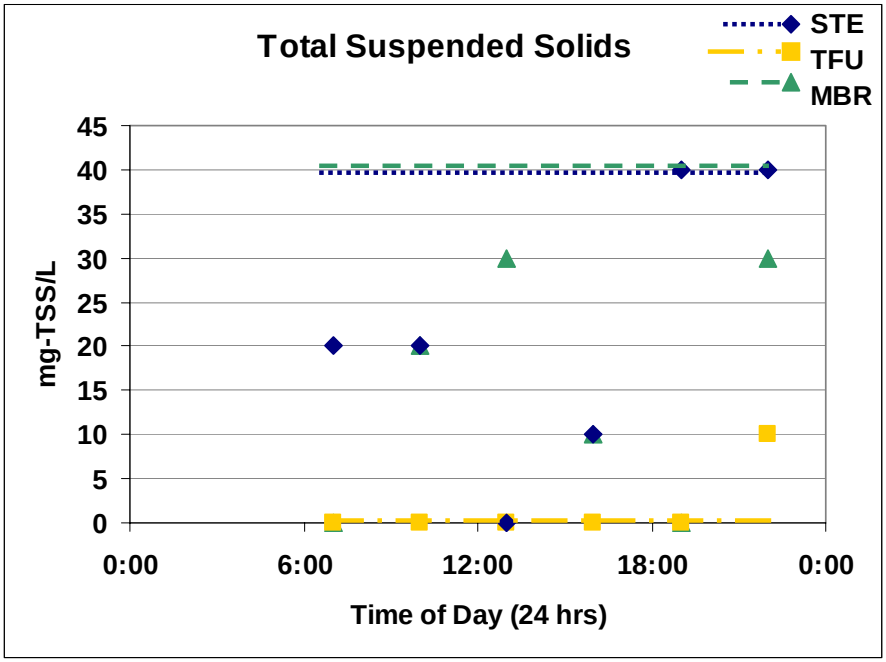


Figure 2-16
pH, COD, and TSS Results for Hourly Grab Samples (Symbols) and Volume-Weighted Composite (Dashed Lines) of Each of the Three Effluents (Cont.)

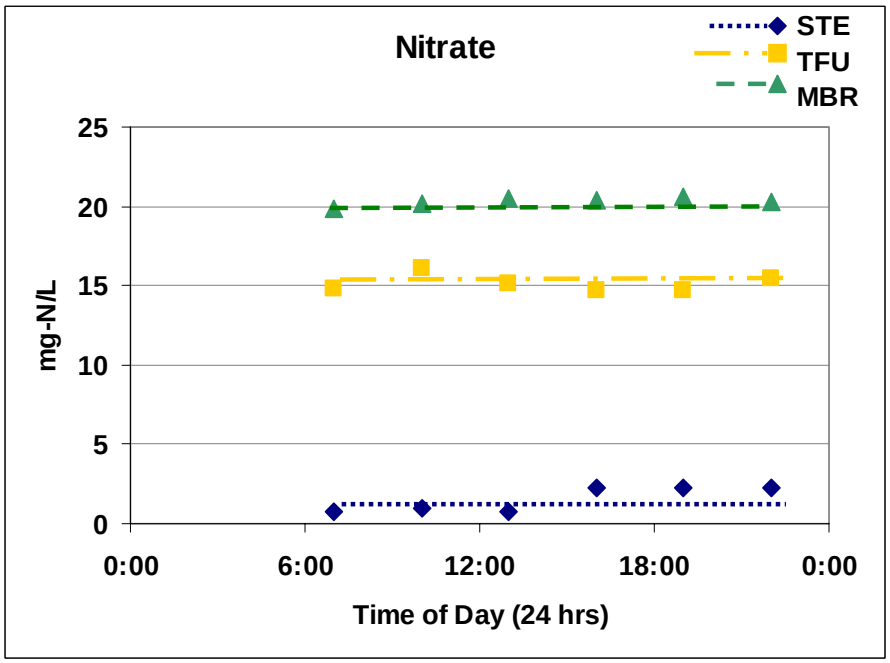
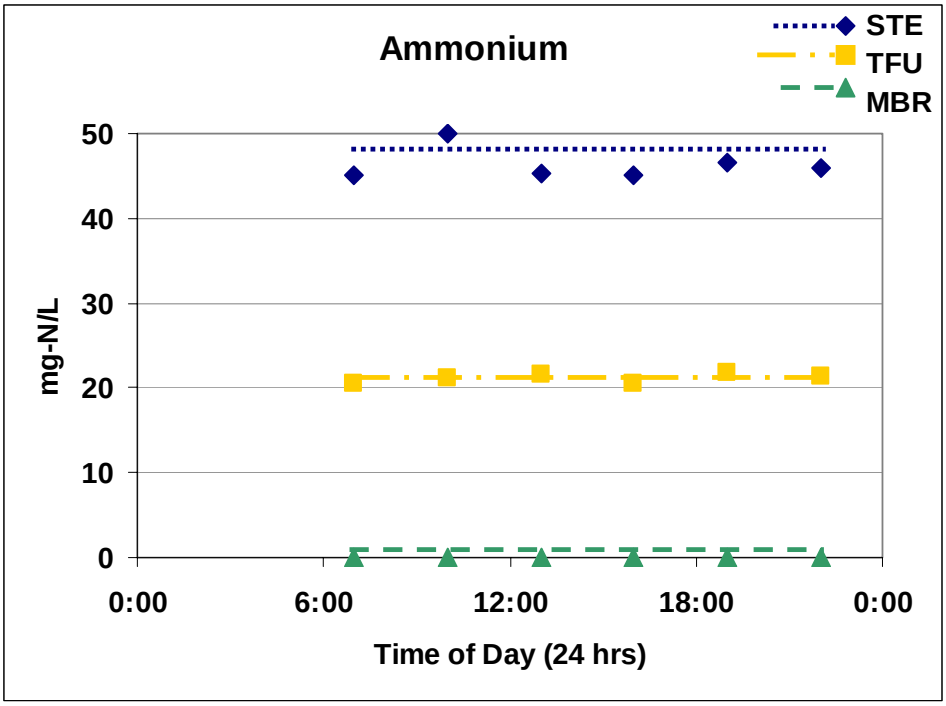


Figure 2-17
Nitrate, Ammonium, and Fecal Coliform Bacteria Results for Hourly Grab Samples (Symbols) and Volume-Weighted Composite (Dashed Lines) of Each of the Three Effluents[4]

[4] No fecal coliform bacteria detected in MBR effluent



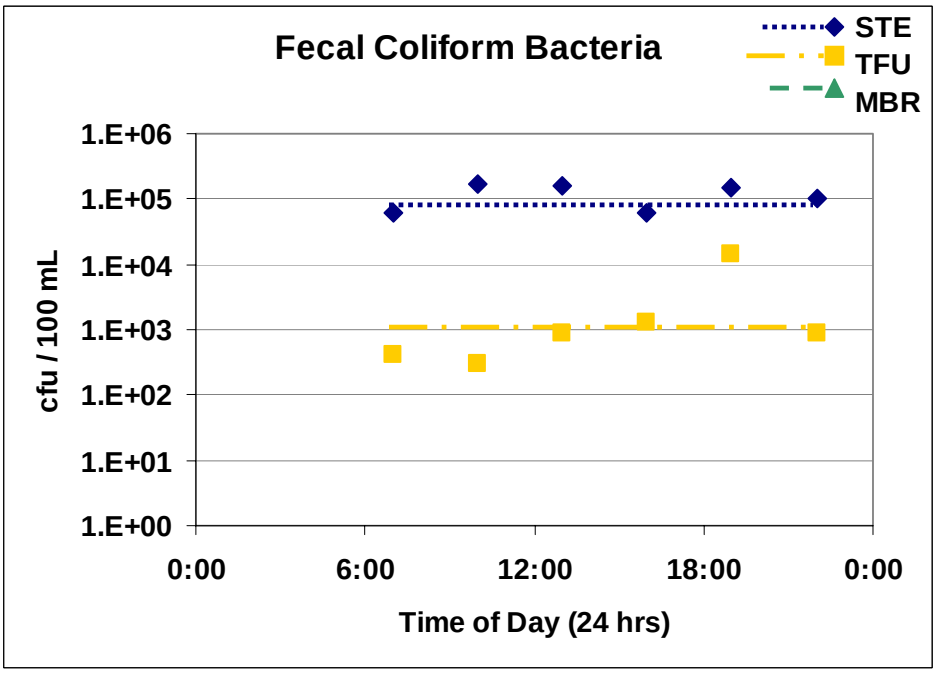


Figure 2-17
Nitrate, Ammonium, and Fecal Coliform Bacteria Results for Hourly Grab Samples (Symbols) and Volume-Weighted Composite (Dashed Lines) of Each of the Three Effluents (Cont.)[5]

[5] No fecal coliform bacteria detected in MBR effluent

The effluent quality for the STE, TFU, and MBR were consistent over the course of the day as indicated by the minimal variability seen in all three effluents for pH, nitrate, ammonium, COD, and fecal coliform bacteria (except for the STE COD). This is not surprising given the residence times of the STE and TFU delivery basins where the samples were collected; 2 days for the STE basin and 2.5 days for the TFU delivery basin. The consistent concentrations in the MBR effluent are due to the stability of the MBR performance during the composite sampling. Higher variability was observed with TSS for all the effluents. This is likely due to sample handling (for example, incomplete mixing of sample) during collection of analysis aliquots. There was no relationship of the coefficient of variance of the individual grab sample or the RPD between the average concentration of all six grab samples and the volume-weighted composite sample concentration due to effluent quality. (Note: The coefficient of variance is an indication of the variability of the sample population.)

Field duplicate samples collected during the composite sampling showed a similar trend, with little variance in the pH, nitrate, or ammonium (RPD of less than 5%; see Section 2.5.2.3). There was slightly more variability in COD and fecal coliform bacteria (RPD of approximately 30%) and significantly more variability in the TSS (RPD of more than 100%). Due to the low variability between the grab samples, an individual grab sample is representative of a volume-weighted daily composite sample, although there is a higher uncertainty in the TSS.

A second composite sampling study involved collection of effluent samples at 4 p.m. every day for a week. Again, all samples were analyzed for pH, COD, TSS, nitrate, ammonium, and fecal coliform bacteria. The composite sample was created by taking an equal volume of each of the individual samples collected at distinct times and pooling and mixing these samples together. Results for the weekly composite versus grab samples are shown in Figure 2-18 and Figure 2-19.

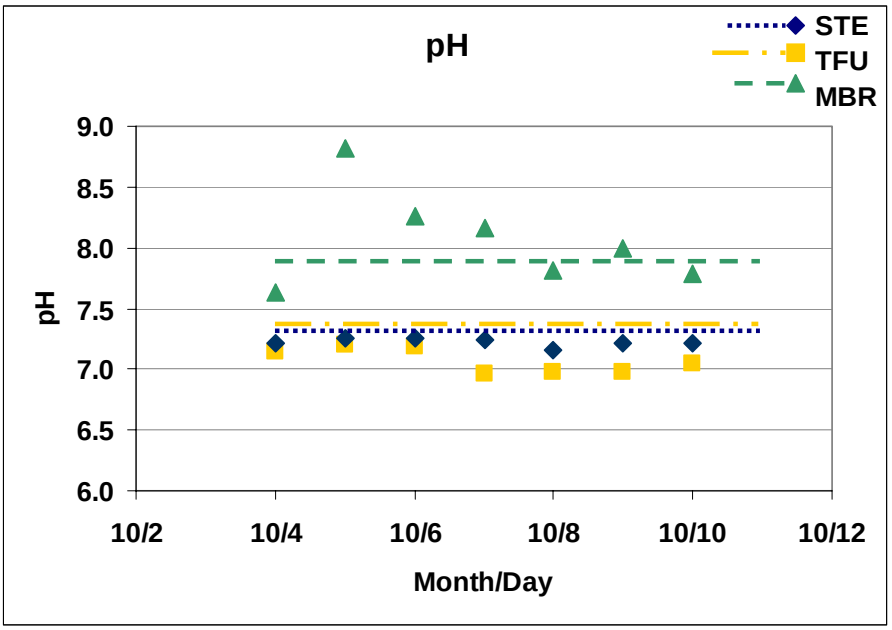
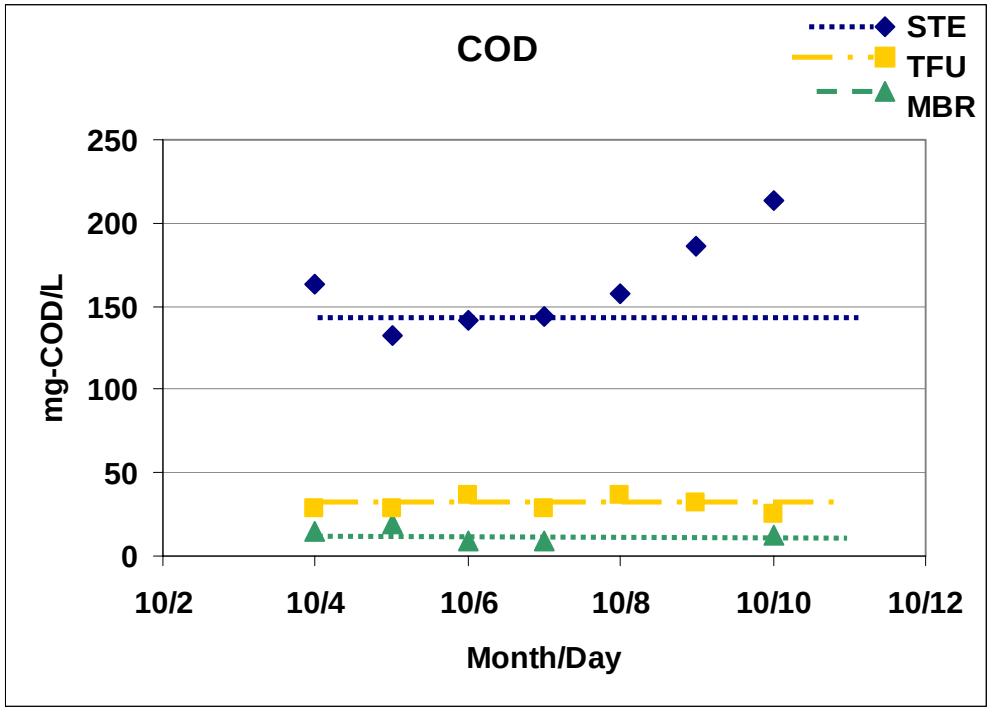


Figure 2-18
pH, COD, and TSS Results for Weekly Grab Samples (Symbols) and Volume-Weighted Composite (Dashed Lines) of Each of the Three Effluents



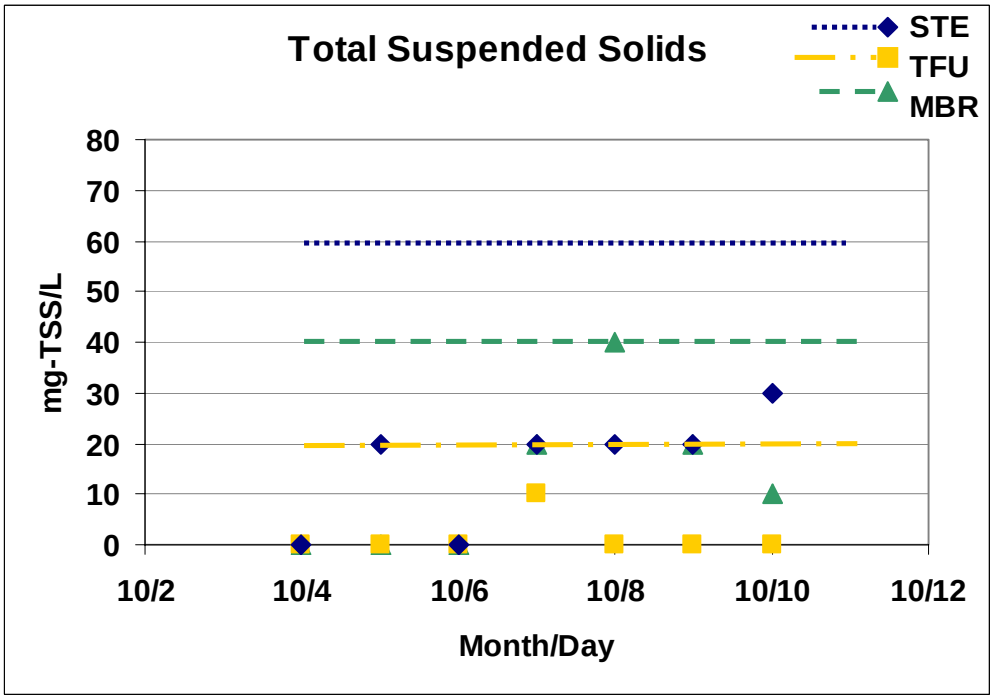


Figure 2-18
pH, COD, and TSS Results for Weekly Grab Samples (Symbols) and Volume-Weighted Composite (Dashed Lines) of Each of the Three Effluents (Cont.)

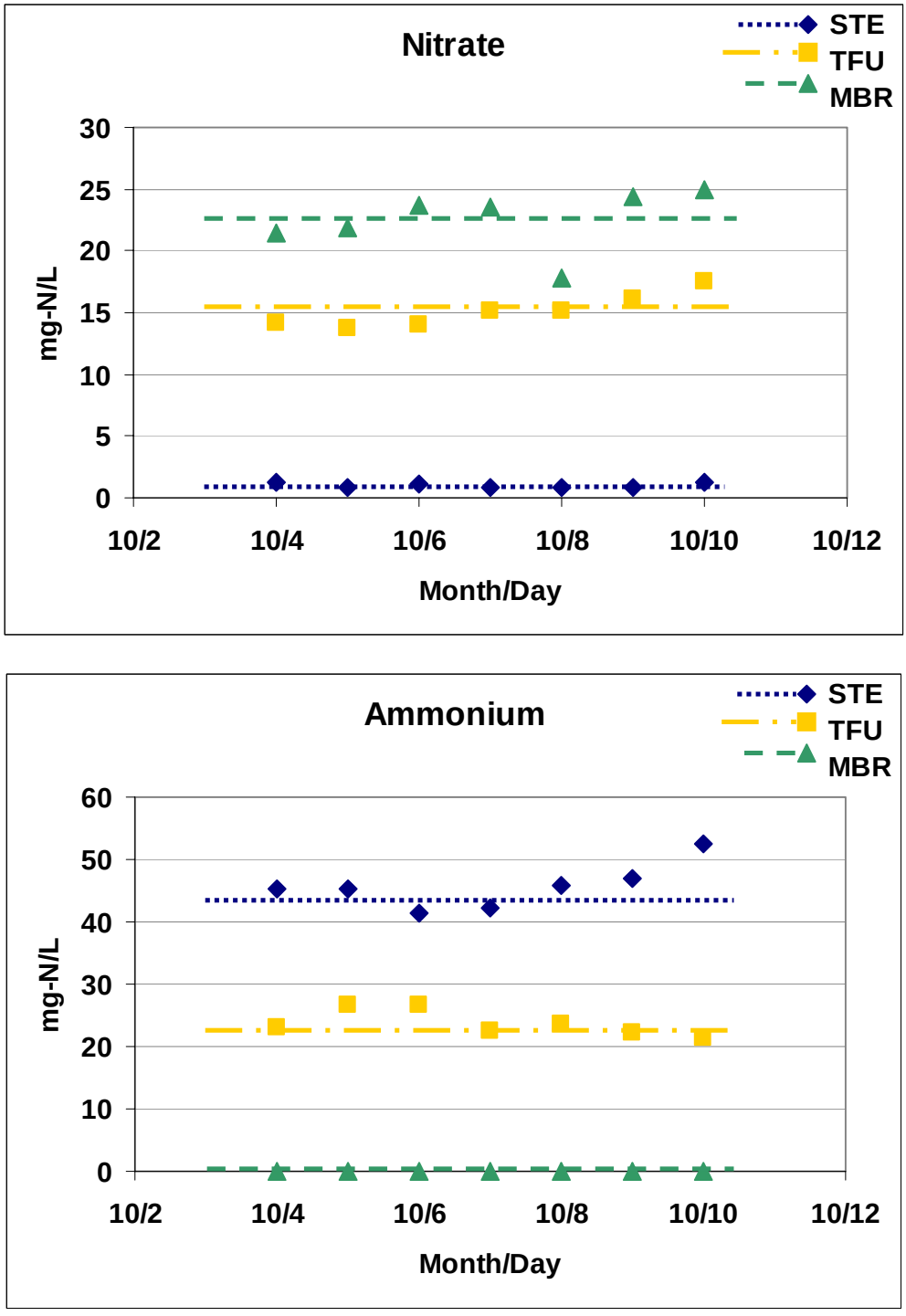


Figure 2-19
Nitrate, Ammonium, and Fecal Coliform Bacteria Results for Weekly Grab Samples (Symbols) and Volume-Weighted Composite (Dashed Lines) of Each of the Three Effluents[6]

[6] No fecal coliform bacteria detected in MBR effluent

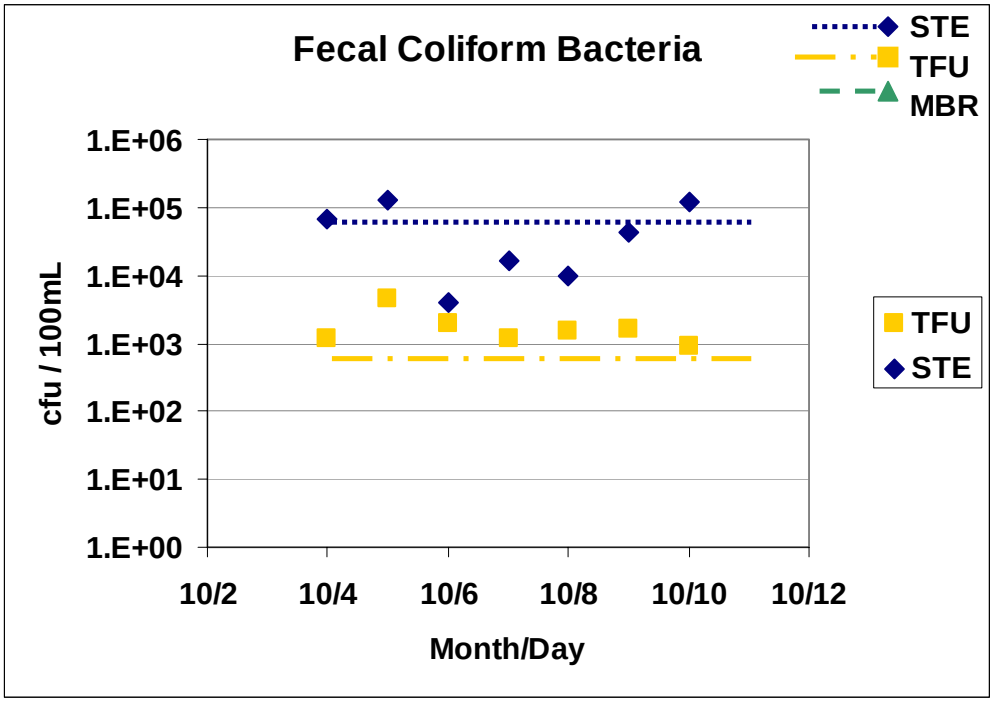


Figure 2-19
Nitrate, Ammonium, and Fecal Coliform Bacteria Results for Weekly Grab Samples (Symbols) and Volume-Weighted Composite (Dashed Lines) of Each of the Three Effluents (Cont.)[7]

Similar to the comparison of hourly grab samples to a daily volume-weighted composite sample, the effluent quality for the STE, TFU, and MBR were consistent over the course of the week. This is indicated by the low variability seen in all three effluents for pH, nitrate, ammonium, COD, and fecal coliform bacteria (except for the MBR pH and the STE COD). However, the variability observed over the week was greater than that observed over the course of one day. The higher fluctuations in the MBR pH are similar to those observed over the operational period and reflect cyclic levels of nitrification during operation. Again, higher variability was observed with TSS for all the effluents and is likely due to sample collection and handling (for example, incomplete mixing of sample) during collection of analysis aliquots.

Field duplicate samples collected during the composite sampling showed similar trends:

- Little variance in the pH, nitrate, or ammonium
- Slightly more variability in COD
- Significantly more variability in the TSS and fecal coliform bacteria

The consistent concentrations in the STE, TFU, and MBR effluents are attributed to the stability of the treatment unit performance during the composite sampling. Some of the weekly variability of the effluent quality is not revealed by weekly grab samples (for example, STE COD). However, the minimal variability between the daily grab samples and the weekly volume-weighted composite samples suggests that the results from weekly grab samples are

[7] No fecal coliform bacteria detected in MBR effluent

representative of the effluent water quality. It should be noted that the TSS results did not correlate well (between the average grab samples and the volume-weighted samples) with the RPD approximately 100 to 175% (the composite sample was consistently greater than the grab sample).

2.5.2.3 Sample Analysis Quality Assurance

Standards checks were run for selected parameters during analyses (nutrients and carbon). Results of the standards checks indicate the sample analysis method accuracy was 21% for total phosphorus, 9% for total nitrogen, 6% for nitrate, and 8% for ammonium. DOC analysis was conducted on two instruments based on the expected DOC concentration. The sample analysis method accuracy was 4% for the Shimadzu (higher DOC detection range of 1 to 500 mg-C/L) and 15% for the Sievers (lower DOC detection range of 0.5 to 10 mg-C/L).

In addition to standards checks, analyses of laboratory duplicate sample analyses were conducted. Duplicate laboratory analyses were performed on 22 to 85% of the samples collected (only 10% duplicate laboratory analyses were performed on pH and alkalinity). The RPD allows a comparison of duplicate analyses as described in equation 2.1 (APHA 1998):

$$\text{RPD} = ((b - c)/((b + c)/2)) \times 100 \qquad (2.1)$$

where each sample analysis result and the corresponding duplicate analysis result is b or c. For each analyte measured in the effluent samples, an average RPD was calculated (Figure 2-20). TSS and BOD analyses presented the highest RPD, with all others below 20% RPD.

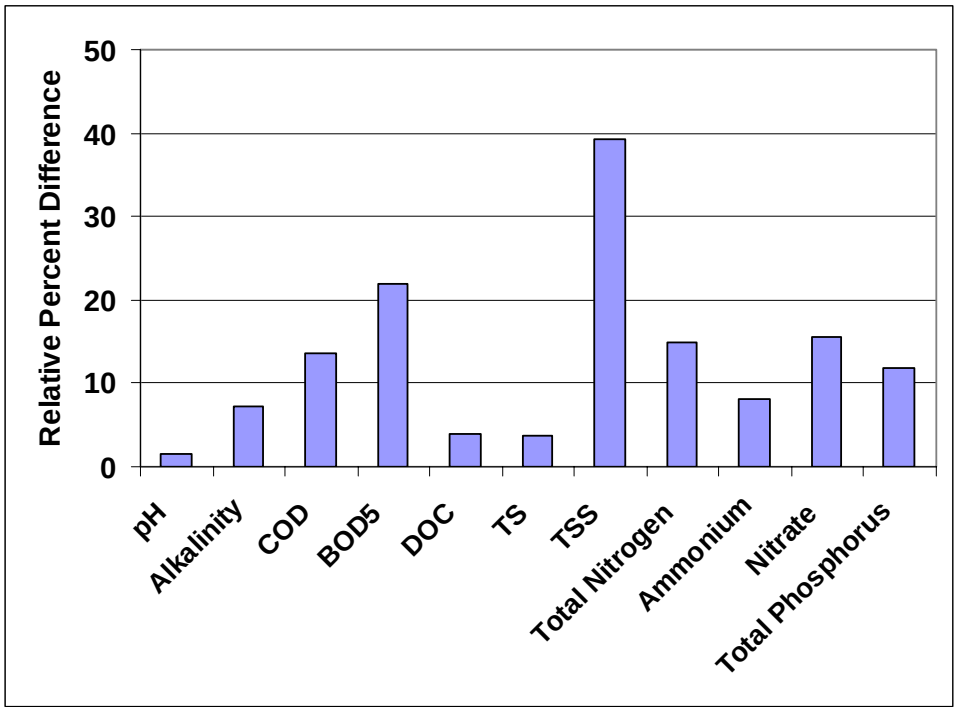


Figure 2-20
RPD for Duplicate Samples

2.5.3 Discussion

This section provides discussion of the routine operational monitoring and effluent quality characterization.

2.5.3.1 Operational Requirements

Following the installation of the treatment units, monitoring of the operation and performance of each unit was conducted. The interception system, in operation for more than five years, required little oversight or operational attention. The first 5,700-L tank of the septic tanks located near the multifamily apartment building was pumped in July 2001. Both 5,700-L septic tanks required pumping in August 2004. This was an expected routine operational requirement (that is, removal of accumulated sludge and scum). The TFU is equipped with an Orenco Systems, Inc. VeriComm communication system that notifies the responsible operator of any problems with pumps, float switches, or effluent height. A single additional pump is used for delivery of effluent to the textile media. No problems with the TFU were encountered after six months of routine operation or one year of operation. Routine cleaning of the TFU was conducted in April 2005.

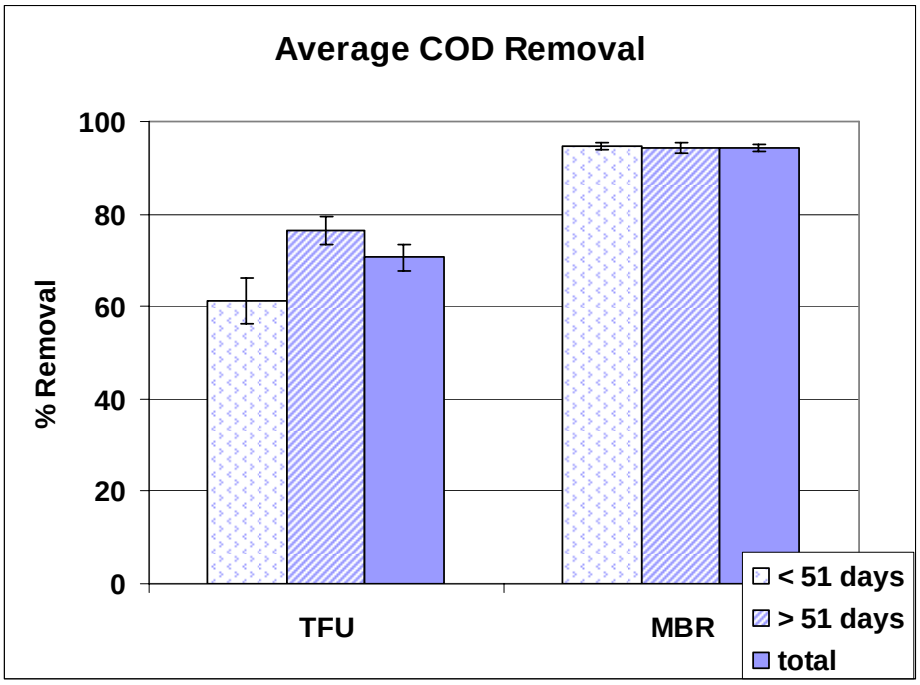
The MBR, by design, requires more operational oversight. In addition, the MBR used in this study was an experimental model. A research building was constructed to house the MBR and maintain a relatively constant temperature (it is not feasible to bury this unit). Reduced flux through the membrane fibers and corresponding increasing operational pressures required membrane cleaning twice during the initial six months of operation. This did not significantly impact the delivery of MBR effluent to the soil test cells (prior to cleaning, sufficient membrane effluent is stored to allow uninterrupted delivery to the soil) or adversely affect the MBR effluent quality (excluding higher variability of pH and alkalinity). However, it required more operator time and monitoring compared to the septic tanks and TFU to ensure successful operation of the unit.

2.5.3.2 Treatment Efficiency

The performance of each of the units was monitored and the results evaluated as percent removals. For this study, removal is a comparison of the TFU and MBR effluent concentration to that of the STE (which was the influent to the TFU and MBR). Figure 2-21 presents COD removal in the TFU and MBR units by day of operation and average removals. Average COD concentration in the STE was 256 mg/L. An improvement in the average removal efficiency was observed in the TFU once nitrification was established (more than 51 days of operation). In contrast, the MBR consistently removed a high percentage of COD (greater than 94%) from the first day of operation.

During this study, the observed ratio of COD to $cBOD_5$ was 1.3 for STE, 8.0 for the TFU, and 11.6 for the MBR. This ratio allows estimation of $cBOD_5$ removals in the absence of $cBOD_5$ analysis results during the first four months of operation. From this estimate, $cBOD_5$ removals were 95% for the TFU and 99% for the MBR (April to August 2004). Measured $cBOD_5$ removals were 97% and 99% for the TFU and MBR, respectively (September to October 2004) (Figure 2-21). While limited DOC data is available before nitrification in the TFU,

approximately 65% of the DOC found in the STE is removed through the TFU unit and 82% of the DOC is removed in the MBR (Figure 2-22).



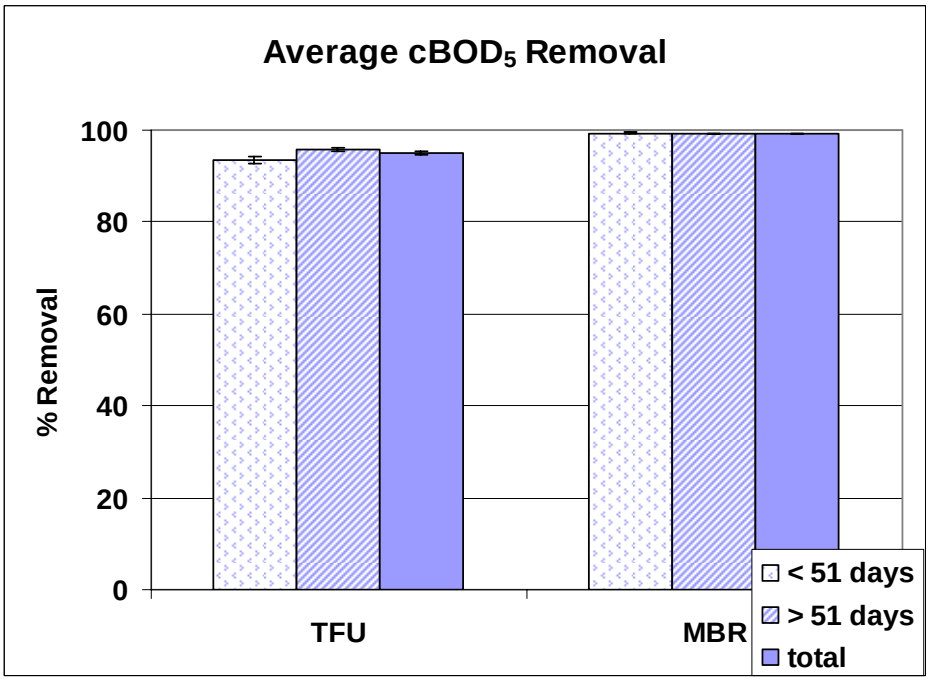
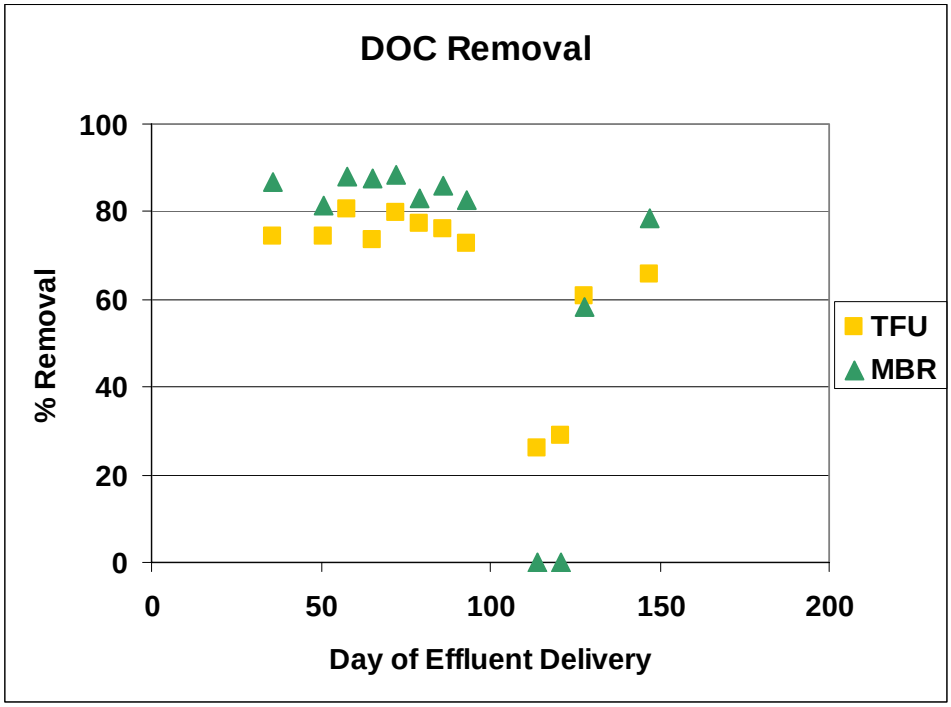


Figure 2-21
COD and $cBOD_5$ Removal (Versus STE) for TFU and MBR Effluents[8]

[8] Error bars are +/- standard error. Note: $cBOD_5$ removals from Day 1–128 based on observed COD to $cBOD_5$ ratio



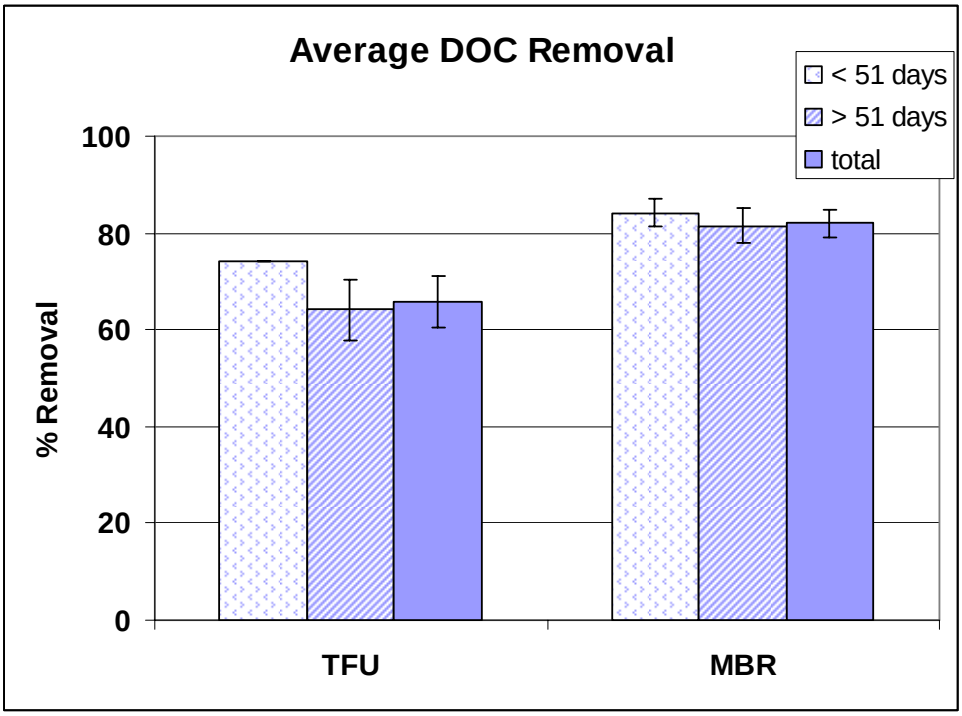


Figure 2-22
DOC Removal (Versus STE) for TFU and MBR Effluents[9]

[9] Error bars are +/- standard error, standard error for TFU < 51 days is 0.01

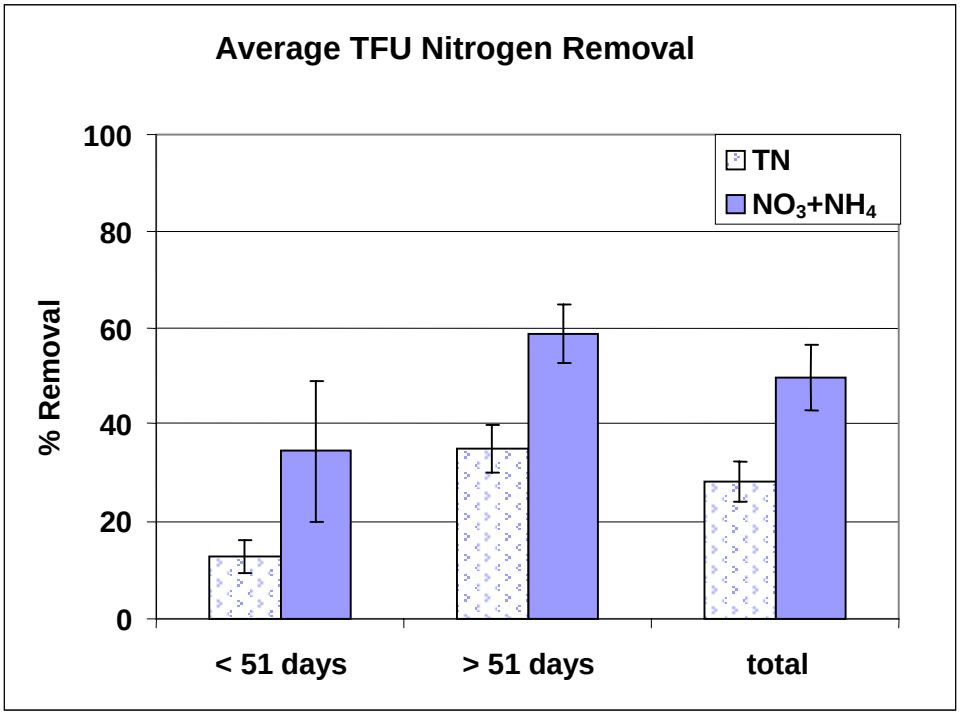
Figure 2-23 presents removal of nitrogen in the TFU and MBR split into less than 51 days of operation (prior to nitrification in the TFU), greater than 51 days of operation (after start of nitrification in the TFU), and over the total period of effluent delivery. The different colored bars represent removals based on either total nitrogen analysis or based on the sum of ammonium and nitrate (assumed to represent the majority of nitrogen in each of the effluents). From Figure 2-23, it is obvious that denitrification occurred in both treatment units.

Based on total nitrogen concentrations, the average removal of nitrogen in the TFU increased after a period of startup of approximately 50 days. It generally ranged between 15 and 68% removal after initiation of nitrification and denitrification (35% average removal after 51 days of operation). This start-up period is consistent with observations made by the manufacturer who suggests that nitrogen removal is expected after approximately the first 30 to 45 days of operation. The start-up period allows establishment of the microbial community within the filter, enabling adsorption of the soluble and colloidal matter in the effluent and the transformation of the material during rest periods (that is, between filter dosing). While the start-up period was within expectations, the slightly longer start-up period may be attributed to daily temperature changes in April during startup before completion of the soil berm along the top of the TFU for insulation. April temperatures ranged from -5 to 25 °C (23 to 77 °F) with 15 °C (59 °F) temperature changes on several individual days. In addition, as previously mentioned, nitrogen removal efficiencies may have been limited by alkalinity, venting of the filter pod, and the recirculation rate of the TFU during this study was approximately 6-to-1. A recirculation rate of 3-to-1 would likely increase the total nitrogen removal.

Higher removals of 45 to 85% were observed in the MBR (56% average removal). During MBR startup, prior to effluent delivery to the soil (January and February 2004), activated sludge was added to the MBR. A start-up period of approximately three weeks was required before nitrification was observed (Larsson and Persson 2004). Total nitrogen removal by the MBR may have been limited due to operation of the unit below optimum MLSS concentrations and low alkalinity. Nitrogen removal efficiencies based on the combination of nitrate and ammonium concentrations indicate similar trends, although the average removal efficiency was higher: 50% for the TFU and 64% for the MBR.

Both the TFU and the MBR are designed to remove TSS by physical filtration. During this six-month study, the average TSS removal efficiency remained constant at approximately 45% and 75% for the TFU and MBR, respectively (Figure 2-24). The higher error bars associated with the TFU are attributed to four TSS concentrations near the method detection limit for both STE and the TFU effluent as well as the higher RPD observed in laboratory quality assurance samples indicating less certainty in the TSS data (duplicate samples). Removal of phosphorus was also observed in the TFU and MBR (Figure 2-24). An average of 10 to 20% removal of total phosphorus was observed in each of these units, with significant variability in the removal values.

Complete removal of fecal coliform bacteria was observed in the MBR (100% average removal), which is based on solids separation and ultra-filtration (nominal membrane pore size of 0.04 μm). The removal efficiency of the TFU improved during the course of this study: 29% average removal before 51 days of operation and 96% removal after 51 days of operation.



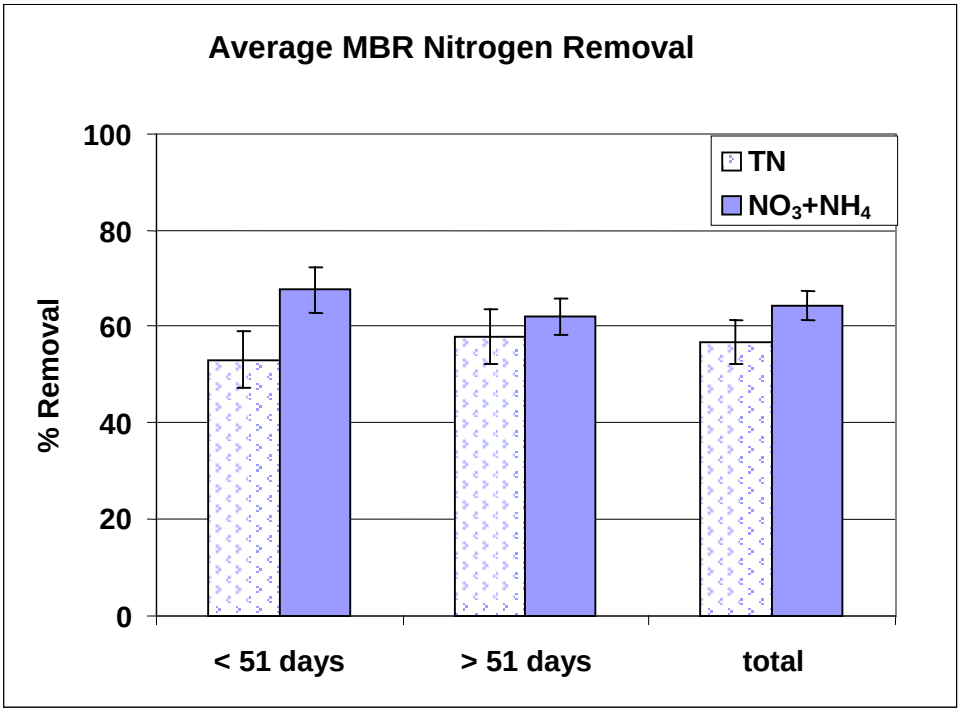


Figure 2-23 Nitrogen Removal (Versus STE), Based on Total Nitrogen (TN) or the Sum of Nitrate and Ammonium (NO_3 + NH_4) for TFU and MBR Effluents[10]

[10] Error bars are +/- standard error

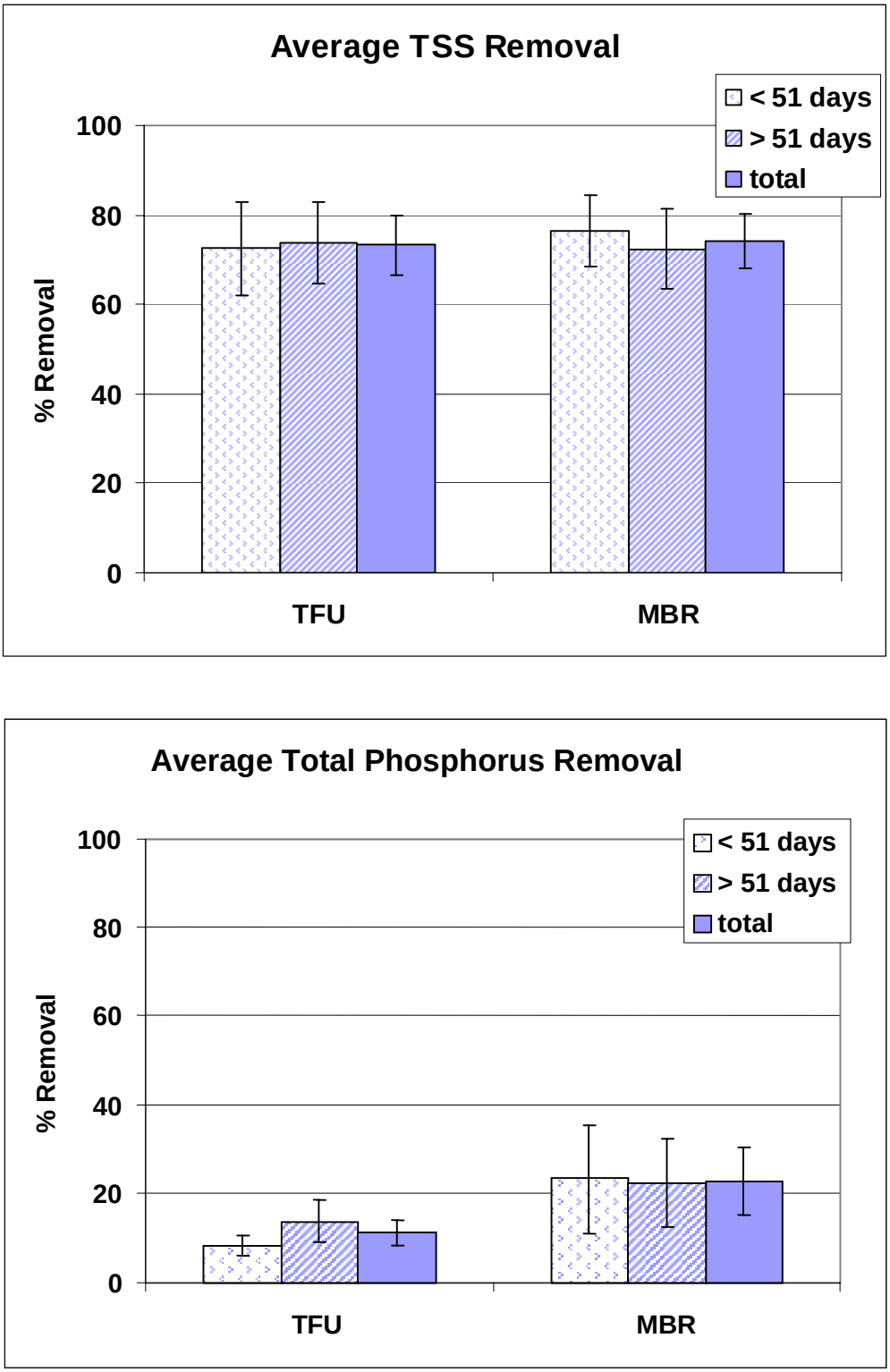


Figure 2-24
TSS and Total Phosphorus Removal (Versus STE) for TFU and MBR Effluents[11]

[11] Error bars are +/- standard error

CFDs are useful tools to estimate the proportion of the members of a population whose measured values exceed or fall short of some stated level (for example, estimation of the percent of effluent samples with a total phosphorus concentration below 10 mg-PO4/L). Figure 2-25, Figure 2-26, and Figure 2-27 present CFDs for COD, TSS, total nitrogen, total phosphorus, and fecal coliform bacteria for data collected from each of the three treatment units over the course of this study (non-detect values were treated as half the detection limit).

The data presented in the CFD figures allow for a comparison of effluent quality samples within an effluent type as well as between effluent types. For example, with respect to total nitrogen based on Figure 2-26, 50% of STE samples had a total N concentration below 63 mg-N/L, while for TFU, that value is approximately 44 mg-N/L, and for the MBR effluent, it is approximately 25 mg-N/L. It is important to note that the CFD for total nitrogen includes all data collected during this study (April through October 2004).

In the case of the TFU, the graph presents a conservative total nitrogen value (that is, highest value of total nitrogen). If the CFD included only values after nitrification began (more than 51 days of operation), the 50% concentration would be less than 44 mg-N/L. Differences among effluents are revealed with respect to all parameters except total phosphorus. The advanced treatment units are not designed to remove phosphorus. The fecal coliform bacteria graph presents only STE and TFU effluent results due to the complete removal of bacteria in the MBR unit.

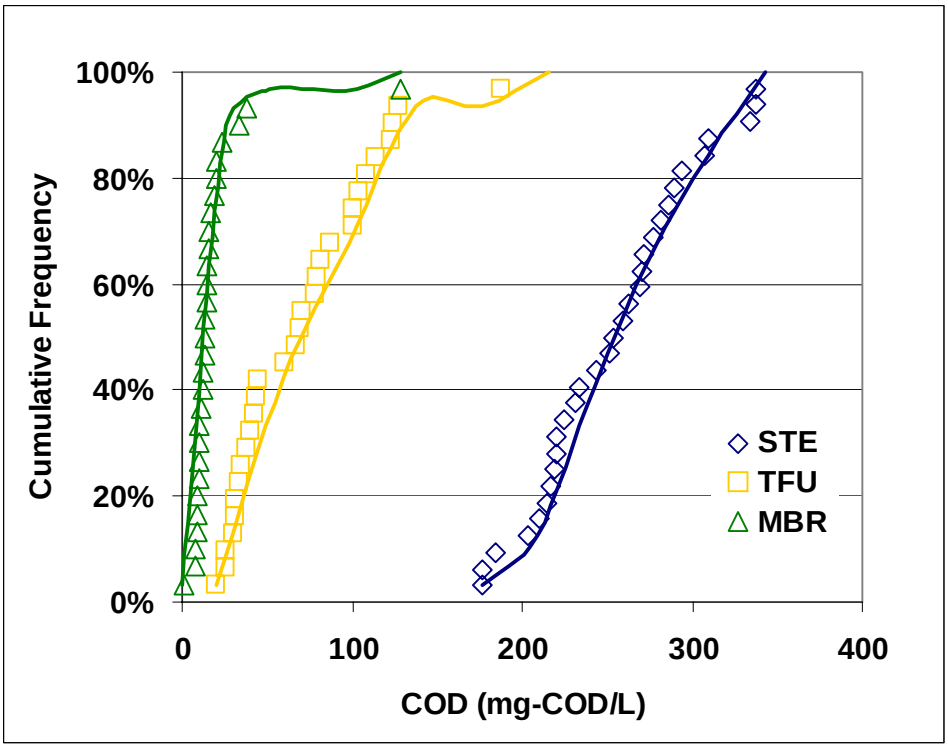


Figure 2-25
Cumulative Frequency Distribution Graphs for COD and TSS From Effluent Samples Taken April–October 2004

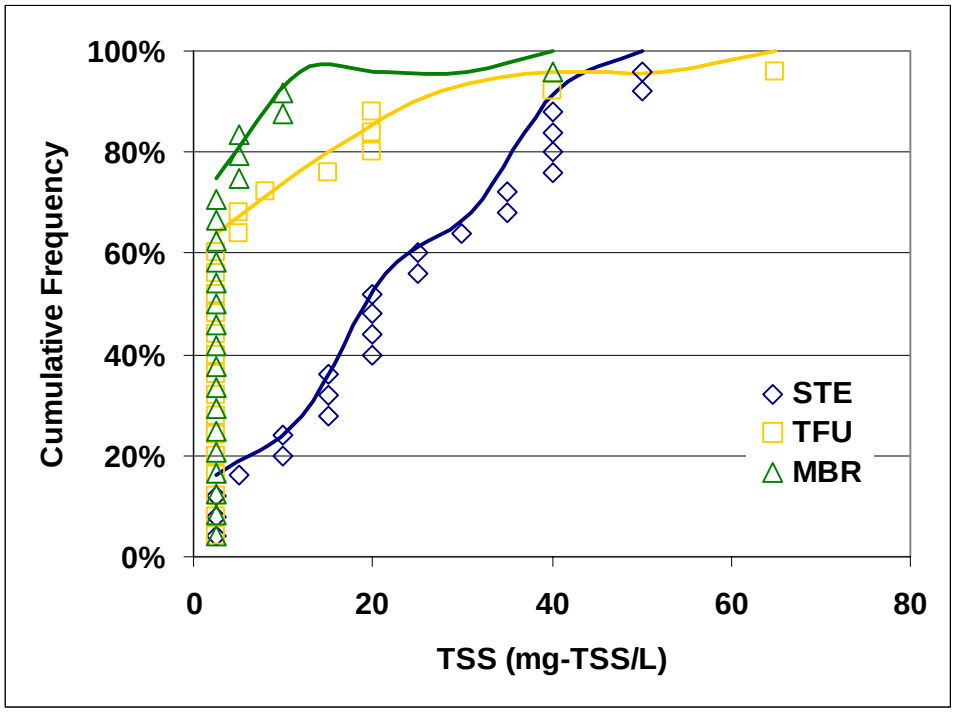


Figure 2-25
Cumulative Frequency Distribution Graphs for COD and TSS From Effluent Samples Taken April–October 2004 (Cont.)

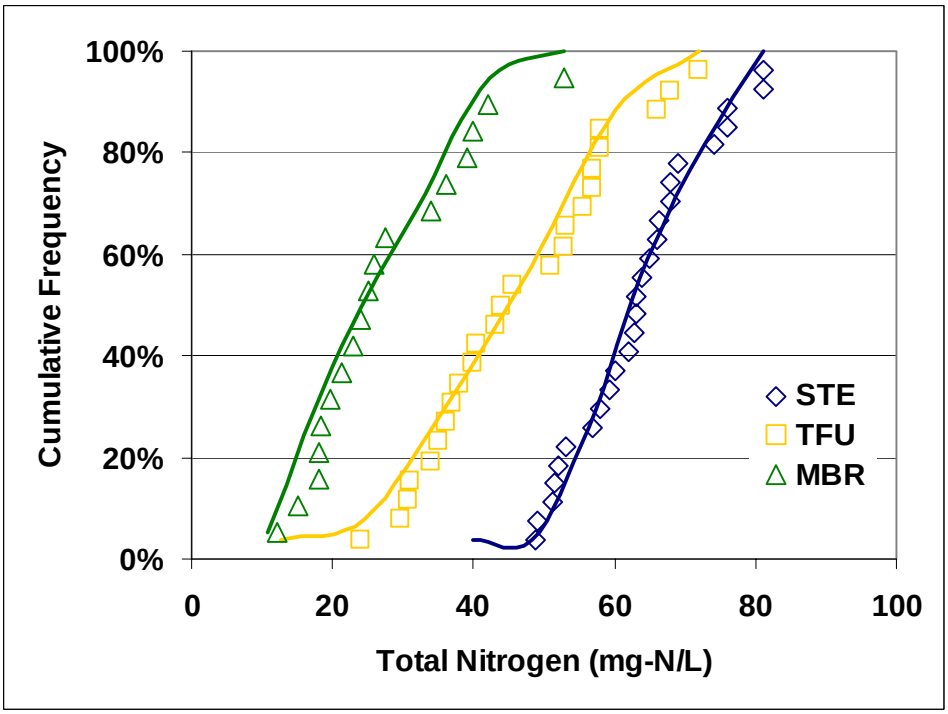


Figure 2-26
Cumulative Frequency Distribution Graphs for Total Nitrogen and Total Phosphorus From Effluent Samples Taken April–October 2004

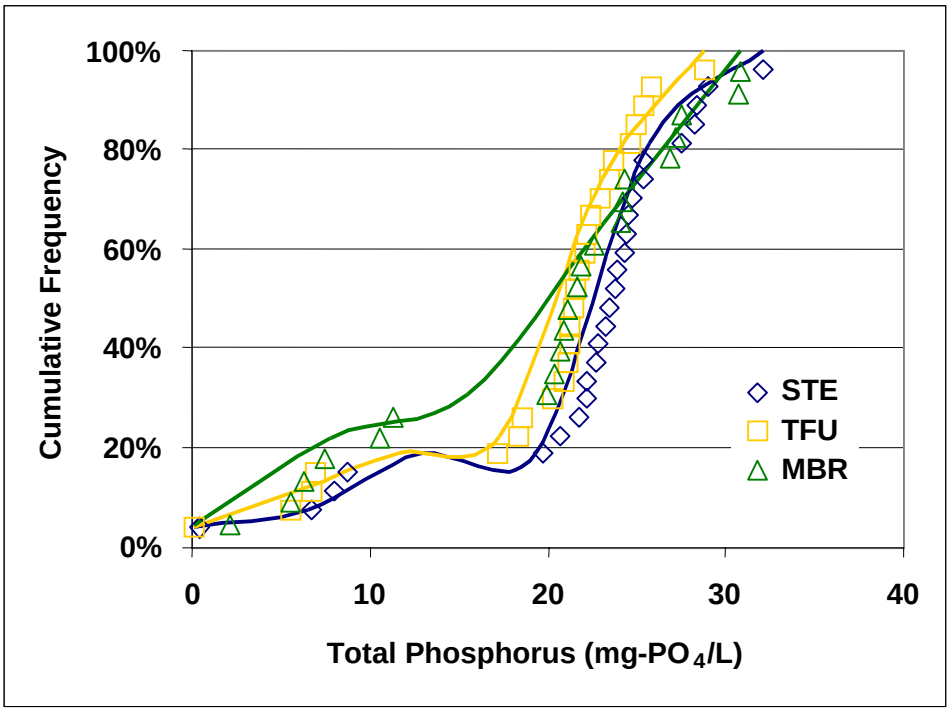


Figure 2-26
Cumulative Frequency Distribution Graphs for Total Nitrogen and Total Phosphorus From Effluent Samples Taken April–October 2004 (Cont.)

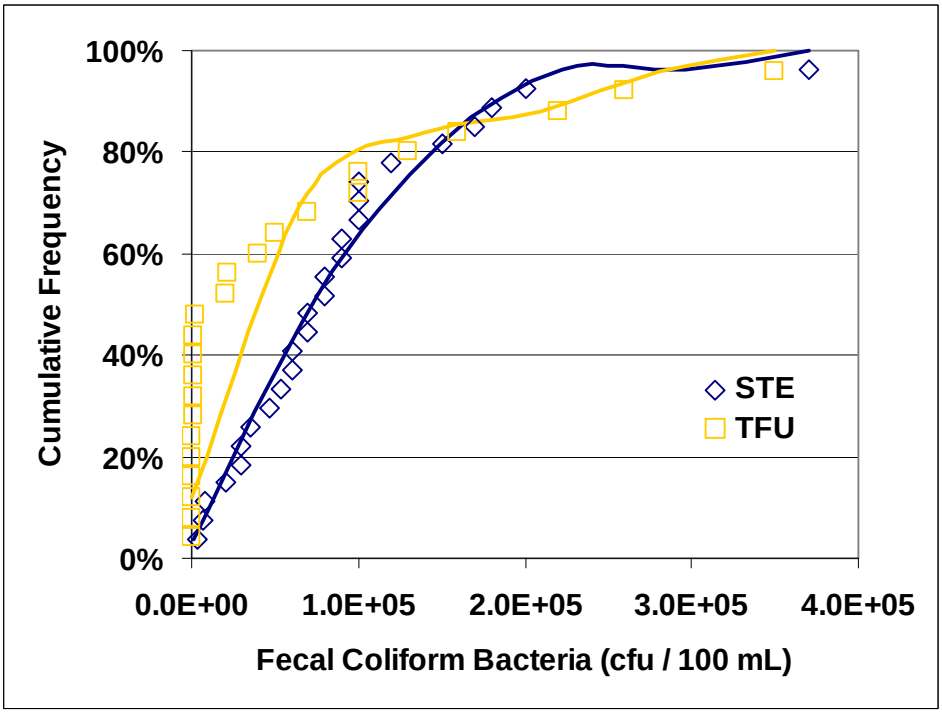


Figure 2-27
Cumulative Frequency Distribution Graphs for Fecal Coliform Bacteria From Effluent Samples Taken April–October 2004

3 WASTEWATER EFFLUENT TREATMENT IN SOIL

3.1 Introduction

The modern OWS includes a wastewater source (typically a dwelling unit), a tank-based treatment unit (for example, septic tank), and a soil infiltration unit (for example, subsurface trench or bed). In these types of systems, water use from all fixtures and activities generates a combined raw wastewater. The wastewater flows into a septic tank buried outside but adjacent to the home or establishment. The principal treatment processes in a septic tank include sedimentation, flotation, and some anaerobic digestion. STE still contains high concentrations of organic matter, TSS, nutrients, and microorganisms. It is not suitable for discharge to a receiving environment without further treatment. Requisite further treatment is commonly achieved by STE discharge into a subsurface trench or bed (typically filled with gravel, a chamber, or some other media from which infiltration occurs), percolation through an underlying vadose zone, and recharge to groundwater (Figure 3-1). When a partially-treated effluent (for example, STE) is applied to soil, infiltration and percolation through the unsaturated porous media (that is, vadose zone) involves a complex set of hydraulic and purification processes. These processes can interact to reliably achieve and sustain advanced treatment efficiencies. The soil functions as a PMB and hydraulic and purification processes interact in a dynamic manner, evolving as the system matures from startup through the first year(s) of operation.

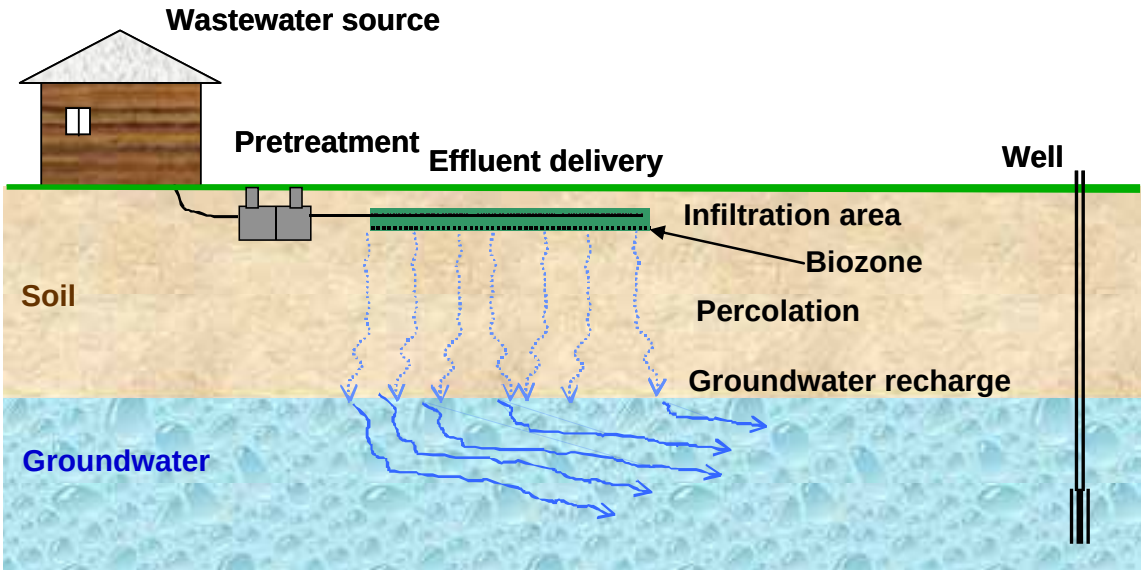


Figure 3-1
Schematic of a Conventional OWS

Research and field experience have shown that soil treatment can achieve high purification efficiencies for wastewater effluents during infiltration and percolation before groundwater recharge (Van Cuyk *et al.* 2001a; Ausland 1998; Schwagger and Boller 1997). Table 3-1 summarizes the purification that can be reasonably expected in soil following treatment through 1 to 1.5 m of the vadose zone. During soil treatment there is extensive and lengthy contact between wastewater constituents and the soil matrix and its associated biofilms, which develop. These biofilms provide treatment of the effluent without significant reduction in effluent infiltration. This soil treatment is enhanced by the unsaturated flow achieved by daily loadings that are limited to a small fraction of the soil saturated hydraulic conductivity (K_{sat}) (for example, 5% or a 2 cm/d HLR compared to an initial K_{sat} of 40 cm/d).

In addition, soil clogging that evolves at the soil infiltrative surface can create a biozone (a few millimeters to a few centimeters thick), which can beneficially impact treatment by altering the hydraulics and purification processes. For example, soil clogging at the infiltrative surface leads to a reduced permeability and more uniform infiltration with a concomitant unsaturated flow almost regardless of hydraulic loading. This biozone development results in an increased retention time and more intimate contact with the soil media surfaces. Moreover, wastewater-induced soil clogging can increase the soil's biogeochemical reactivity. The clogging can enhance sorption, biotransformation, and die-off/inactivation processes within the biozone at the infiltrative surface itself or in the underlying unsaturated soil (Siegrist 1987; Siegrist *et al.* 1991; Ausland 1998; Van Cuyk *et al.* 1999; McCray *et al.* 2000).

Biozone genesis has been described as a humification-like process and modeled as a function of the mass loading rates of wastewater TSS and total BOD (Siegrist 1987; Siegrist and Boyle 1987). In most soil, biozone genesis must occur to some degree to foster the advanced purification required before groundwater recharge, but not to the point where soil clogging causes hydraulic problems (for example, infiltration rate capacity much less than the applied HLR which causes backups or surfacing of applied effluent).

Soil treatment of STE is commonly relied on as an integral component of an OWS due to its high purification performance with limited operation and maintenance requirements and the relatively low cost and long service life of a soil treatment unit. However, there is a growing interest in developing engineered tank-based treatment systems that can produce higher-quality effluents than STE. These systems could reduce the reliance on soil for further treatment. In concept, the use of a reliable and efficient engineered treatment unit can enable OWSs to be used in settings with unsuitable or poorly suited site conditions (for example, limited lot sizes, limited depth of vadose zone). However, there has been little field research that has demonstrated the treatment efficiency achievable in soil as a function of the effluent quality applied at different HLRs.

To enhance understanding on this subject, research involving controlled field experimentation using effluents of three qualities applied at two loading rates to replicate *in situ* soil test cells is described in this chapter.

Table 3-1
Wastewater Constituents of Concern and Representative Concentrations in Effluents Applied to Soil and Percolates Reaching Groundwater (Siegrist *et al.* 2001)

Constituents of concern (examples)		Domestic STE	Soil percolate reaching groundwater at 1 to 1.5 m depth (% reduction of effluent applied)
Oxygen-demanding substances	BOD_5 (mg/L)	140 to 200	>90%
Particulate solids	TSS (mg/L)	50 to 100	>90%
Nitrogen	Total nitrogen (mg-N/L)	40 to 100	10 to 20%
Phosphorus	Total phosphorus (mg-P/L)	5 to 15	100 to 0%; highly variable due to soil's P sorption capacity
Bacteria (for example, *Clostridium perfringens, Pseudomonas aeruginosa, Salmonella, Shigella*)	Fecal coliform (cfu /100 mL)	10^6 to 10^8	>99.99%
Virus (for example, enteric virus such as hepatitis, polio, echo, and coxsackie; coliphage)	Specific virus (pfu/mL)	0 to 10^5 (episodically present at high levels)	>99.9%

3.2 Materials and Methods

The field component of this research was completed through controlled experimentation at the Mines Park Test Site located on the Colorado School of Mines (CSM) campus in Golden, Colorado. This test site had been established with funding provided by sources other than NDWRCDP, but provides a key resource for accomplishing the objectives of this project. In addition to characterization of effluent quality generated from each of the treatment units (Chapter 2), the performance of the soil treatment component of an OWS was studied as described in this chapter.

3.2.1 Experimental Approach

This field experiment was initiated during fall 2003 when 18 *in situ* test cells were established at the site. Each of these test cells was installed with an open horizontal soil infiltrative surface and triplicate cells were loaded with three different effluents at two HLRs. A set of six ancillary test cells was also installed with gravel at the infiltrative surface. Similar test cells have been employed by other researchers (Siegrist 1986; Tackett 2004). In addition, a study designed to look at the performance effects of infiltrative surface architectures and HLRs of STE on soil was initiated at the Mines Park Test Site in May 2003 (Appendix B). This companion study enabled comparisons to be made with the results of this work and provided a set of test cells dosed with tap water.

After installation, the soil test cells were characterized for hydraulic and purification performance. The hydraulic characterization included measurements of infiltration rates in each test cell. Multicomponent tracer testing, using bromide as a conservative tracer, was employed to evaluate travel times in the vadose zone. During effluent loading, hydraulic monitoring includes measurement of ponding heights above the infiltrative surface and periodic measurement of infiltration rates. Purification for chemicals and pathogens is being studied through sampling of the effluent applied to the soil and soil solution in the vadose zone at 60, 120, and 240 cm (2, 4, and 8 ft) below the infiltrative surface using microporous stainless steel suction lysimeters. Access ports are also provided for inspection of the infiltrative surface and for collection of intact soil cores.

3.2.2 Test Site Characteristics

The Mines Park Test Site was established on the CSM campus southwest of the Mines Park student housing complex located on the southwest corner of Highway 6 and 19th Street in Golden, Colorado. The purpose of the test site is to facilitate research related to conventional unit operations, advanced unit operations, and OWS treatment train operations.

The establishment of the test site was completed in two phases. Phase 1 was completed in July 1998. A wastewater interception and treatment system was installed to support onsite pilot-scale experiments, laboratory research, and teaching (see Figure 2-1). This wastewater interception and treatment system has been used as the source of STE during laboratory testing at CSM over the past several years. A site evaluation was completed in the spring of 2002 and indicated suitable site conditions for OWSs (see Appendix A). Phase 2 initially involved establishment of the field research area in the fall of 2002 (Figure 3-2). For the research described in this report, a series of test cells were installed in fall 2003.

3.2.2.1 Location

The Mines Park Test Site is located on CSM property in the SE¼ SE¼, Section 33, T3S, R70W. The site is currently undeveloped and in a natural state, typical of the Rocky Mountain Front Range foothills. The land is covered primarily with native grasses, yucca, native shrubbery, and scattered small boulders. A small, unnamed drainage flows in the early spring and trends southwest to northeast, parallel to the site approximately 15 m from the southern site boundary. Two City of Golden water lines run north/south through the western portion of the site. There are no other disturbances to the surface or subsurface.

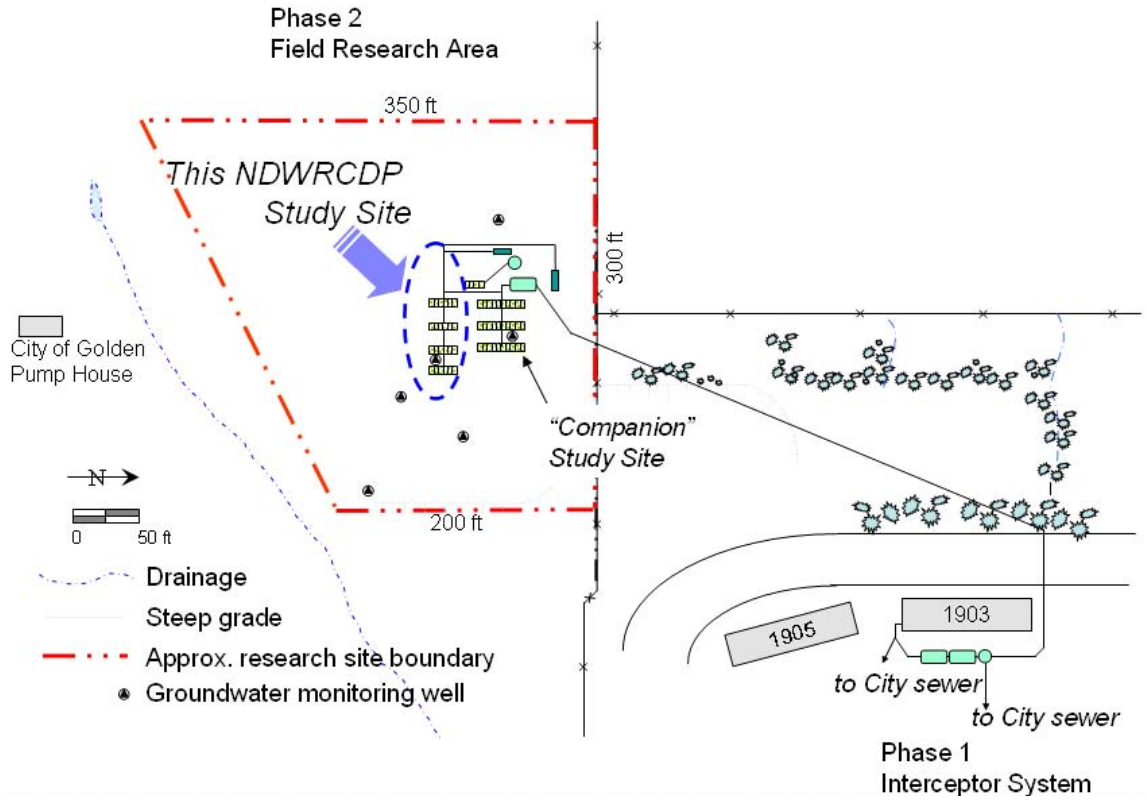


Figure 3-2
Overview of the Mines Park Test Site

3.2.2.2 Geology and Soils

General soil characteristics for Mines Park were initially assessed from the US Department of Agriculture (USDA) Natural Resources Conservation Service (USDA 1983) as primarily fine loamy soils (mixed, mesic Aridic Argiustolls). The parent materials are generally derived from igneous and metamorphic rocks of the mountains and sedimentary rocks of the foothills. The typical soil profile includes:

- Neutral, grayish brown and dark grayish brown sandy loam surface layer (0–18 cm)
- Mildly alkaline, brown sandy clay loam (18–28 cm)
- Moderately alkaline, pale brown sandy loam subsoil layer (28–46 cm)
- Mildly alkaline and moderately alkaline, pale brown sandy loam and gravelly sandy loam substratum (46–152 cm)

It is listed by the USDA Natural Resources Conservation Service as having moderate permeability and an average depth to bedrock of 1.5 m.

Morphologic inspection of the natural soil profiles exposed in two backhoe test pits and nine soil borings was conducted according to accepted procedures (USDA 1981; SSSA 1986; US EPA 1991). Locations of the backhoe test pits and soil borings are shown in Figure 3-3. Photographs of representative soil profiles and the morphologic characteristics are in Appendix A. These inspections revealed a varied soil profile between the western and northwestern areas and the southeastern corner of the site.

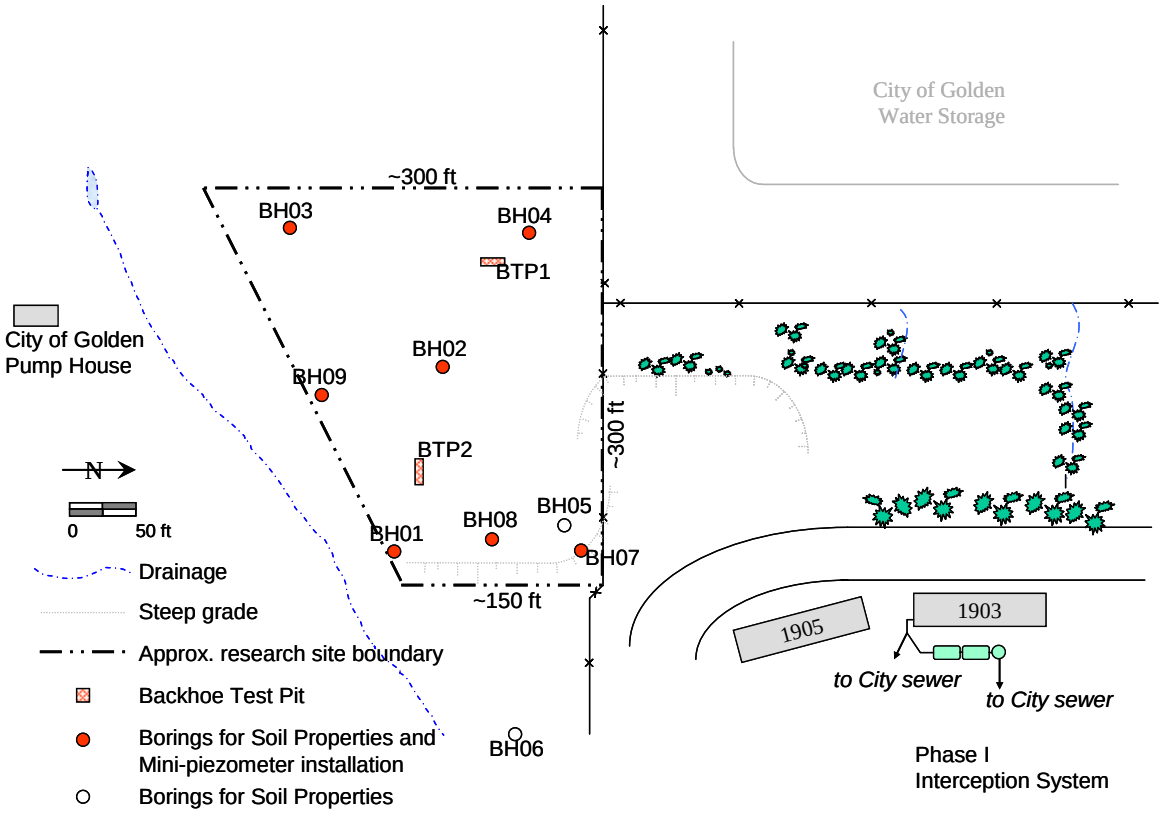


Figure 3-3
Soil Test Locations

Backhoe test pit #1 (BTP1), located in the northwestern portion of the site, revealed soil conditions that are generally dominated by unconsolidated, fine sandy loam soils with little bedding structure and/or macro pores. Roots were observed as deep as 1 m and a transition zone from fine sandy loam to highly weathered, friable igneous rock was observed at 1.7 m below ground surface (bgs). The soil matrix color of the fine sandy loam was generally in the 7YR4/4 range with soil mottling absent. The weathered igneous rock was generally in the 2.5YR5/4 range.

Backhoe test pit #2 (BTP2), located in the southeastern portion of the site, revealed a 6-inch layer of loam underlain by highly fractured, weathered siltstone (fine-grained consolidated sedimentary rock with particles of predominantly silt grade). Bedding planes in the siltstone ranged from 1 to 6 cm thick near the surface with bedding planes up to about 8 cm thick at 1.2 m bgs. The soil matrix color was generally in the 5YR6/4 range with soil mottling along root zones observed to a depth of 60 cm bgs. A transition zone from the weathered siltstone to weathered

igneous rocks (poorly sorted sub-angular conglomerate with less than 3% schist, 40% silica, 55% feldspar (K_{spar}) and less than 2% hornblende) was observed at 1.4 m bgs.

Observations of subsurface lithology in the nine soil borings revealed similar soil conditions with fine sandy loam soils ranging from approximately 60 cm thick in the southwestern portion of the site to 1.8 m thick in the northern portions of the site. The transition zone to weathered igneous rock was encountered at each location underlying the sandy loam. Samples of soil materials were collected from soil borings at 60 cm intervals and analyzed for:

- Water content
- TOC
- Organic matter
- pH
- Total nitrogen
- Nitrate-nitrogen
- Ammonia-nitrogen
- Available potassium
- Percent sand/silt/clay
- Grain size distribution
- Cation exchange capacity

Grain size distribution for the bulk soil fraction was determined by sieving dry soil and weighing the various sand fractions. Results indicated the following soil characteristics:

- 9 to 52% (average 24%) coarse sand to fine gravel (greater than 2 mm)
- 46 to 85% (average 73%) medium to fine sand (2 mm to 0.075 mm)
- 1.3 to 9% (average 3%) silt and clay (less than 0.075 mm)

The grain size distribution was uniform across the site with a general trend of increasing sand particle size with depth. Summary statistics and representative grain size distribution graphs from BH03 are presented in Appendix A. To better define the fine earth fractions, percent silt/sand/clay analysis was determined by hydrometer analysis. Results from this analysis revealed the soils as sandy loam across the site. Complete silt/sand/clay results are presented in Appendix A.

Total percent organic matter in the upper 3 m of soils ranged from 0.1 to 1.4% (average 0.6%) with a general trend of decreasing organic matter with depth. Cation exchange capacity ranged between 2.5 and 22.1 meq/100 g dry soil (average 8.2 meq/100 g dry soil) and was relatively constant across the site. As expected, soils with higher clay content had a slightly higher cation exchange capacity. Other properties of interest include pH, total nitrogen, ammonium-nitrogen (NH_4-N), nitrate-nitrogen (NO_3-N), available phosphorus (avail. P), available potassium (avail. K), and exchangeable calcium and magnesium (exch. Ca and Mg). These soil properties are summarized in Table 3-2.

No saturated conditions, either perched or continuous, were observed in the two backhoe test pits or eight of nine soil borings. In the southeastern corner (soil boring, BH01) of the site located closest to the unnamed surface drainage, continuously saturated conditions were observed at 5 m bgs during sampling of soil borings in April 2002. Shallow piezometers were installed at seven soil boring locations. After installation, groundwater was present at two locations, BH01 at 2.9 m bgs and at BH08 at 4.4 m bgs. See Figure 3-3 for shallow piezometer locations. Soil moisture content, based on dry weight, was determined at each borehole location at 60 cm intervals to

approximately 6 m bgs. Results indicated no marked change with depth (excluding BH01) across the site with most values ranging between 4 and 9%, dry weight basis.

A complete summary of the soil moisture content results is included in Appendix A. While it is acknowledged that these groundwater observation wells were installed during an unusually dry year, mottling indicative of high groundwater was limited across the site. Mottling was observed in three of the nine borings, all located in the southeast portion of the site at depths greater than 1.5 m bgs (BH01 at 3 m bgs, at BH02 at 1.8 m bgs, and at BH08 at 2.3 m bgs).

Table 3-2
Summary of Soil Properties

Statistic	pH	Org. mat. (%)	Total nitrogen (ppm)	NH_4-N (ppm)	NO_3-N (ppm)	Avail. P (ppm)	Avail. K (ppm)	Exch. Ca (ppm)	Exch. Mg (ppm)	CEC (meq/ 100 g)
High	9.1	1.4	585.7	32.2	1.5	26.0	322.0	3770.0	440.0	22.1
Low	5.2	0.1	6.8	1.9	0.5	1.0	50.0	310.0	70.0	2.5
Average	7.3	0.5	124.0	5.2	0.7	4.4	117.3	1214.8	232.4	8.2
Median	7.4	0.5	77.4	3.7	0.6	2.5	109.0	1005.0	230.0	6.8
Std. dev.	1.01	0.33	138.41	4.93	0.19	4.80	46.62	791.94	113.31	4.67
C.V.	0.14	0.64	1.12	0.94	0.28	1.10	0.40	0.65	0.49	0.57

While it is widely recognized that conventional percolation tests are a poor measure of soil hydraulic capacity, tests were completed as required in the Jefferson County Regulations (Jefferson County 1999) and provide a relative measure of hydraulic capacity across the site. Tests were performed on 10 cm diameter holes at a total depth of 1 m bgs. Each hole was filled with water to at least 36 cm for 20 to 24 hours before testing. Following saturation of the test hole, the time for the water to drop 2.5 cm within the lower 15 cm of the hole was measured and recorded as the number of minutes per inch drop (min/in.). Comparison of the individual rates and the average of all rates together indicates that the percolation rates were within the regulatory limits of between 5 and 60 min/in. (2 to 23.5 min/cm). The average percolation rate of 15.5 min/in. (6 min/cm) indicates that the site is suitable for conventional soil absorption of STE.

3.2.3 Soil Test Cell Installation

Pilot scale test cells were employed to mimic a typical soil absorption trench used as the soil treatment unit within an OWS (Figure 3-4). Installation of these soil test cells began with excavation of four trenches, each approximately 74 cm (30 in.) wide, 10 m (30 ft) long, and 90 cm (36 in.) deep. In the State of Colorado, the minimum depth from the ground surface to the infiltrative surface is 60 cm (24 in.) with typical trench installations ranging between 76 and 90 cm (30 and 36 in.) due to irregularities in the topography.

A backhoe with a toothless bucket was employed for trench excavation to minimize disruption of the infiltrative surface. The soil that remained in the trench at the infiltrative surface was carefully removed, leaving the trench bottom free of soil debris and ready for preparation of the future infiltrative surfaces. Galvanized steel culvert sections, each 60 cm (2 ft) in diameter, were placed in the trenches and pressed into the soil surface to approximately 2.5 cm below the infiltrative surface. The culverts were sealed along their outside circumferences with a native soil slurry and bentonite clay pellets. The culverts were placed at least 60 cm apart, with six culvert sections (individual test cells) in each trench. Each individual test cell infiltrative surface (as defined by what lies within each culvert section with no sidewall contribution) was prepped with meticulous care. The infiltrative surface preparation utilized methods described by Siegrist (1987) and Tackett (2004) to yield a surface that was as uniform as possible across all test cells. Any spatial variability associated with the trench excavation process was minimized. Infiltrative surfaces were wetted with tap water and the top 6–10 mm of intact trench bottom (following trench excavation) was scarified and loose soil was removed with a vacuum.

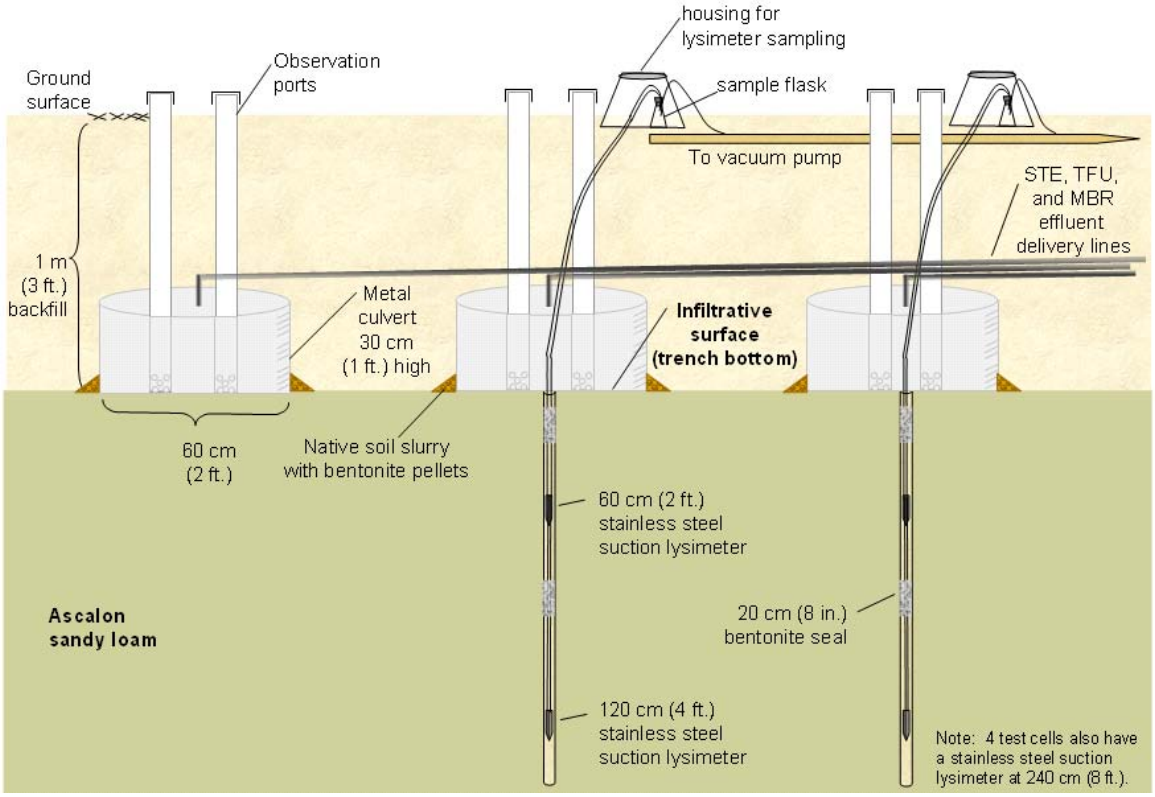


Figure 3-4
Schematic of Test Cell Installation

Access to the infiltrative surface from the ground surface was achieved using vertically placed 7.5 cm diameter PVC casings as access ports. The portion of PVC touching the infiltrative surface was perforated with holes to allow effluent contact throughout the test cell. The PVC rests on the infiltrative surface and protrudes upward to ground surface. Each test cell has three such access ports for determination of infiltration rates, observation of effluent ponding heights, and to allow access for future coring of the vadose zone beneath the infiltrative surface.

The first three trenches (TA, TB, and TC) contained test cells with open infiltrative surfaces (Figure 3-5, Table 3-3). The culvert sections placed in a fourth trench (TG) were filled with approximately 25 cm of gravel (2 to 5 cm diameter) that was washed prior to delivery to the test site and again after receipt at the site. The State of Colorado requires at least 15 cm of gravel with a size range of 1 to 6 cm be placed between the trench bottom and the bottom of the distribution pipe with at least 5 cm of gravel placed above the pipe (CDPHE 2000). Gravel was poured from a height of approximately 80 cm above the infiltrative surface in all six gravel-laden test cells in an attempt to ensure uniform emplacement of gravel within the test cells. Plywood lids were designed as caps to each open and gravel-laden test cell. The plywood and cut geotextile material were placed on top of the metal culvert (with PVC access ports protruding through), allowing for isolation of the test cell before backfilling the trench with native soil.

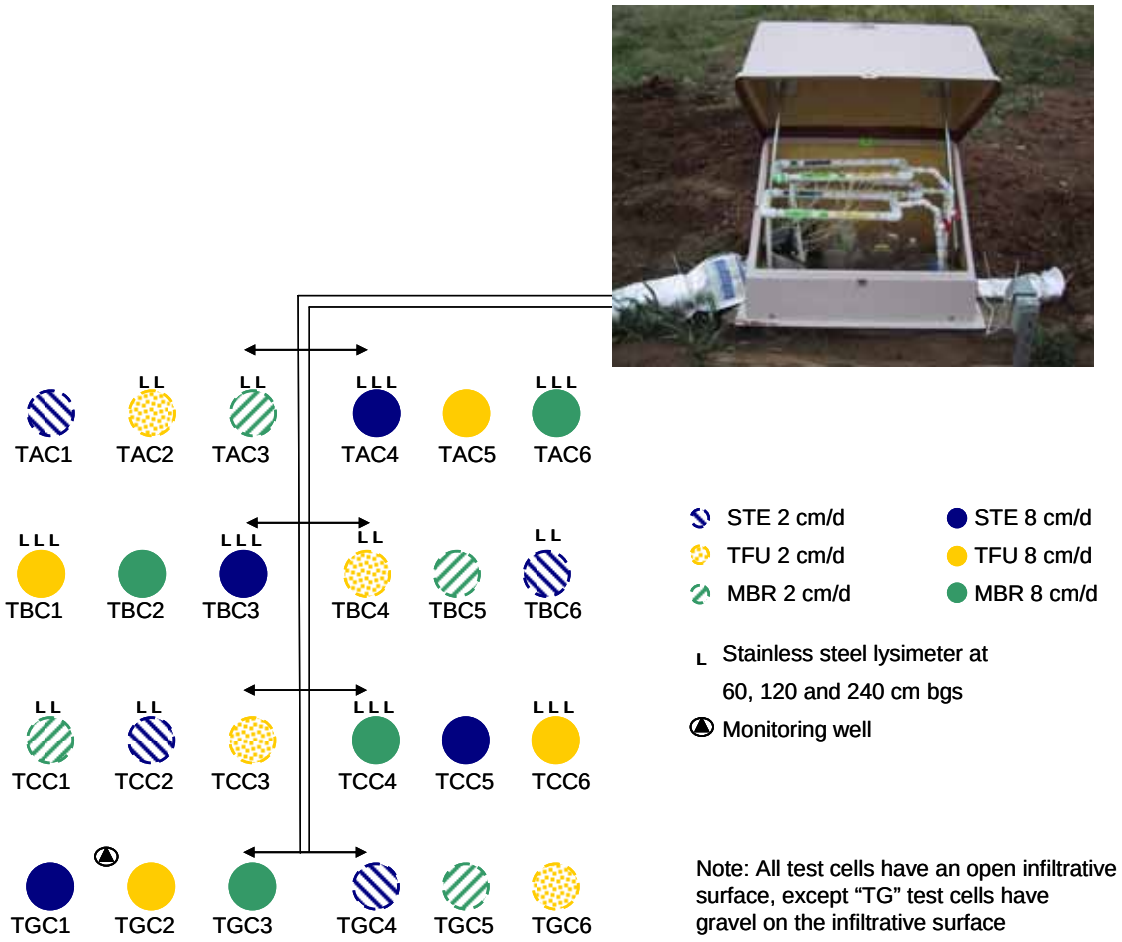


Figure 3-5
Test Cell Location and Delivery Apparatus

Table 3-3
Test Cell Identification, Location, Effluent Applied, HLR, and Lysimeter and Soil Moisture/Temperature Probe Placement

Test Cell Identification	Effluent	Experimental Design HLR	Application Design	Lysimeters	Soil Moisture/ Temperature
TAC1	STE	2 cm/d 5.8 L/d (0.5 gpd/ft^2 0.78 gal/d)	16 doses/day 362 mL/dose	None	None
TBC6			16 doses/day 362 mL/dose	60 & 120 cm (2 & 4 ft)	None
TCC2			16 doses/day 362 mL/dose	60 & 120 cm (2 & 4 ft)	60 & 120 cm (2 & 4 ft)
TAC4		8 cm/d 23.3 L/d (2 gpd/ft^2 3.1 gal/d)	16 doses/day 1456	60, 120, & 240 cm (2, 4, & 8 ft)	None
TBC3			16 doses/day 1456	60, 120, & 240 cm (2, 4, & 8 ft)	60, 120, & 240 cm (2, 4, & 8 ft)
TCC5			16 doses/day 1456	None	None
TAC2	TFU	2 cm/d 5.8 L/d (0.5 gpd/ft^2 0.78 gal/d)	16 doses/day 362 mL/dose	60 & 120 cm (2 & 4 ft)	None
TBC4			16 doses/day 362 mL/dose	60 & 120 cm (2 & 4 ft)	60 & 120 cm (2 & 4 ft)
TCC3			16 doses/day 362 mL/dose	None	None
TAC5		8 cm/d 23.3 L/d (2 gpd/ft^2 3.1 gal/d)	16 doses/day 1456	None	None
TBC1			16 doses/day 1456	60, 120, & 240 cm (2, 4, & 8 ft)	60 & 120 cm (2 & 4 ft)
TCC6			16 doses/day 1456	60, 120, & 240 cm (2, 4, & 8 ft)	None
TAC3	MBR	2 cm/d 5.8 L/d (0.5 gpd/ft^2 0.78 gal/d)	16 doses/day 362 mL/dose	60 & 120 cm (2 & 4 ft)	60 & 120 cm (2 & 4 ft)
TBC5			16 doses/day 362 mL/dose	None	None
TCC1			16 doses/day 362 mL/dose	60 & 120 cm (2 & 4 ft)	None
TAC6		8 cm/d 23.3 L/d (2 gpd/ft^2 3.1 gal/d)	16 doses/day 1456 ml/dose	60, 120, & 240 cm (2, 4, & 8 ft)	60 & 120 cm (2 & 4 ft)
TBC2			16 doses/day 1456 ml/dose	None	None
TCC4			16 doses/day 1456 ml/dose	660, 120, & 240 cm (2, 4, & 8 ft)	None

3.2.4 Soil Vadose Zone Samplers

To investigate the purification of effluent during percolation through the vadose zone required a mechanism that could acquire samples directly below the infiltrative surface without negatively impacting the soil profile and normal flow and transport processes. Horizontal pan lysimeters have been used by some investigators to collect percolating water in the vadose zone, but these pans are difficult to emplace in the subsurface. They can also have difficulties in collecting representative samples from the vadose zone. Porous cup suction lysimeters have been used extensively in the past as tools for evaluating the migration of solutes through the vadose zone (Hart and Lowery 1997; Tackett 2004).

Soil Measurement Systems, Tucson, Arizona, provides instruments for soil investigations and manufactures high-quality stainless steel suction lysimeters. Making lysimeters with porous stainless steel enables acquisition of soil solution samples for chemical analysis from the unsaturated zone. A hydrophilic material with small pores is used. Lysimeter model SW-074, a small single-chamber unit, was chosen for this study. The SW-074 has been successfully used in the ongoing companion experiment at the Mines Park Test Site (Tackett 2004). The SW-074 unit is 11.4 cm long with 9.4 cm of porous steel. It has an outside diameter of 2.2 cm (Figure 3-6). The lysimeters have a nominal pore size of 0.2 microns. The small pore size limits sampling for bacteria, but is needed to inhibit air from entering the lysimeters in place of soil water solution. Lysimeters were installed at 60, 120, and 240 cm (2, 4, and 8 ft) below the infiltrative surface in 12 of the 18 cells. In some cases, an electrical resistance wafer was also installed to enable measurements of soil temperature and moisture levels. Table 3-3 shows the location of the lysimeter and soil temperature/moisture probes.

The lysimeters were preconditioned (according to manufacturer specifications) before being placed. They were flushed with 70% isopropyl alcohol and rinsed with deionized water. Black, high-density polyethylene suction tubing for sample collection was connected to the lysimeters using stainless steel unions. Each lysimeter was leak-tested under water and every union was taped to avoid possible leakage. Duplicate cells representing each effluent type at each loading rate were outfitted, with lysimeters in 12 of the 18 test cells for comparison.

Figure 3-6
Microporous Stainless Steel Suction Lysimeter

Installation of the lysimeters occurred within a single borehole (Figure 3-7 and Figure 3-8). A 3-to-1 (volume-to-volume) slurry of sieved (2.0 mm) native soil and water was poured into the bottom of the borehole prior to placement of the lysimeters. This allowed for at least 10 cm of native soil slurry on top of and below the lysimeter. Lysimeter tubing (for vacuum and sample collection) was pulled through a graduated length of 1.3 cm PVC pipe, until the top of the lysimeter was pulled tightly against the bottom of the pipe. The lysimeter was pushed into the slurry until the middle of the porous section was at the design depth below the infiltrative surface. The deeper lysimeters were placed and allowed to settle overnight to enable the dewatering and consolidation of the slurry before installing the shallower lysimeter. A thinner slurry (2-to-1, soil-to-water) was poured into the borehole to fill in cracks that formed overnight and to create a uniform filter pack (that is, soil) around the lysimeter. Bentonite pellets were then poured down the borehole to act as a seal between lysimeters and to prevent any short-circuiting that might occur between lysimeters. Next, the shallower lysimeter was placed in a similar fashion as the deeper completion. Following installation of the 60 cm lysimeters, the bentonite seal was stopped approximately 15 cm below the infiltrative surface and the remainder of the borehole was filled in with sieved native soil. This was to avoid any complication with having bentonite near the infiltrative surface. Excess lysimeter tubing and soil temperature/moisture wiring was pulled snuggly through the plywood cap, so there was no interference at the infiltrative surface (plywood was used for lysimeter placement and was removed before backfilling the trench). A schematic of lysimeter placement is shown in Figure 3-8 with additional detail available in Tackett (2004) and Dimick (2005).

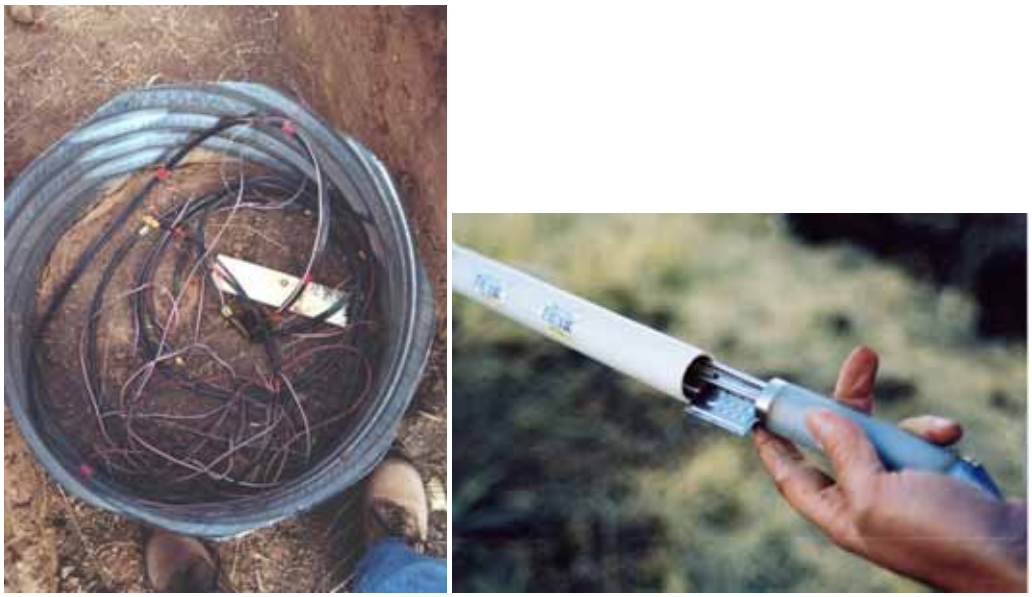


Figure 3-7
Test Cell Infiltrative Surface, With Lysimeter Tubing and Soil Temperature/ Moisture Wiring, and PVC Pipe Used to Install the Lysimeter and Soil Temperature/Moisture Probe[12]

[12]Tubing and wiring was lifted off the soil infiltrative surface during normal operation

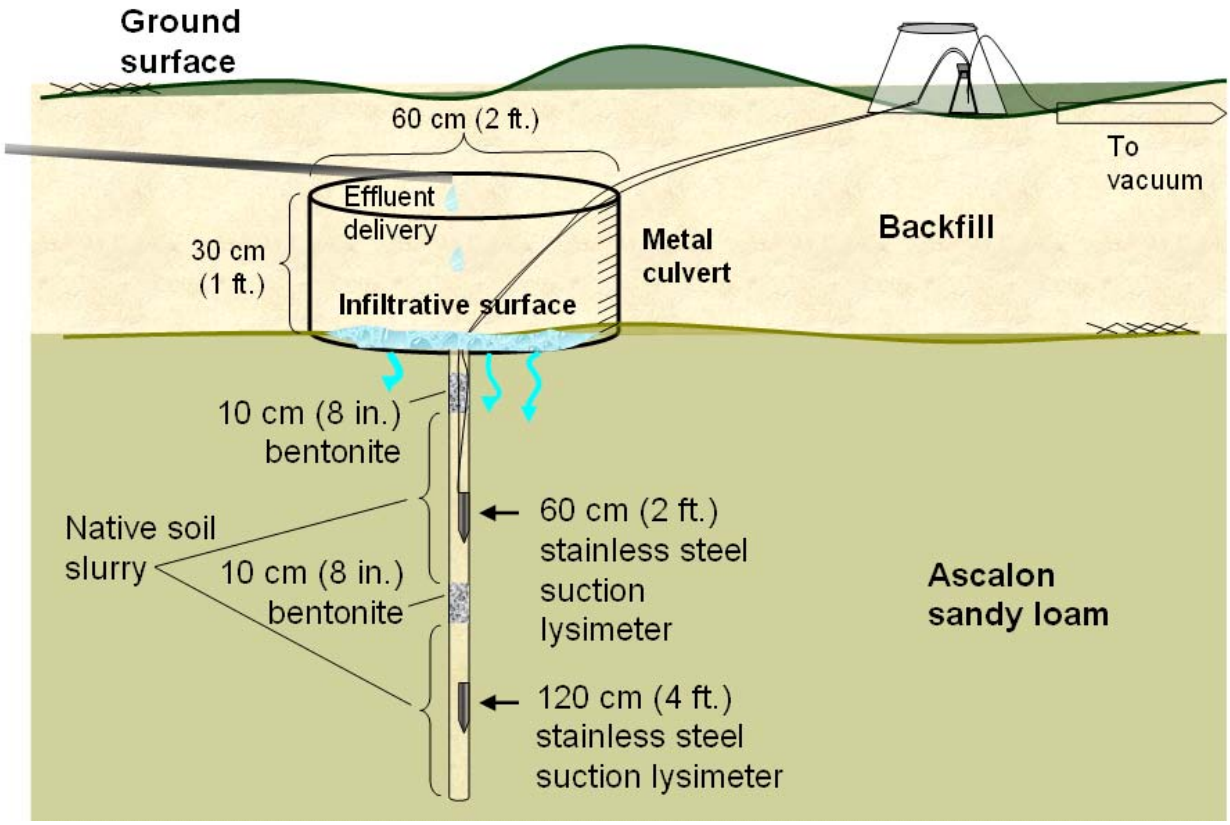


Figure 3-8
Schematic of Lysimeter and Delivery Configuration Below Infiltrative Surface[13]

These lysimeters are used as sampling devices where a vacuum is applied to facilitate sample collection from the vadose zone using a porous surface in contact with soil (Wolt 1994). The vacuum applied must be strong enough to overcome the soil moisture tension and to draw soil water present in the vadose zone into the lysimeter. The SW-074 lysimeters have a bubbling pressure of 700 millibars. This pressure is the air entry value, which is the air pressure required to force air through the thoroughly wetted porous material. The bubbling pressure is a function of pore size. The smaller the pores, the higher the bubbling pressure value. When this critical value is exceeded, the bonds attaching water to the porous material can be broken. To provide the vacuum needed for soil solution sampling, a manifold of 1.3-cm diameter PVC pipe was buried alongside the trenches and connected with flexible tubing to vacuum pumps located in rectangular irrigation boxes in the middle of each trench. Sections of PVC pipe tee off to smaller, round irrigation boxes located near each test cell, buried halfway beneath the ground surface. The round irrigation boxes house the ends of the associated test cell lysimeter tubing, which can be connected to the vacuum line. Ball valves isolate vacuum lines to test cells that are not actively being sampled. Vacuum pumps for sampling are operated between 250–300 millibars.

[13] Not to scale

3.2.5 Effluent Delivery to the Soil Test Cells

Test cells were dosed with septic tank, TFU, or MBR effluent at design hydraulic-loading rates of 2 or 8 cm/d (0.5 or 2 gpd/ft^2). Effluent was delivered to test cells in 16 equal volume doses per day. The 16 doses to the test cells occurred between 7 a.m. and 10 p.m. Each test cell with a design HLR of 2 cm/d was designed to receive approximately 364 mL of the prescribed effluent at each hourly dose. Those cells designed to receive an HLR of 8 cm/d received approximately 1,458 mL per hourly dose. The duration time for delivery of the doses was approximately 90 seconds. Effluent delivery to the soil test cells began on April 6, 2004. The MBR unit had been in operation for approximately two months before MBR effluent delivery to the test cells. Startup of the TFU coincided with effluent delivery to the soil test cells.

Effluent pumps (STEP 20 pumps) were installed in each of the effluent delivery basins. For the STE, this was the holding tank at the Mines Park Test Site; for the TFU, it was the delivery basin that received effluent from the RSV; for the MBR, it was the MBR effluent basin in the insulated research building. Effluent from each of these basins is pumped through 1.3-cm diameter buried PVC lines to a protective unit that houses the test cell delivery apparatus. Figure 3-5 shows a photograph of the protective housing. Effluent delivery lines within this housing have heat trace cables to avoid freezing. From the three effluent delivery banks, effluent flowed by gravity from the pressurized bank to the center of the respective test cell. Each effluent PVC bank was tapped with delivery orifices (1 orifice for 2 cm/d or 4 orifices for 8 cm/d) calculated to deliver the proper volume (364 mL or 1,458 mL for 2 or 8 cm/d, respectively). The effluent delivery pumps are on for 90 seconds at the top of the hour for each dose of the 16 total daily doses.

Delivery volumes were measured for an individual dose one day each week (usually triplicate measurements) in order to compare actual HLRs to the design HLR. In order to measure actual dose volumes delivered to each test cell (that is, to measure the HLR), the flexible lines from the orifice were removed from the gravity delivery line, the dose was captured, and the volume was measured. Each individual line to each test cell was measured to ensure proper design loading. The volume delivered could be altered (increased or decreased), if necessary, by adjusting a ball valve located near each of the orifice delivery banks. Results for volume of effluent delivered per dose are presented in Section 3.3.1.

3.2.6 Soil Solution Characterization

This section provides description of soil solution sampling, handling, and analyses methods.

3.2.6.1 Sample Collection

Sample collection of the soil solution was conducted using microporous stainless steel section lysimeters.

Individual lysimeter tubing is inserted into one hole of a rubber stopper with another set of tubing leading to the vacuum line. Soil solution travels up from the lysimeter by vacuum and drops into a pre-cleaned stoppered flask for sample collection (Figure 3-9).

All glassware used in the soil solution sampling was washed in phosphorus-free soap, followed by acid/base baths separated by DI water rinses. Glassware (250-mL glass Erlenmeyer flasks and 250-mL glass amber bottles) was then allowed to air dry and covered with foil until use.

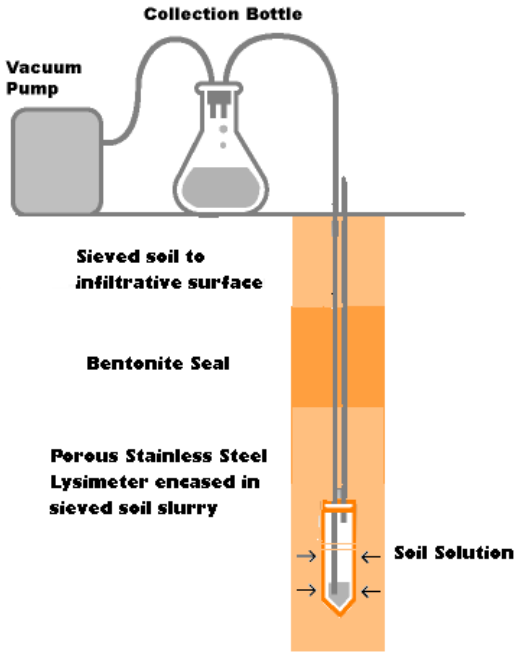


Figure 3-9
Configuration of Soil Suction Lysimeters Used for Pore Water Sample Collection[14]

The first step in sample collection was to purge the soil solution from the lysimeter and surrounding borehole to obtain a representative sample from the soil profile. This required one to two days of continuous vacuum for the lysimeters at 60 cm, two days for the 120 cm lysimeters, and from two to seven days (or more) for the 240 cm-deep lysimeters. A low vacuum (200–300 millibars) was used to be well below the air entry value of the lysimeter and to help ensure that soil pore solution was pulled horizontally from the vadose zone and that the soil did not dry out causing discontinuity in the sample. The initial purge volume (approximately 55 mL) collected was discarded and pre-cleaned flasks were then attached to the vacuum system for sample collection.

Lysimeters that yielded higher sample volumes required that the sample be collected after one day, refrigerated, and composited with the sample collected the next day. Samples were collected in glass Erlenmeyer flasks and transferred to amber glass jars for transport to the laboratory. Each lysimeter was sampled every three weeks for the first three rounds (that is, 10 out of the possible 30 lysimeters per week) and every four weeks for the last three rounds of sampling (that is, six to eight lysimeters per week). Six rounds of sample collection and analyses occurred,

[14] From http://www.soilmeasurement.com

when adequate volume could be pulled, during the first six months of operation (approximately once each month). Approximately 75 mL of sample was required from each lysimeter to perform the full suite of analyses.

3.2.6.2 Sample Analysis

The soil solution samples were transported to a CSM lab. Either analyses were completed as soon as possible, or the samples were preserved and maintained at 4 °C until all analyses could take place. Information such as sample identification number, date, time, purge and sample volume, as well as the associated analytical results, were recorded in a log book. The analyses performed, methods used, and minimum detection limit for pore water samples are presented in Table 2-7. Sample duplicates/triplicates were performed for at least 10% of the samples collected, and standard concentrations were analyzed for nitrate, total nitrogen, ammonium, phosphorus, COD, and DOC. These methods used for the soil solution samples are the same methods used for analyses of the effluent samples (see Chapter 2).

3.2.6.3 Data Analysis

Each test cell condition (for example, TFU applied at a design HLR of 8 cm/d) was replicated three times with two of the three test cells containing lysimeters (for example, TBC1 and TCC6). Analytical results were compared between the two duplicate test cells. Laboratory duplicates/triplicates from the same test cell sampling event were also run on 10% of the samples. Standards were run for pH, COD, nitrate, total nitrogen, ammonium, DOC, and total phosphorus analyses. These results were used to identify outliers and ensure that laboratory methods were accurate.

3.2.7 Soil Solids Characterization

Due to the short duration of this study, coring of the soil test cells after only six months of loading with septic tank, TFU, or MBR effluents was not deemed insightful. Therefore, to assess changes in soil properties over a longer period of operation, the soils below the infiltrative surface within the companion study at the Mines Park Test Site were characterized. These test cells were similar in design, installation, and operation to the current study and had been in operation for 13 months. Details of this companion study are presented in Appendix B. The only direct comparisons made here are between test cells operated under similar conditions as this NDWRCDP study (that is, with an open infiltrative surface and design HLR of 8 cm/d). The companion test cells were dosed continuously (44 mL/min for 16 hours each day) for 13 months with STE or tap water. A similar characterization is planned for the soil test cells in this NDWRCDP study and any results generated will be presented elsewhere.

3.2.7.1 Soil Core Collection

During the three weeks before the collection of soil cores, a multi-component surrogate tracer test (bromide and MS-2 and PRD-1 bacteriophage) was conducted. Immediately before coring an individual test cell, the ponding height was measured and an infiltration rate test completed. After the infiltration rate test, the remaining effluent ponding was slowly pumped out of the test

cell using a low flow (approximately 150 mL/min) peristaltic pump. The test cell was allowed to rest for 4 to 18 hours before coring. The rest helped ensure that there would be no residual effluent ponded on the infiltrative surface that might migrate down the sampling equipment and cross-contaminate the underlying soil samples.

Duplicate cores were collected from 12 test cells using direct-push soil coring techniques. Direct-push soil core collection with the GeoProbe™ 6610DT used the GeoProbe/Terraprobe sampling method (Figure 3-10). A 1.2 m long × 5 cm inner diameter (ID) core sampling assembly with polyethylenterephthalate (PETG) liners was used to collect continuous undisturbed samples from the infiltrative surface to 150 cm below the infiltrative surface. The core sampling assembly is made up of an outer stainless steel core barrel (5 cm ID) with inner PETG liners (5 cm ID) inserted into the core barrel. The assembly is hammered, without rotation, approximately 1 m into the ground surface. The core barrel with PETG sleeve and soil core is then retrieved to the surface.

Figure 3-10
Soil Core Collection: Mobile Direct Push Drilling Rig and Intact Soil Core Collection

Upon retrieval to the surface, the PETG liner with the intact soil core was removed from the sampler (Figure 3-10), capped, and stored at 4 °C before being transported to a CSM laboratory for analyses. A clean PETG sleeve was then placed into the inner core barrel and inserted into a clean outer core barrel. The assembly was then again advanced approximately 0.5 m. Each stainless steel core barrel was steam-cleaned after use to prevent cross-contamination between sample locations. These soil core collection methods enabled relatively intact core samples to be

aseptically collected from the test cell infiltrative surface vertically downward to a depth of 150 cm below the infiltrative surface. All samples were stored at 4 °C until laboratory analyses were performed at CSM.

3.2.7.2 Sample Analysis

Following collection in the field, the core samples were transported to a CSM laboratory for inspection and analyses. In the laboratory, the cores were carefully opened and the outer-most soil media was removed and wasted (Figure 3-11). Sub-samples of the interior of the core were then taken with sterile utensils at multiple intervals and soil sample extractions were performed for nutrients, bromide, bacteriophages, and fecal coliform bacteria. Each sample interval was based on the mass of soil required to conduct the planned analyses. In addition, the top 4 cm of the core was set aside for analysis of the composition and structure of the biozone. Sub-samples for biozone characterization were taken at the following intervals (below the infiltrative surface): 0–1, 1–2, 2–4, and 9–10 cm. Selected biozone samples were evaluated with an environmental scanning electron microscope.

Figure 3-11
Soil Sample Preparation in the Laboratory

Sample preparation of the infiltrative surface, all extractions for microbial samples, and laboratory analyses for water content were completed within 24 hours of sample collection. See Appendix C for detailed analysis methods for the soil sub-samples taken from the soil cores. All soil results are expressed per gram or kilogram of dry soil.

Table 3-4
Summary of Soil Sample Intervals and Analyses

Interval (depth below infiltrative surface)	Sample Analysis
0–6 cm	pH, soil moisture, total bacteria, TOC humic substances, soil organic matter, sodium, polysaccharide carbon, scanning electron microscope
6–10	Soil biomass or, heterotrophic bacteria, fecal coliforms, and *E. coli*[1]
7–13	Color, pH, water content, bromide, TOC, nitrate, exchangeable ammonium, available and total phosphorus[2]
13–17	Soil biomass or, heterotrophic bacteria, fecal coliforms, and *E. coli*
17–23	Color, pH, water content, bromide, TOC, nitrate, exchangeable ammonium, available and total phosphorus
27–33	Color, pH, water content, bromide, TOC, nitrate, exchangeable ammonium, available and total phosphorus
33–37	Soil biomass or, heterotrophic bacteria, fecal coliforms, and *E. coli*
42–48	Color, pH, water content, bromide, TOC, nitrate, exchangeable ammonium, available and total phosphorus
57–63	Color, pH, water content, bromide, TOC, nitrate, exchangeable ammonium, available and total phosphorus, and bacteriophage
63–67	Soil biomass or, heterotrophic bacteria, fecal coliforms, and *E. coli*
77–83	Color, pH, water content, bromide, TOC, nitrate, exchangeable ammonium, available and total phosphorus
97–103	Color, pH, water content, bromide, TOC, nitrate, exchangeable ammonium, available and total phosphorus
117–123	Color, pH, water content, bromide, TOC, nitrate, exchangeable ammonium, available and total phosphorus, and bacteriophage
123–127	Soil biomass or, heterotrophic bacteria, fecal coliforms, and *E. coli*

See Appendix C for soil sample materials and methods
[1] Biomass assays were completed on 6 test cells and heterotrophic bacteria, fecal coliform, and *E. coli* analyses were completed on the remaining 6 test cells
[2] Total P was analyzed on 6 of 24 soil cores

3.2.8 Soil Test Cell Monitoring

Hydraulic behavior of the test cells was assessed through monitoring of the effluent ponding on the infiltrative surface and measurement of individual infiltration rates. This section provides description of the methods used to assess the hydraulic behavior.

3.2.8.1 Ponding Height

As an indication of infiltration rate loss within a test cell over time, the height of effluent ponding on the test cell infiltrative surface was measured weekly during the first four months and then once every two weeks. As the infiltration rate of the test cell surface (no sidewall contribution) declines to near the applied HLR, effluent ponding develops. While initially the ponding heights are intermittent and variable, they slowly become continuous and increase in height with time.

After completion of test cell setup and before clean water delivery, a reference mark was made on the observation port casing and the distance from this reference to the infiltrative surface was measured. All ponding height measurements were made from this reference point. Ponding was measured by lowering a measuring tape, with a hook on the tip, down the observation port. When the tip of the hook breaks the surface of the effluent, the distance on the measuring tape can be recorded. All measurements are accurate to ±1 mm (approximately ±1/32 in.). Ponding measurements were taken before the 9 a.m. dose. At selected times, daily variations in ponding were also monitored.

Variability in the ponding heights is attributed to variations in the hydraulic loading and instabilities in the hydraulic properties of the biozone. Taking this variability into account and to prevent overflowing of effluent from the test cell, the “end state” of an individual test cell was operationally defined as the condition when continuous ponding heights of 20 cm or more persisted over three consecutive weeks (the culvert section sides of the test cell are about 28 cm high). At end state, effluent delivery to an 8 cm/d test cell was terminated, the infiltration rate was measured, and effluent delivery to the test cell was restarted at a reduced HLR of 2 cm/d.

3.2.8.2 Infiltration Rates

Infiltration rates were measured for each test cell prior to effluent delivery, after six months of effluent delivery, and at end state (if applicable). Infiltration rates were determined through measurement of the volume of water accepted (that is, rate of discharge) by the individual test cell (bottom area only, no sidewall contribution) over a given time.

Prior to effluent delivery, constant head infiltration rates were measured using a fabricated permeameter (Figure 3-12). The test cell was flooded to saturate the soils for approximately two hours. Care was taken during flooding to avoid any disruption to the infiltrative surface. The permeameter was then filled with tap water, sealed, and the valve opened. Once the flooded level in the test cell reached the air inlet holes, air would enter the permeameter column and water would be released to the test cell, maintaining a constant head of 2.5 cm of water on the test cell. The rate of hydraulic head change in the permeameter column with time was recorded. Estimated

rates were plotted in the field and the test continued until the measurements reached a constant rate (a minimum of nine measures were made for each test cell). Data were analyzed to determine the baseline infiltration rate of an individual test cell. Measurements greater than or equal to 75% of the rate of water exiting the air entry holes of the permeameter were removed from the dataset.

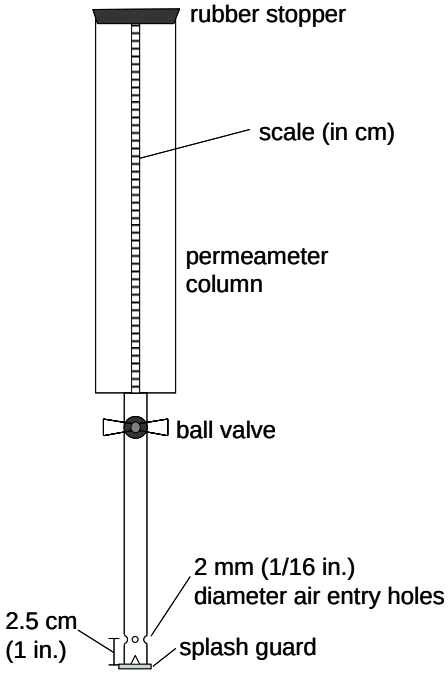


Figure 3-12
Permeameter Used During Baseline Infiltration Rate Measurements

After six months of operation, the infiltration rates were sufficiently reduced to the point that the infiltration rate was measured by monitoring the decline in ponding height within the test cell over time. This approach provides a close approximation of the constant head infiltration rate, as the change in head over time is very small (for example, less than 0.5 to 4 cm/hour [0.25 to 2 in./hour]). During the six-month infiltration rate test, delivery to the test cell was temporarily stopped to ensure the change in a known volume of effluent was monitored. For test cells with less than 5 cm of ponding, approximately 17 L of effluent was added to the test cell to raise the ponding height to 5 cm or more. The decrease in ponding height over time was recorded. For test cells with more than 5 cm of ponding, no additional effluent was added and the decrease in ponding height over time was recorded. Again, the data was plotted in the field and the test continued with addition of more effluent to the test cell as necessary until a constant infiltration rate was observed.

3.2.9 Tracer Testing

Tracer tests were conducted at two time points during test cell operation: before effluent delivery and after three months of effluent delivery. Prior to effluent delivery to the test cells, each test cell was dosed with City of Golden tap water at the design-loading rate for that cell. The delivery of tap water (“clean water”) to the test cells allowed for background characterization of the delivery lines and apparatus and pre-startup characterization of each individual test cell. The first

tracer test occurred after 29 days of clean water delivery. The pre-startup tracer test surrogates and tracers were added to clean water and delivered to all test cells (tracer was not delivered to the TFU or MBR).

The second tracer test was conducted after three months of effluent delivery to the test cells. During this tracer test, the same surrogates and tracers (excluding rhodamine) were added to the STE holding tank for delivery to the TFU and MBR prior to application to the soil.

3.2.9.1 Tracer Characteristics

The tracers included in the tests were:

- Bromide
- Rhodamine WT
- MS-2 and PRD-1 Bacteriophages

3.2.9.1.1 Bromide

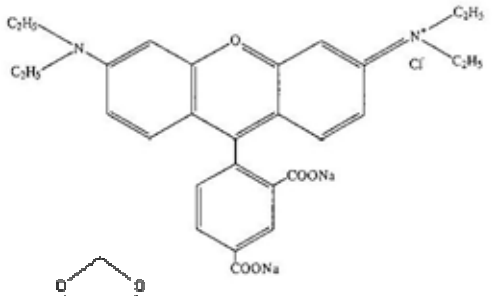
Bromide (Br^-) used in the tracer test was added to the STE (or to tap water, in the case of the clean water tracer test) as potassium bromide (KBr). Bromide serves as a conservative tracer representative of the water movement through soil, though some diffusion from mobile to immobile water may occur. The target bromide concentration for each tracer test is presented in Table 3-5.

3.2.9.1.2 Rhodamine WT

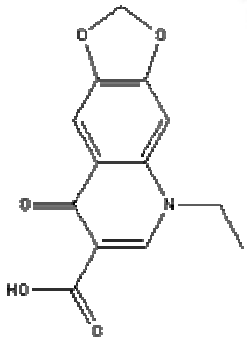
Rhodamine WT (RWT) is a non-conservative tracer used in this study as a surrogate for organic chemicals. RWT is a florescent dye commonly used in surface and groundwater tracer studies because of its relatively low cost, strong fluorescence, high diffusivity, and benign character in the environment (Kilpatrick and Wilson 1989). RWT (Figure 3-13a) is a large organic molecule with positive and negative ionic functional groups. These ionic groups increase the water solubility (180 g/L, Gaspar 1987), allowing for quick and complete dissolution into water. The large size of the molecule and presence of both positive and negative functional groups resembles the structure of some pharmaceutical compounds, such as oxolinic acid (Figure 3-13b) and pipemidic acid (Figure 3-13c). These pharmaceuticals belong to the class of quinolones that are used to treat a wide variety of bacterial infections in humans, as well as to treat livestock and fish in the aquaculture industry (Miao *et al.* 2004).

Numerous studies in the last few decades have reported the presence of residual amounts of pharmaceuticals, consumer product chemicals, and other organic wastewater contaminants in surface waters, groundwater, and even finished drinking water from municipal drinking water treatment plants (Kolpin *et al.* 2002; Belfroid *et al.* 1999; Sekela *et al.* 1999; Ternes *et al.* 1999a and b; Desbrow *et al.* 1998). Some studies suggest wastewater as a primary source of these organic wastewater contaminants but few have quantified their occurrence and fate in OWSs and the aquatic environment to which the treated wastewater is recharged.

(a)



(b)



(c)

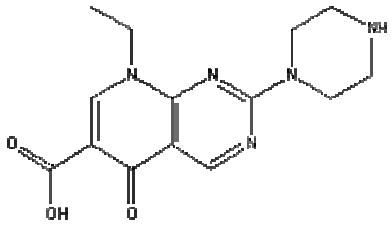


Figure 3-13
Structure of RWT and Compounds for Which it May Be a Surrogate Tracer; (a) RWT, (b) Oxolinic Acid, and (c) Pipemidic Acid

The occurrence of a suite of organic wastewater contaminants in the Mines Park Test Site STE has been quantified by DeJong *et al.* (2004), identifying the presence of a number of compounds including caffeine, cholesterol, the non-ionic surfactant 4-nonylphenol, and the antimicrobial agent triclosan. To aid in understanding the fate of these compounds and other organic wastewater contaminants through a wastewater treatment system (soil component of an OWS) with recharge to the aqueous environment, RWT was used as a surrogate tracer.

3.2.9.1.3 MS-2 and PRD-1 Bacteriophages

Two bacteriophages, MS-2 and PRD-1, were used in this study as models for human pathogenic enteric viruses. MS-2 is an icosahedral single-stranded RNA coliphage with an average diameter of around 25 nm and an isoelectric point (pH_{iep}) of 3.9 (Powelson *et al.* 1990). PRD-1 is an icosahedral lipid phage with a diameter of 62 nm and a pH_{iep} of less than 4.5. The host of PRD-1 is *Salmonella typhimurium* and the host for MS-2 is *E.coli* (Ryan *et al.* 1999; Bales *et al.* 1991). Liquid samples were analyzed for these bacteriophages following the plaque-forming unit (pfu) technique as described by Adams (1959). For this assay, all samples were serially diluted in phosphate buffered saline (PBS), plated with the bacterial host on a layer of agar, and incubated overnight at 37 ^{o}C. Plates were enumerated by counting plaques formed in the host lawn. MS-2 and PRD-1 bacteriophages and host bacteria were obtained from the United States Geological Survey (USGS) in Boulder, Colorado. Bacteriophages were added to applied effluents (such as septic tank, TFU, or MBR effluents) at target concentrations of approximately 10^7 pfu/mL.

3.2.9.2 Tracer Delivery

During the clean water tracer test, tap water from the City of Golden in three holding tanks was spiked with bromide, RWT, and the two bacteriophages. Tracer was added to each of the delivery basins and then mixed using PVC pipe as a stirring rod. The spiked water was then delivered (at design dosing and delivery rates) to the infiltrative surface of the soil test cells (the TFU and MBR units did not receive tracer). These cells had been recently installed and effluent had not yet been applied to them. During the tracer test after three months of operation, bromide and the two bacteriophages were added to the STE for delivery to the TFU and MBR prior to the soil test cells. The target concentrations for each tracer added are presented in Table 3-5.

Table 3-5
Conditions and Design of Tracer Tests

	Tracer Test I	**Tracer Test II**
Effluent	Clean water	STE, TFU, or MBR
Start of Test	Prior to effluent delivery	After 3 months of effluent loading
Start Date	February 26, 2004	July 6, 2004
Duration of Tracer Addition (Days)	22	23
Target Tracer Concentrations		
• Bromide (mg-Br/L)	1500	2500
• RWT (μg/L)	2000	None added
• MS-2 and PRD-1 (pfu/mL)	10^7	10^7

Bromide was added in the form of potassium bromide salt. The necessary amount of potassium bromide was calculated, measured, and any large clumps were broken up before adding it to the tanks. If necessary, clumps that were more resistant were placed in a clean bucket and dissolved in water before being added to the tanks to ensure dissolution of the salt. During the clean water tracer test, a stock solution of liquid RWT was diluted with deionized water and added to the clean water to achieve average influent concentrations around 2,000 μg/L. For 22 consecutive days, RWT was dosed to the test cells at an average concentration of:

- 2,341 μg/L (ranging from 1,228 to 3,462 μg/L, 706 SD, n=27) in the MBR tank
- 1,888 μg/L (ranging from 728 to 3,864 μg/L, 868 SD, n=23) in the TFU tank
- 2,335 μg/L (ranging from 1,073 to 4,965 μg/L, 1,123 SD, n=31) in the STE tank

RWT was not added during the second tracer test.

Bacteriophages (MS-2 and PRD-1) were added to the tanks by taking the appropriate volume of high titer stocks into 50 mL conicals and then adding PBS up to a larger volume (50 mL). This

sample was poured into each tank and the conicals were rinsed (following tracer addition to tanks) with DI, and rinsate was delivered to the tanks.

3.2.9.3 Sample Collection

Following addition of tracers to each of the delivery basins, daily samples were taken (in triplicate) for quantification and verification of concentrations of added tracers. With daily addition of tracers, three samples were taken from delivery basins before more tracer was added (representing a 24-hour time point) and immediately after tracer addition and mixing.

Samples were collected using the stainless steel lysimeters placed below the infiltrative surface. The vacuum apparatus that allows for lysimeter sample collection was continuously on, allowing for uninterrupted soil water sample collection. Every 24 or 48 hours, entire samples were taken from the Erlenmeyer collection flasks. The volume of the sample collected and the time of collection were recorded. Then samples were placed in a cooler for transport to the laboratory for subsequent analysis for added tracers and surrogates.

During the clean water tracer test, lysimeter samples were taken every 24 hours for the first month of sample collection. After that point, samples were taken every 48 hours (with sample collection occurring over that entire time interval). Periodic samples are collected weekly or monthly since completion of the clean water tracer test, depending on the activity within the cell. This collection is done to monitor additional RWT tracer breakthrough. During the three-month tracer test, all lysimeter samples were collected continuously for 48 hours (48-hour composite sample).

During sample collection, all sample volumes were measured and recorded. Samples for laboratory analysis for bromide and bacteriophages were collected in sterile, 50-mL conicals and immediately placed on ice. Since RWT undergoes first-order decay in the presence of light, a subset of the sample volume was transferred to 125-mL amber bottles for RWT analysis. RWT samples were stored at room temperature and were kept in the dark until analysis was completed. After sample collection, Erlenmeyer flasks used for lysimeter sample collection were washed at the point of collection. Washing included disinfecting with 70% ethanol followed by triple-rinsing with deionized water.

3.2.9.4 Sample Analysis

Samples were analyzed for bromide using an ion-selective electrode following manufacturer's specifications. Standard calibration curves were generated daily from stocks of bromide solution. The correlation between electrode mV response and the standard concentration (in mg-Br/L) was used to determine the bromide concentration. The electrode was recalibrated as recommended by the manufacturer if the correlation (r^2) was less than 0.95 or if mV measurements were observed to drift during analysis.

RWT samples were analyzed for fluorescence using a Turner 800 Fluorometer (546 nm excitation filter, greater than 570 emission filter). A standard curve ranging from 0.1 μg/L to 500 μg/L was developed from a stock solution of RWT to determine background fluorescence levels

and dye concentrations in field samples. Fluorescence values obtained on RWT samples were converted to concentration using a linear regression equation of a standard curve of known concentrations.

Bacteriophages were measured in the samples following the PFU technique as described by Adams (1959). For this assay, all samples were serially diluted in PBS, plated with the bacterial host on a layer of agar, and incubated overnight at 37 ^{o}C. Plates were enumerated by counting plaques formed in the host lawn. MS-2 and PRD-1 bacteriophages and host bacteria were obtained from the USGS in Boulder, Colorado.

3.2.9.5 Data Analysis

Time series data were plotted to present the breakthrough curves for each of the added tracers. Previous studies have shown that time to 50% breakthrough of bromide taken from time series graphs is representative of 50% values delineated from moment analysis (Beach 2001). An estimated time to 50% bromide breakthrough is used as a benchmark for comparison of vadose zone travel times for each of the tracer tests.

For bacteriophages, the percent removal was calculated using the following equation:

$$100 \times \frac{\left[(V_t)(C_o)\right] - \left[\sum_{j=1}^{n}(V_j)(C_j)\right]}{\left[(V_t)(C_o)\right]} = \%\ \text{Removal} \qquad (3.1)$$

where

V_t=total volume of the dose containing bacteriophages (mL)
C_o=concentration of bacteriophage added to the dose volume (pfu/mL)
V_j=volume of column outflow collected over a sampling time (mL)
C_j=concentration of viruses measured in the column outflow (pfu/mL)
n=number of outflow samples collected

3.3 Results

This section provides results of the characterization, effluent HLRs, soil hydraulic performance, and tracer testing.

3.3.1 Effluent Composition and Hydraulic Loading Rates

During the clean water tracer test, delivery rates varied due to an inferior delivery/dosing design (Figure 3-14). There was a miscalculation of the orifice size and pump requirements for accurate and reproducible delivery. This problem was overcome with a new orifice assembly. In addition, high head effluent pumps were installed prior to effluent delivery. As a result, comparison of the

behavior of the test cells by HLR during the initial tracer test was complicated. Whenever possible, data presented for clean water delivery will include measured HLR.

The composition of effluent applied is presented in Section 2.5.2. The design HLR for the effluents was either 2 or 8 cm/d. These design HLRs are 3.2 and 12.7%, respectively, of the measured K_{sat} for the test cells (see Section 3.3.2.2).

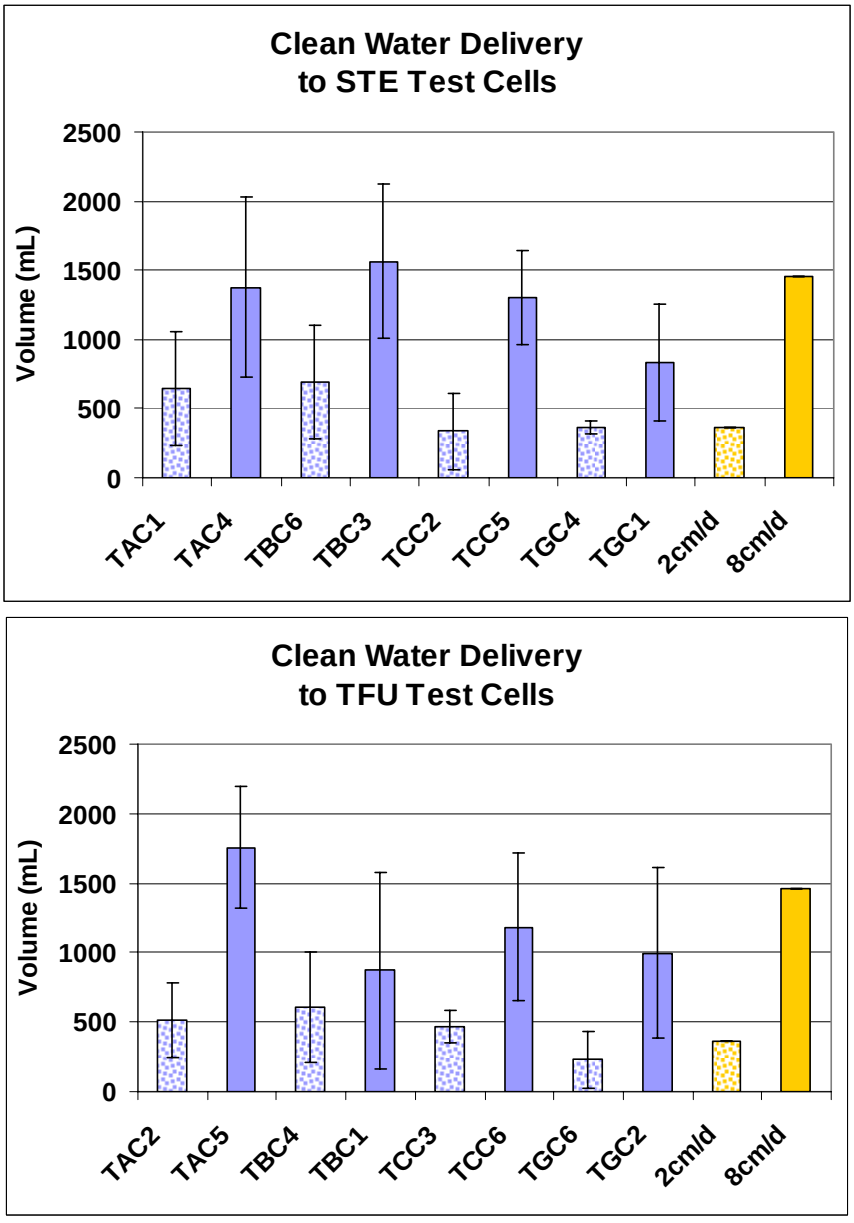


Figure 3-14
Average Measured Volume for an Individual Dose (16 Doses Per Day) to Test Cells During Clean Water Delivery[15]

[15] Open bars represent 2 cm/d design HLR; filled bars represent 8 cm/d design HLR. The last two bars represent design volume per individual dose (+/- one standard deviation).

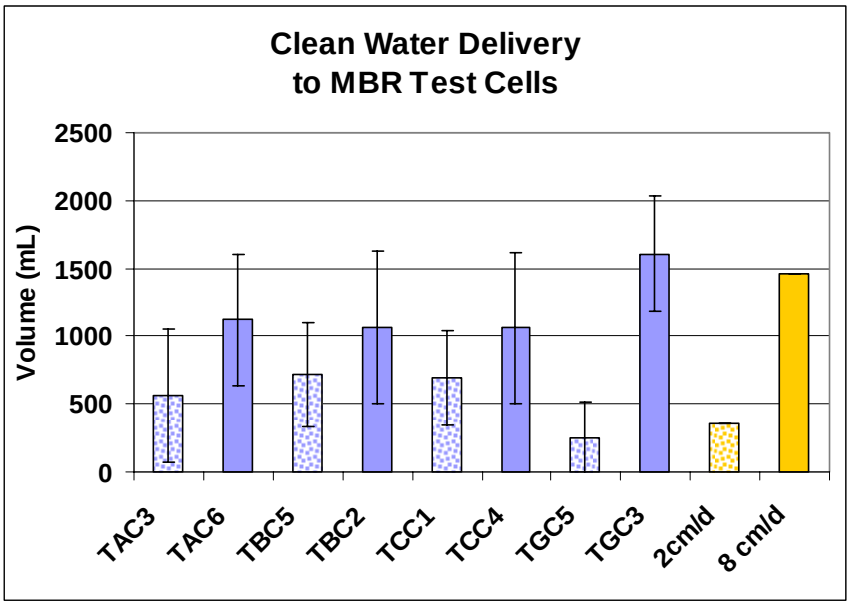


Figure 3-14
Average Measured Volume for an Individual Dose (16 Doses Per Day) to Test Cells During Clean Water Delivery (Cont.)[16]

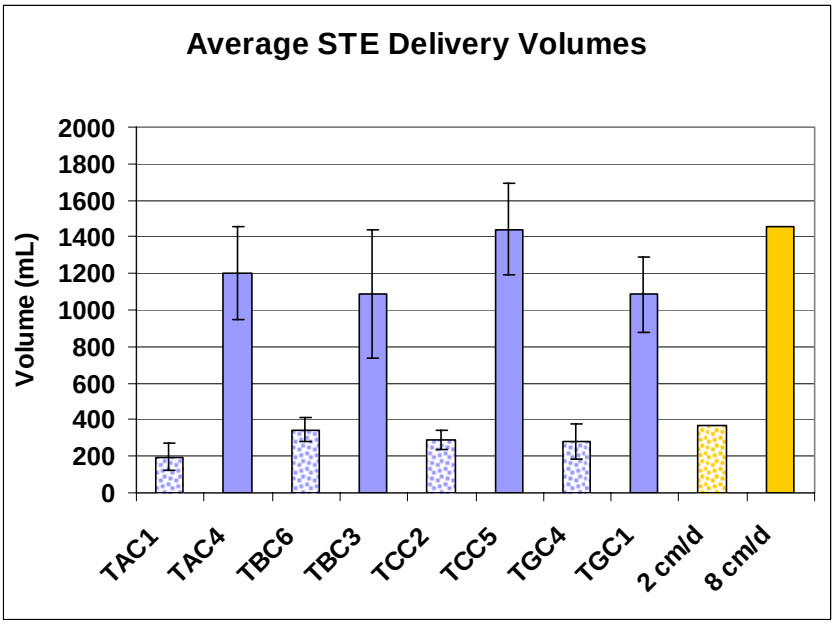


Figure 3-15
Average Measured Volume for an Individual Dose (16 Doses Per Day) for Septic Tank, TFU, and MBR Effluent to Test Cells During Six Months of Operation (April–October 2004)[16]

[16] Open bars represent 2 cm/d design HLR; filled bars represent 8 cm/d design HLR. The last two bars represent design volume per individual dose (+/- one standard deviation).

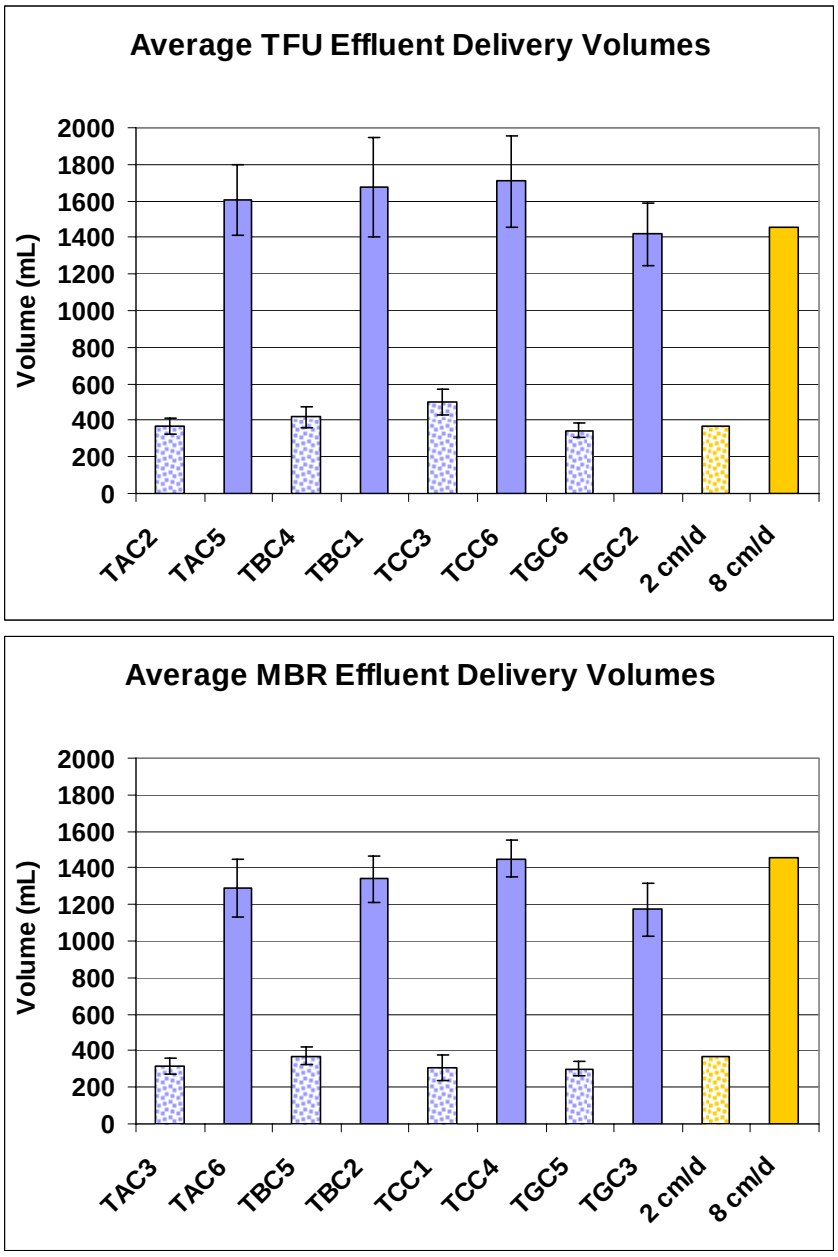


Figure 3-15
Average Measured Volume for an Individual Dose (16 Doses Per Day) for Septic Tank, TFU, and MBR Effluent to Test Cells During Six Months of Operation (April–October 2004) (Cont.)[17]

Figure 3-15 presents measured delivery values for each effluent for each design HLR. Values shown represent volume delivered during individual dosing occurrence as compared to design. Actual HLRs based on the measured delivery volumes were 1.5 and 6.8 cm/d for STE, 2.3 and 9.1 cm/d for TFU, and 1.8 and 7.4 cm/d for MBR.

[17] Open bars represent 2 cm/d design HLR; filled bars represent 8 cm/d design HLR. The last two bars represent design volume per individual dose. (+/- one standard deviation).

3.3.2 Soil Hydraulic Performance

This section provides the results of the effluent ponding monitoring and the infiltration rate measurements.

3.3.2.1 Ponding Heights

Ponding heights were routinely measured as indicators of reductions in the soil acceptance rates of the test cell infiltrative surfaces (no sidewall contributions). As the soil acceptance rates declined, incipient to continuous ponding was observed. Continuous effluent ponding within an individual test cell indicated the soil acceptance rate was possibly near or below the daily HLR. Variations within ponding heights of an individual test cell are due to method of delivery (dosed once every hour), actual delivery rate, time of day, and biozone fluctuations and instabilities.

Table 3-6 presents a summary of effluent ponding heights for each test cell measured at one, three, or six months. If no value is present, there is no ponding. A ponding value of < 1.3 cm means ponding is present, but the method of measurement does not enable accurate readings below this value.

Table 3-6
Summary of Effluent Ponding Trends

Effluent, Design HLR	Test Cell Identification	Actual HLR	Depth of Effluent Ponding (cm)		
			1 month	3 month	6 month
MBR, 2 cm/d	TAC3	1.7			
	TBC5	2.0			
	TCC1	1.6	<1.3	<1.3	<1.3
TFU, 2 cm/d	TAC2	2.0			<1.3
	TBC4	2.3			
	TCC3	2.7	<1.3	<1.3	<1.3
STE, 2 cm/d	TAC1	1.1	<1.3		
	TBC6	1.9	<1.3		<1.3
	TCC2	1.6	<1.3		
MBR, 8 cm/d	TAC6	7.0			
	TBC2	7.3		2.1	3.8
	TCC4	7.9	<1.3	1.1	2.5
TFU, 8 cm/d	TAC5	8.8	<1.3		<1.3
	TBC1	9.2	10.9	12.5	27.9[a]
	TCC6	9.4	1.7	<1.3	<1.3
STE, 8 cm/d	TAC4	6.6	14.3	1.6	1.3
	TBC3	6.0	27.9[a]	28.3[a]	27.8[a]
	TCC5	7.9	5.1	1.8	1.8

[a] Test cells have reached end state

No value=no ponding; less than 1.3 cm=ponding present, cannot measure below this value

For the 2 cm/d design-loading rate, incipient ponding was observed within the first week of effluent delivery. However, continuous ponding was not evident until after approximately four months of effluent delivery. At the end of six months of operation, four test cells were continuously ponded at less than 1.3 cm:

- 1 (of 3) MBR test cells
- 2 (of 3) TFU test cells
- 1 (of 3) STE test cells

For the 8 cm/d design-loading rate, incipient ponding was again observed within the first week of operation (Figure 3-16). However, after six months of operation, all but one MBR test cell were continuously ponded at approximately 3.5 cm, all TFU cells were at less than 1.3 cm, and all STE cells were ponded at approximately 6.1 cm. Two test cells reached the predefined "end state" (continuous ponding heights of 20 cm or more over three consecutive weeks) within the first and fourth months of operation (STE test cell TBC3 and TFU test cell TBC1, respectively). These cells reaching the "end state" is attributed to the lower-than-average baseline infiltration rate measured within these two test cells (see Section 3.3.2.2). However, the MBR test cell with a similarly low baseline infiltration rate (TBC2) had not reached "end state." After six months of operation, TBC2 had average ponding heights of approximately 3 cm.

A set of four test cells began receiving tap water at either 4 or 8 cm/d in May 2003 as part of the companion study. Incipient ponding was first observed in the test cells receiving clean water at a design HLR of 8 cm/d after six months of operation with continuous ponding of approximately 3 cm after 12 months (ponding depth had not increased at 23 months of operation). Incipient ponding was not observed in the test cells receiving clean water at a design HLR of 4 cm/d after nearly 2 years (23 months) of operation.

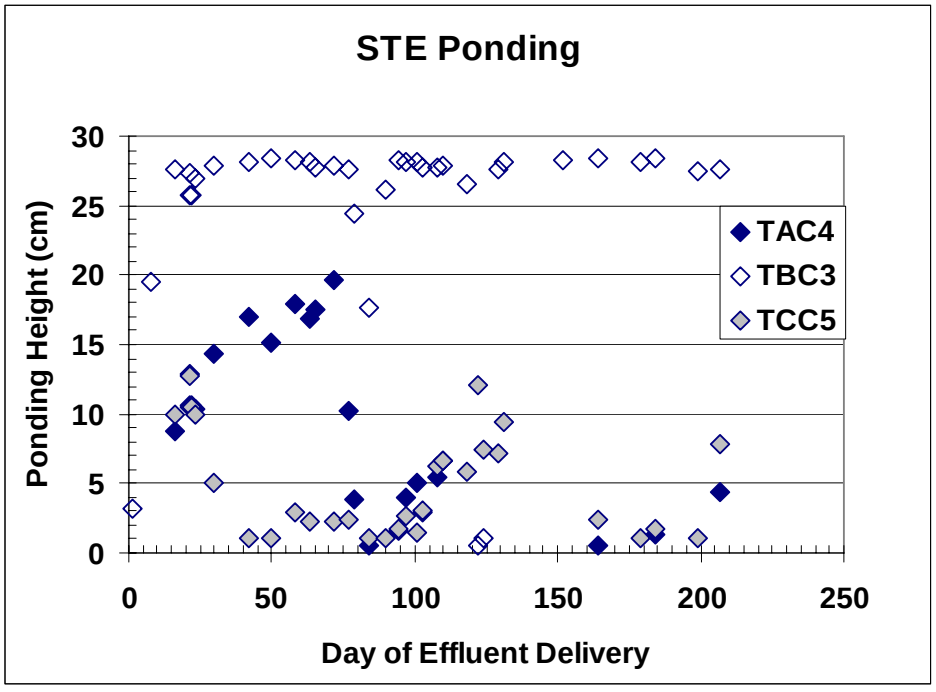
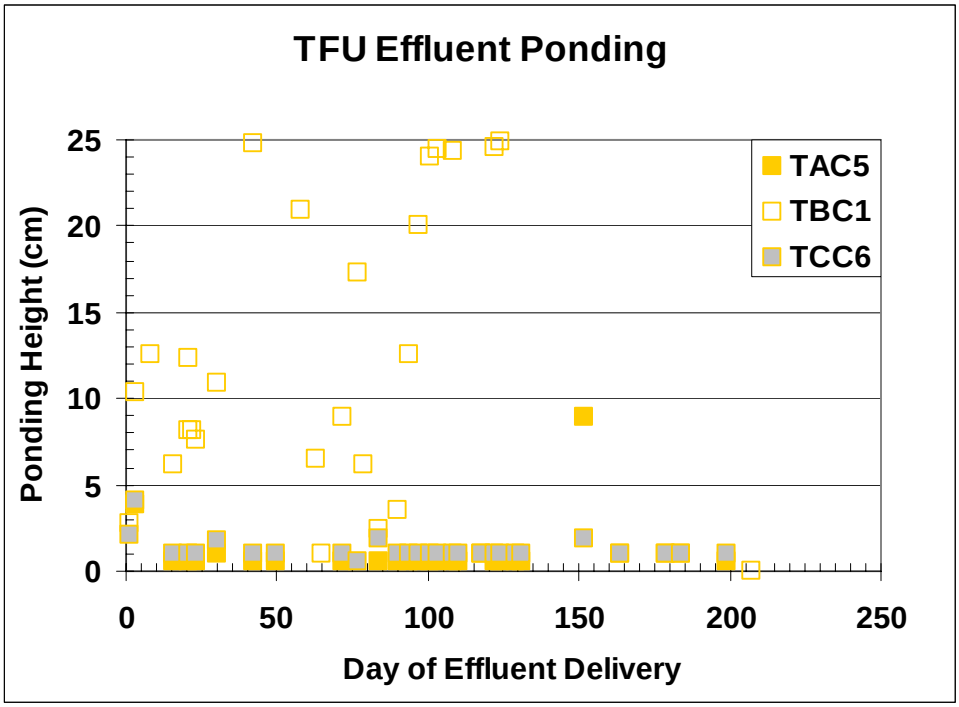


Figure 3-16
Effluent Ponding Trends for 8 cm/d Design HLR



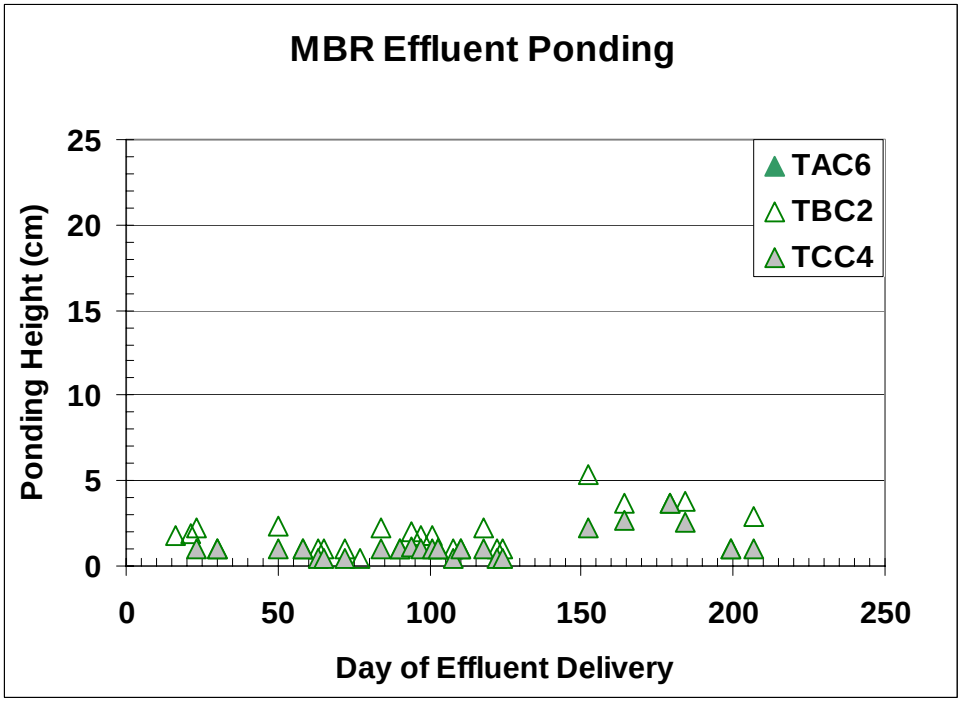


Figure 3-16
Effluent Ponding Trends for 8 cm/d Design HLR (Cont.)

3.3.2.2 Infiltration Rates

A total of 606 constant head baseline infiltration rates were measured. They ranged from 4.6 to 195 cm/d. The average baseline infiltration rate across the test site was 62.9 cm/d (median value=59.5; standard deviation=34.5 cm/d; coefficient of variance (CV) = 0.548). As previously mentioned, at least nine tests were completed within each test cell. Average baseline infiltration rates for individual test cells ranged from 23.2 to 158 cm/d (Figure 3-17).

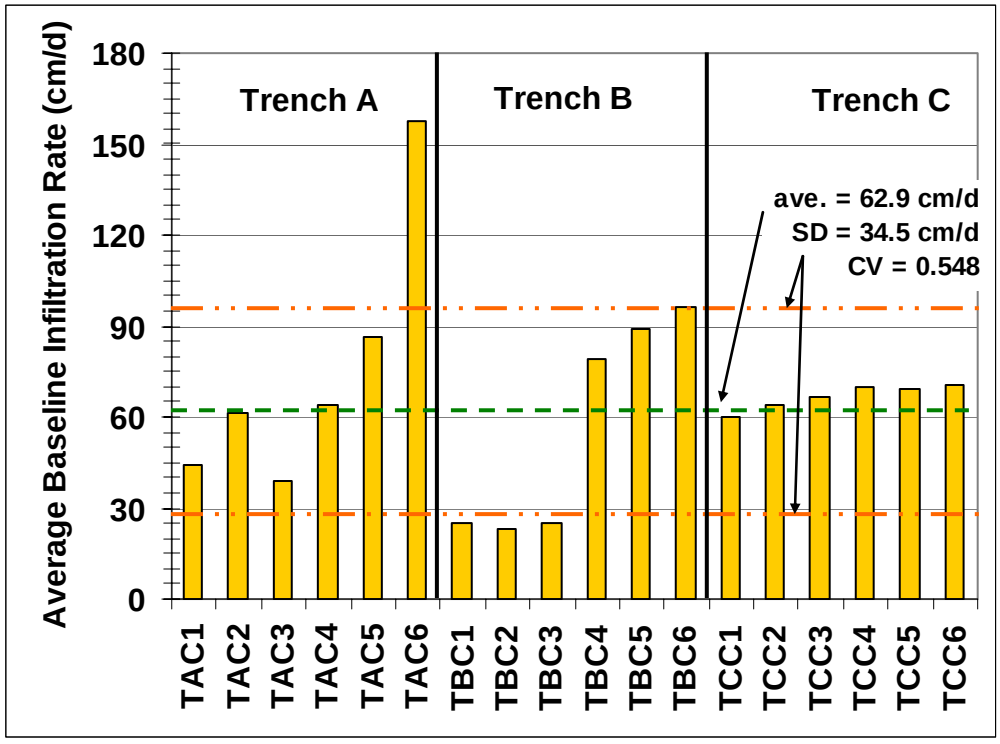


Figure 3-17
Average Constant Head Baseline Infiltration Rates

Due to the construction of the test cell, the infiltration rate is representative of the bottom surface area only (no sidewall infiltration occurs). The test cells were saturated prior to infiltration rate measurements, and the tests were repeated until the measurements reached equilibrium (Section 3.2.8.2). Therefore, the constant head baseline infiltration rate should be approximately equivalent to the K_{sat} of the soil prior to effluent delivery.

Due to heterogeneities in the soil, infiltration rates and K_{sat} are often referred to as highly variable data. Warrick (2003) reports typical coefficient of variance ranges for K_{sat} and infiltration rate of 0.48 to 3.2 and 0.23 to 0.97, respectively. While there is inherent variability, the CV for the baseline infiltration rates measured (CV = 0.548) for this study is within the typical range and suggests relatively uniform conditions across the site.

Figure 3-18 shows the baseline infiltration rates are nearly normally distributed, although a positive skew (biased toward the higher infiltration rates; data tails toward the positive standard deviations) and significant kurtosis (biased toward the average value; data is too "tall") is present.

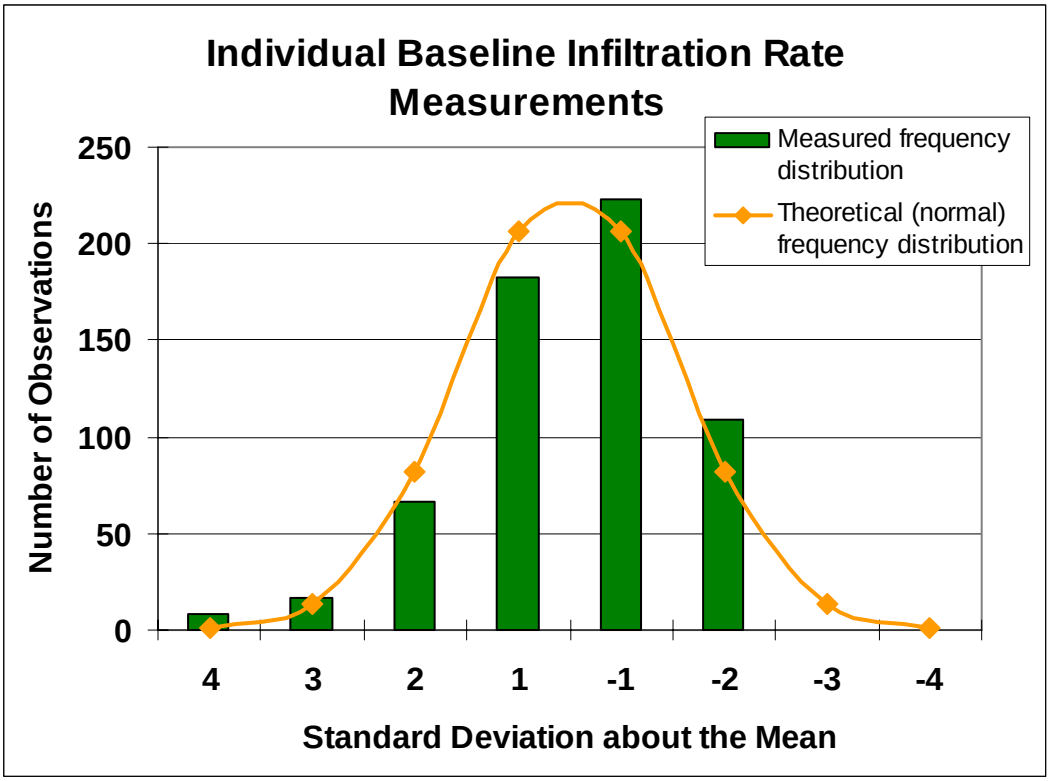


Figure 3-18
Histogram of Baseline Infiltration Rates Compared to Theoretical Distribution

Baseline infiltration rates measured in the southern portion of trench B (TBC1, TBC2, and TBC3) are less than 50% of the average baseline infiltration rate across the site. The experimental layout randomly called for 8 cm/d design-loading rate to this portion of trench B that is nearly 33% of the clean water infiltration capacity. While no attempt was made to modify the experimental layout, the lower-than-average infiltration rates were noted and the test cells may not be true replicates of the experimental condition.

After six months of effluent delivery, infiltration rates were again measured to assess potential changes in the soil infiltration capacity, which may be poorly captured by the ponding height measurements. For each individual test cell, the average baseline infiltration rate for that test cell was compared to its six-month infiltration rate to estimate the percent reduction (IR_t/IR_o). Infiltration rates after six months of operation were reduced to 11 to 97% of the baseline infiltration rate.

Figure 3-19 presents the average infiltration rates of the three replicate test cells for each condition, plus the minimum and maximum values (as summarized in Table 3-7). At two locations, the six-month infiltration rate was higher than the baseline infiltration rate. This rate is attributed to error in the test method and variability in the biozone development and behavior at TAC2. The cause for the significant increase in infiltration rate for TAC3 (nearly double the baseline infiltration rate) is unclear and was screened from the average infiltration rate as shown in Figure 3-19.

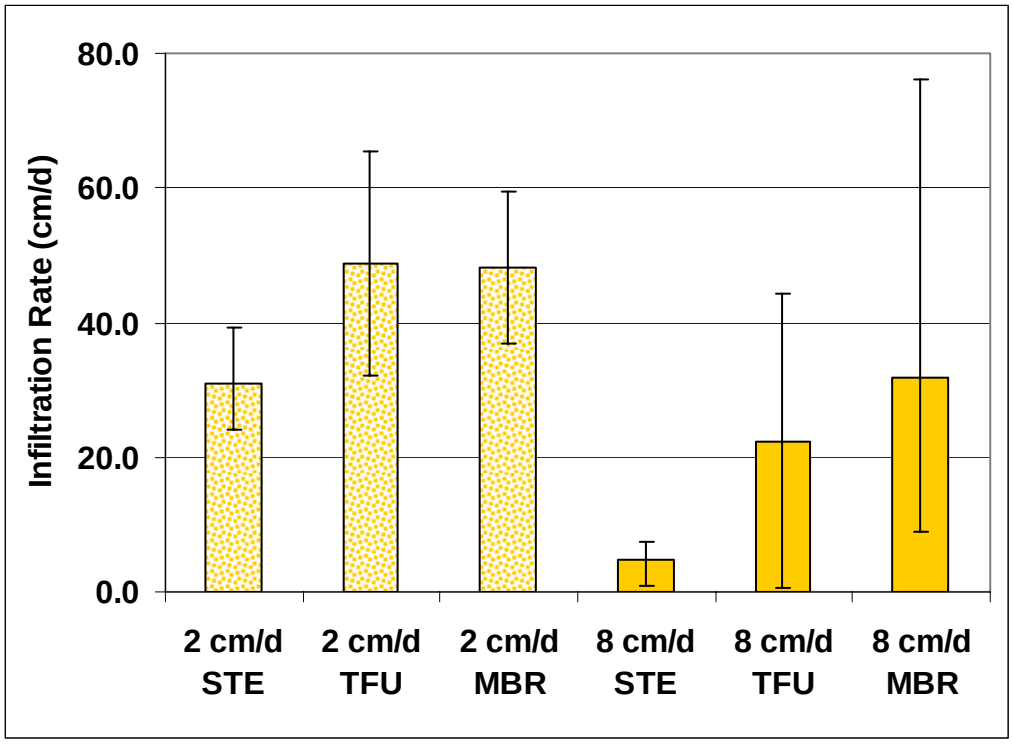


Figure 3-19
Average Infiltration Rates (cm/d) After Six Months of Effluent Delivery[18]

On average, the MBR infiltration rates were reduced by 36 and 66% for the 2 cm/d and 8 cm/d design HLRs, respectively (TAC3 infiltration rate was excluded from this average value). The TFU infiltration rates were reduced by 45 and 72% for the 2 cm/d and 8 cm/d design HLRs, respectively. The STE infiltration rates were reduced by 48 and 92% for the 2 cm/d and 8 cm/d design HLRs, respectively (Figure 3-20).

Error bars presented on Figure 3-20 indicate that after six months of operation, a significant difference in infiltration rate loss is apparent between some conditions (for example, STE at 2 cm/d compared to STE at 8 cm/d), but not between all conditions (for example, TFU at 8 cm/d compared to MBR at 8 cm/d). Continued effluent delivery and monitoring are required.

[18] Bars indicate high and low infiltration rate from all three replicate test cells, excluding TAC3

Table 3-7
Summary of Infiltration Rates

Effluent Applied	Design HLR (cm/d)	Test Cell Identification	Actual HLR (cm/d)	Baseline IR_o (cm/d)	6-Month IR_t (cm/d)	IR_t/IR_o
MBR	2	TAC3	1.7	38.8	66.1	1.71
		TBC5	2.0	89.0	59.6	0.67
		TCC1	1.6	60.0	37.0	0.62
TFU	2	TAC2	2.0	61.4	65.5	1.07
		TBC4	2.3	79.4	48.8	0.62
		TCC3	2.7	66.3	32.2	0.48
STE	2	TAC1	1.1	44.3	39.3	0.89
		TBC6	1.9	95.9	29.5	0.31
		TCC2	1.6	64.2	24.0	0.37
MBR	8	TAC6	7.0	157.9	76.1	0.48
		TBC2	7.3	23.1	8.9	0.38
		TCC4	7.9	70.0	10.9	0.16
TFU	8	TAC5	8.8	86.4	44.2	0.51
		TBC1	9.2	25.2	0.7	0.03
		TCC6	9.4	70.6	22.1	0.31
STE	8	TAC4	6.6	63.7	7.5	0.12
		TBC3	6.0	25.3	0.9	0.04
		TCC5	7.9	69.5	5.7	0.08

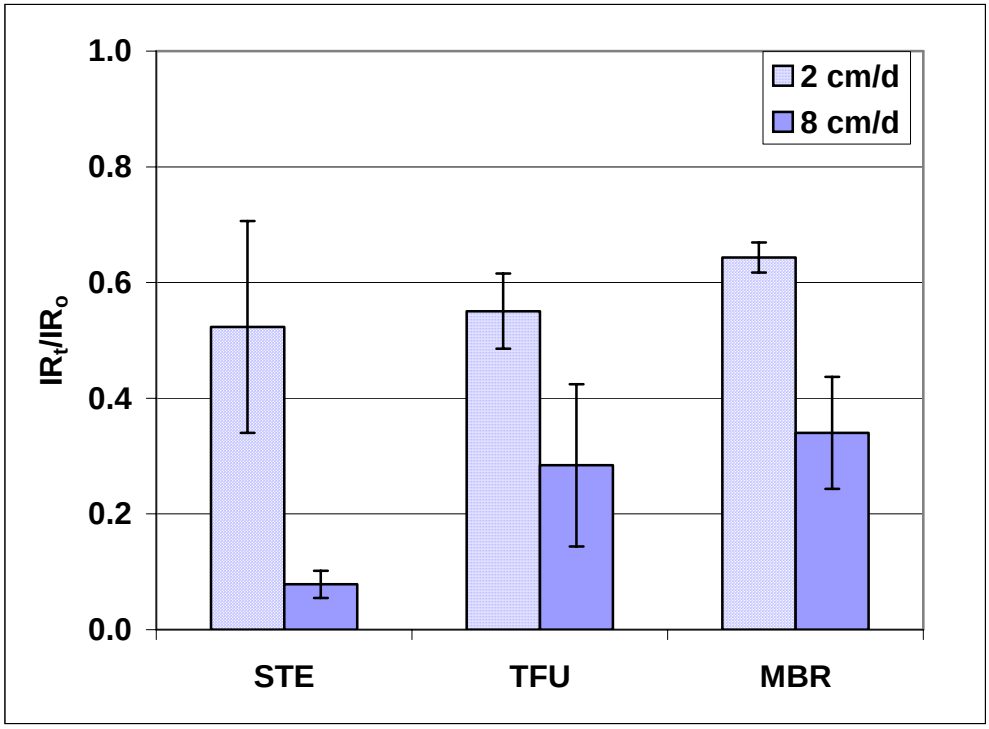


Figure 3-20
Ratio of Infiltration Rate Loss Compared to Baseline Infiltration Rate (IR_o) After Six Months of Effluent Delivery (IR_t) (Excluding Rates That Increased Between Baseline and Six Months)[19]

3.3.3 Vadose Zone Travel Times

Results of the tracer tests are provided in this section.

3.3.3.1 Clean Water Bromide Tracer Test

The clean water tracer test was conducted to gain an understanding of the travel times to the lysimeters prior to effluent dosing, and for comparison of duplicate test cell conditions. As previously mentioned, an estimated time to 50% bromide breakthrough is used as a benchmark for comparison of vadose zone travel times for each of the tracer tests. The estimated travel time provides insight into the hydraulic behavior of the test cell before effluent is applied. It also enables comparison of changes within the test cell after effluent delivery. For example, similar travel times measured during the clean water tracer test indicate homogeneity between test cell replicates and across the site. Alternatively, an increase in measured estimated travel times within the same individual test cell (or within replicate test cells for a given effluent quality) after a period of effluent delivery (three months) indicates a relative change in the hydraulic behavior attributed to effluent delivery.

[19] Error bars are +\- one standard error

Due to the inconsistent delivery volumes during clean water delivery as described earlier (see Figure 3-14), it was difficult to compare duplicate test cells. However, an overall understanding of the pre-startup vadose zone travel times was obtained for a range of HLRs. Figure 3-21 presents representative breakthrough curves for the soil test cells that had baseline infiltration rates within 5% of the average baseline infiltration rate (63.3 cm/d ± 3 cm/d). These cells received clean water at near the design HLR of either 2 (TCC2) or 8 cm/d (TAC4).

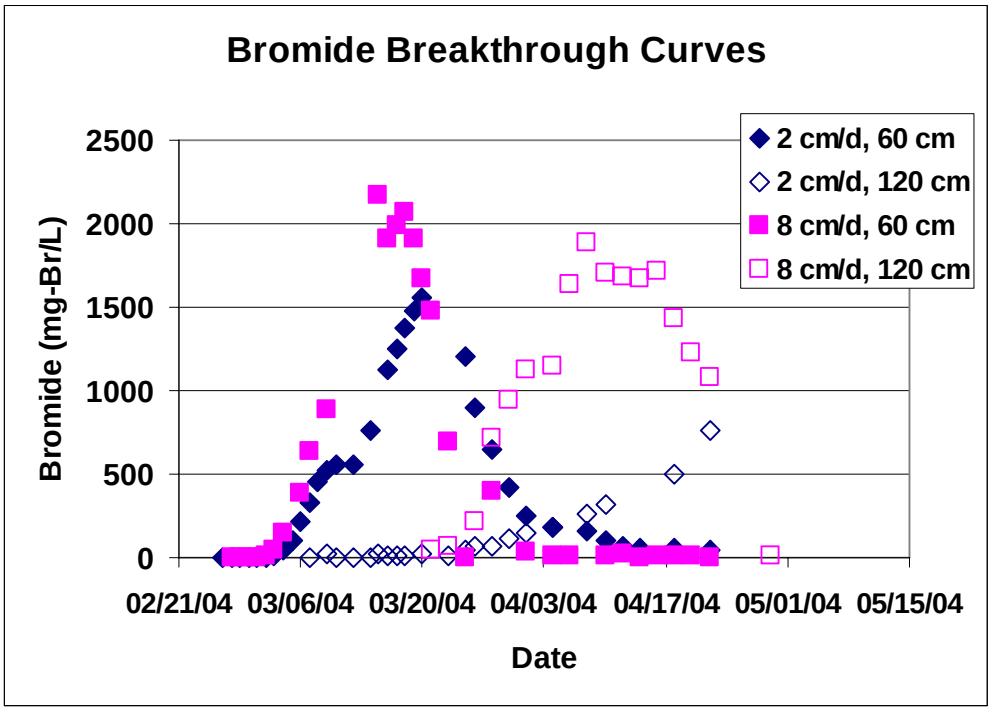


Figure 3-21
Representative Bromide Breakthrough Curves During the Clean Water Tracer Test

From these representative test cells, an estimated average travel time (based on 50% bromide breakthrough) of 18 days to the 60 cm lysimeter was observed, with a longer average time of 43 days to the 120 cm lysimeter for the 2 cm/d design HLR (Figure 3-22). Estimated travel times (11 days to 60 cm and 32 days to 120 cm) were observed for the higher design loading rate of 8 cm/d.

Figure 3-23 correlates the actual (measured) HLR (cm/d) to the time for 50% bromide breakthrough at 60 cm below the infiltrative surface. While there is an overall trend of a decrease in time to 50% breakthrough with an increase in HLR, there is significant variability in the numbers. However, this graph shows that in the range of 2 to 10 cm/d HLR, the travel time to 60 cm in the vadose zone is greater than eight days, a period that can allow reactions to occur for the removal and transformation of pollutants. This figure also demonstrates the variability that occurred in dosing test cells during the pre-startup period. The problem of inconsistent delivery was corrected before delivery of effluent occurred.

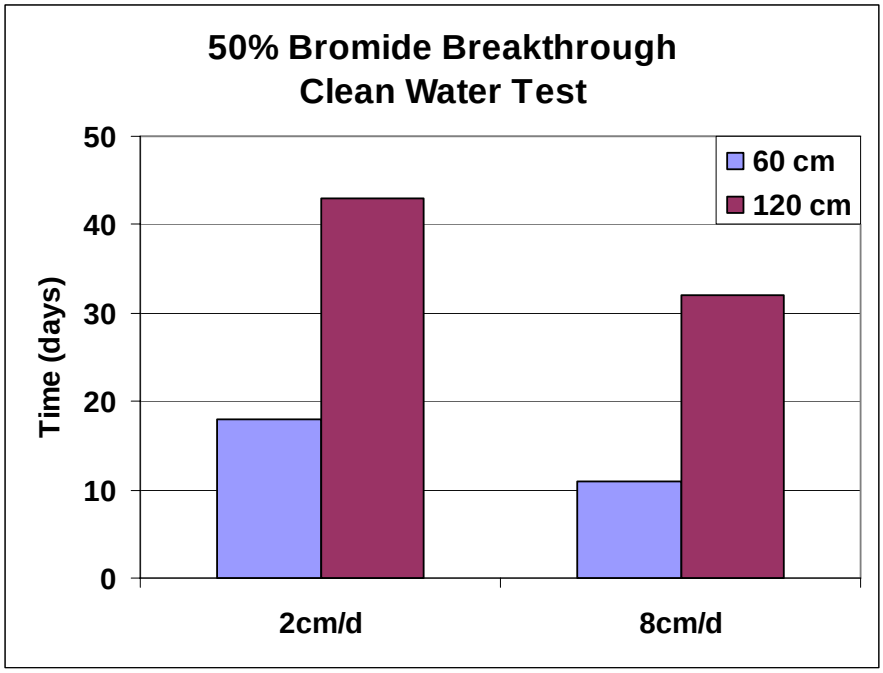


Figure 3-22
Time to 50% Bromide Breakthrough During Clean Water Tracer Test Based on Representative Test Cells Only (Received Near the Design HLR)

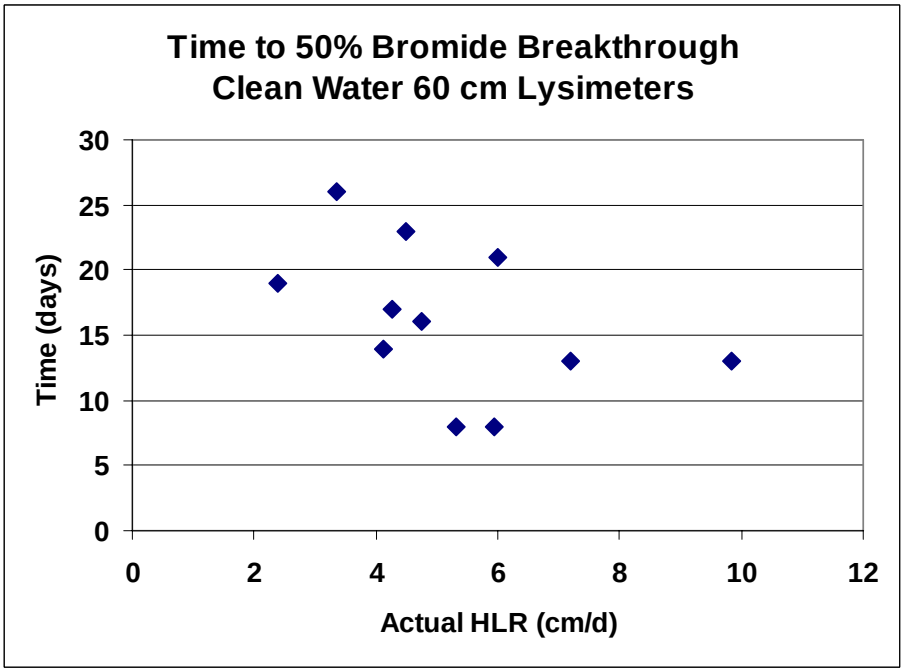


Figure 3-23
Time to 50% Bromide Breakthrough (in 60 cm Deep Lysimeters) as Function of Actual (Measured) HLR in Test Cells During the Clean Water Tracer Test

3.3.3.2 Three-Month Bromide Tracer Test

A second tracer test, conducted in July 2004 after three months of effluent dosing to the test cells, allowed comparison of travel times in the vadose zone to initial travel times observed in the clean water tracer test. Figure 3-24 presents a comparison of average values for time to 50% bromide breakthrough under each design HLR (2 or 8 cm/d) for each effluent type at pre-startup (clean water) to three months. The values presented represent the average of duplicate test cells. While the clean water delivery rates were highly variable, a general increase in travel time is observed and is consistent with biozone development.

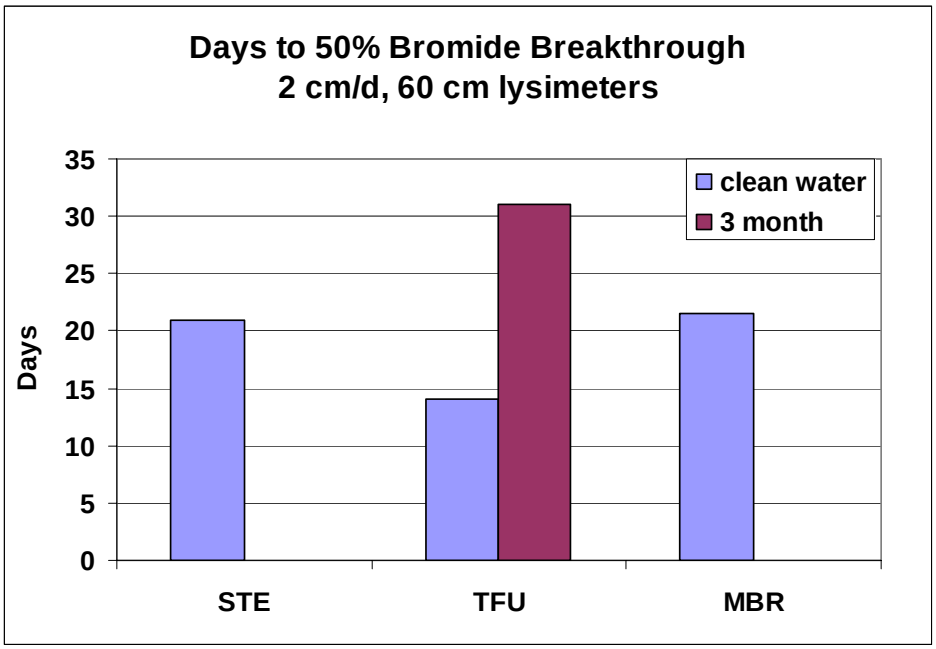
Higher bromide concentrations were observed at 120 cm compared to the shallower 60 cm lysimeter for the 2 cm/d design HLR. Alternatively, for the 8 cm/d design HLR test cells, while significant bromide concentrations were detected at 60 cm, only relatively low concentrations were detected in the 120 cm lysimeters. Representative bromide breakthrough curves for the TFU test cells are shown in Figure 3-25. This suggests that residual tracer from the clean water test remained within the soil below the test cells during initial sampling for the three-month tracer test (100 days between the end of tracer addition for the clean water test and the start of tracer addition for the three-month test).

Furthermore, 50% breakthrough concentrations were not observed in any 120 cm lysimeter samples (independent of effluent type or loading rate). Similarly, lysimeter samples taken from the 60 cm lysimeters dosed with STE and MBR effluent at 2 cm/d never reached bromide concentrations equivalent to 50% of the initial concentration. This precludes comparison of 50% breakthrough concentrations for each condition (such as STE at 2 cm/d) and indicates that the travel times in the subsurface were slower than expected and dispersion of the bromide may have been higher than expected.

3.3.3.3 RWT Transport Through the Vadose Zone

RWT was added to the test cells during the clean water tracer test. Transport through the vadose zone continues to be monitored. As previously discussed, RWT is a non-conservative tracer used in this study as a surrogate for organic chemicals. Figure 3-26 illustrates the comparison of the breakthrough curves of the conservative tracer bromide with the non-conservative tracer RWT within three test cells.

The concentration of RWT (in ppb) at 60 and 120 cm below the infiltrative surface was plotted over time for each of the test cells. An example of the breakthrough curves of three test cells is shown in Figure 3-27. The three cells received clean water spiked with an average RWT concentration of 2,335 μg/L. Water was delivered to each cell at an average delivery rate of 9.8 cm/d, 7.2 cm/d, or 2.4 cm/d, as noted in the Figure 3-27 legend. The peak concentrations seen in the 60 cm lysimeters ranged between 310 and 350 μg/L, 35 to 50 days after the start of tracer addition. By 170 days after the start of tracer addition, measurable concentrations of RWT were beginning to be seen in some test cells at 120 cm below the infiltrative surface. In only two test cells, those reporting the highest concentrations of RWT at the 60 cm depth, had RWT peaked in concentration at the 120 cm depth, which occurred 20 to 30 days after peaking at 60 cm (these test cells are not shown on Figure 3-27). The remaining cells had no or low concentrations of RWT at 120 cm depth at 170 days after the start of tracer addition.



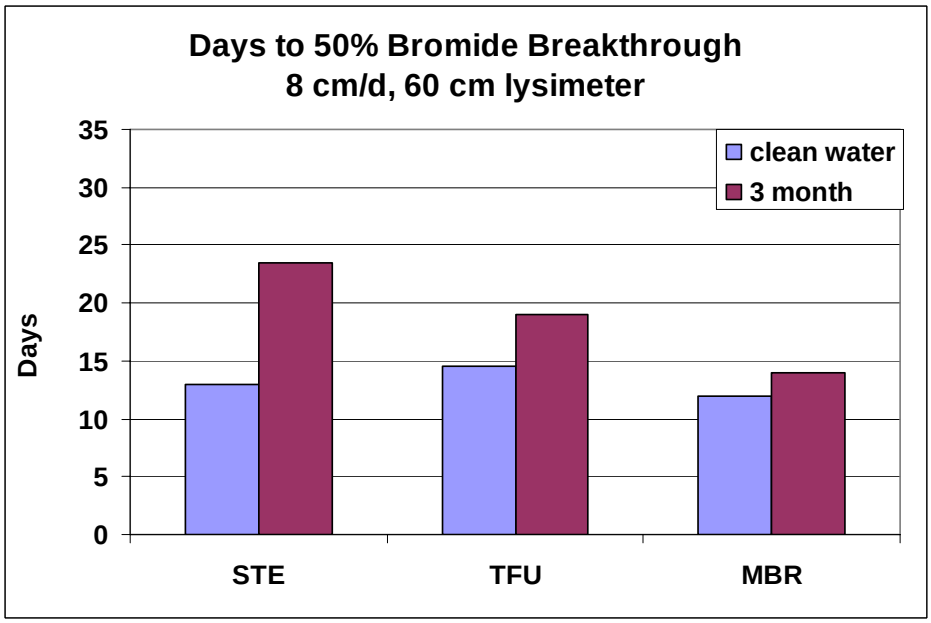
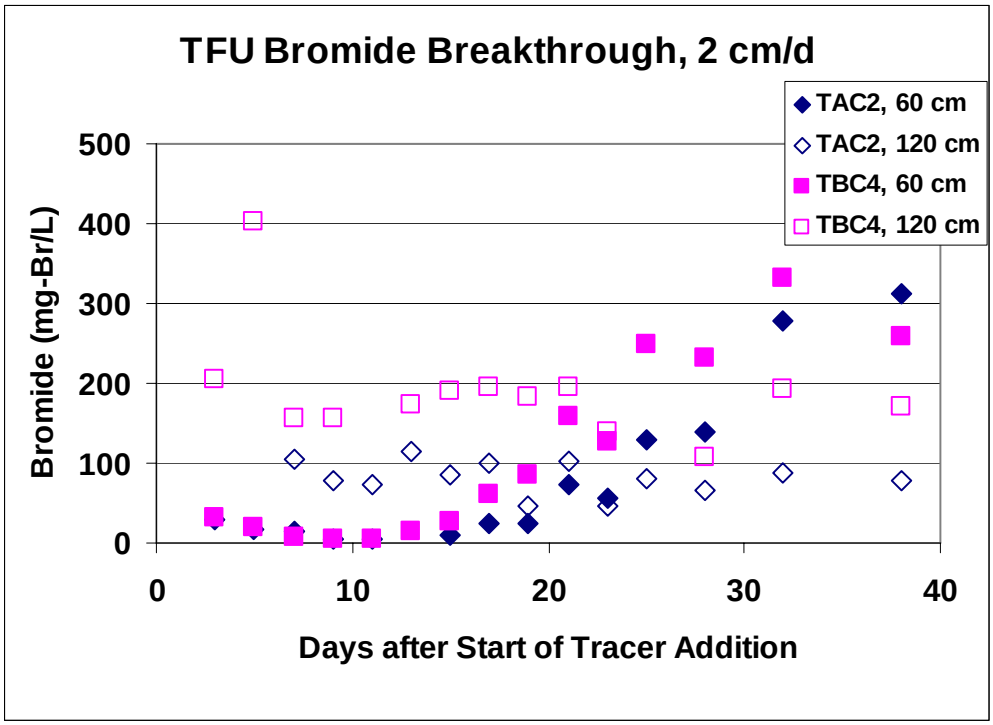


Figure 3-24
Comparison of the Time to 50% Breakthrough During Clean Water and Three-Month Tracer Tests for Samples Collected 60 cm Below the Infiltrative Surface at Design HLR of 2 cm/d or 8 cm/d[20]

[20] 50% breakthrough not observed during the three-month tracer test at 60 cm lysimeter for STE and MBR dosed at design HLR of 2 cm/d



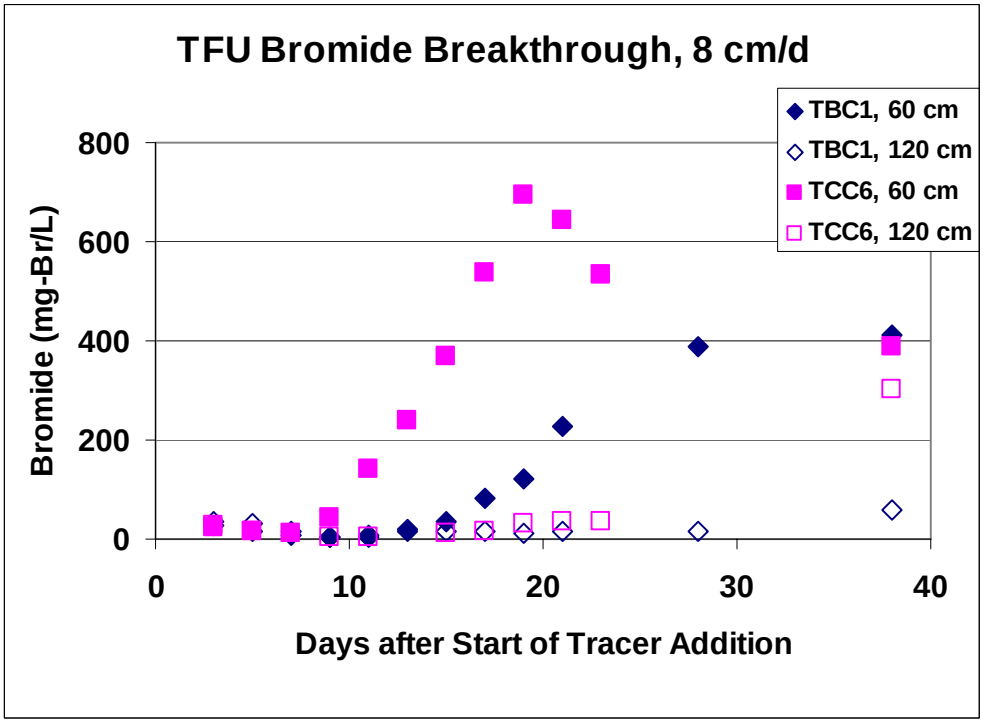


Figure 3-25
Representative Bromide Breakthrough Curves for TFU Test Cells During the Three-Month Tracer Test

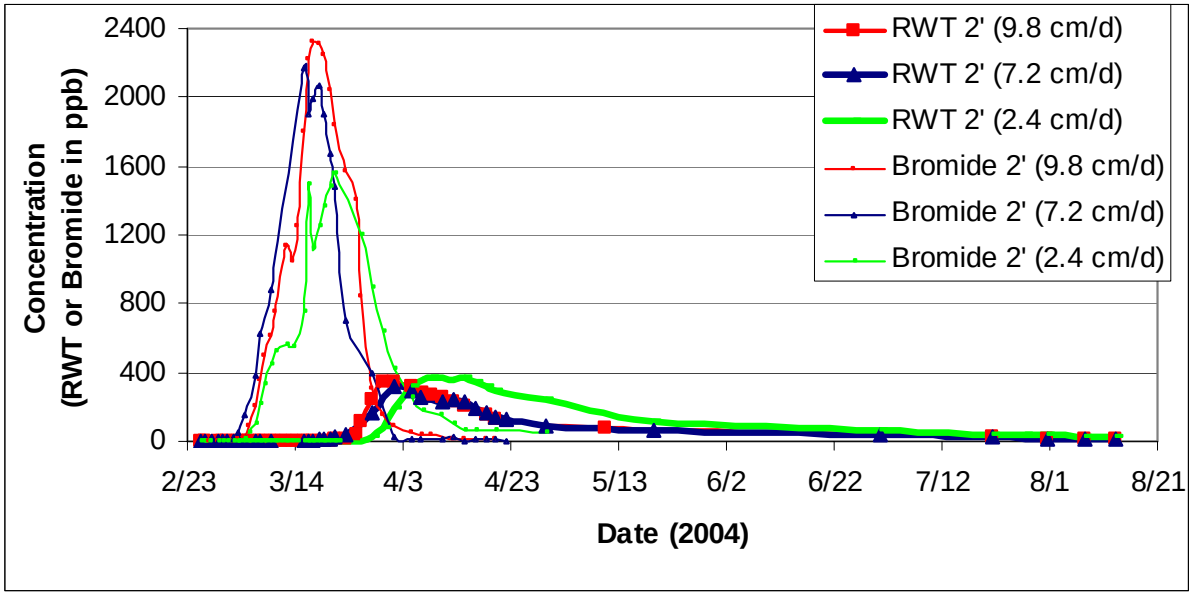


Figure 3-26
Comparison of Breakthrough Curves for Bromide and RWT in Three Test Cells[21]

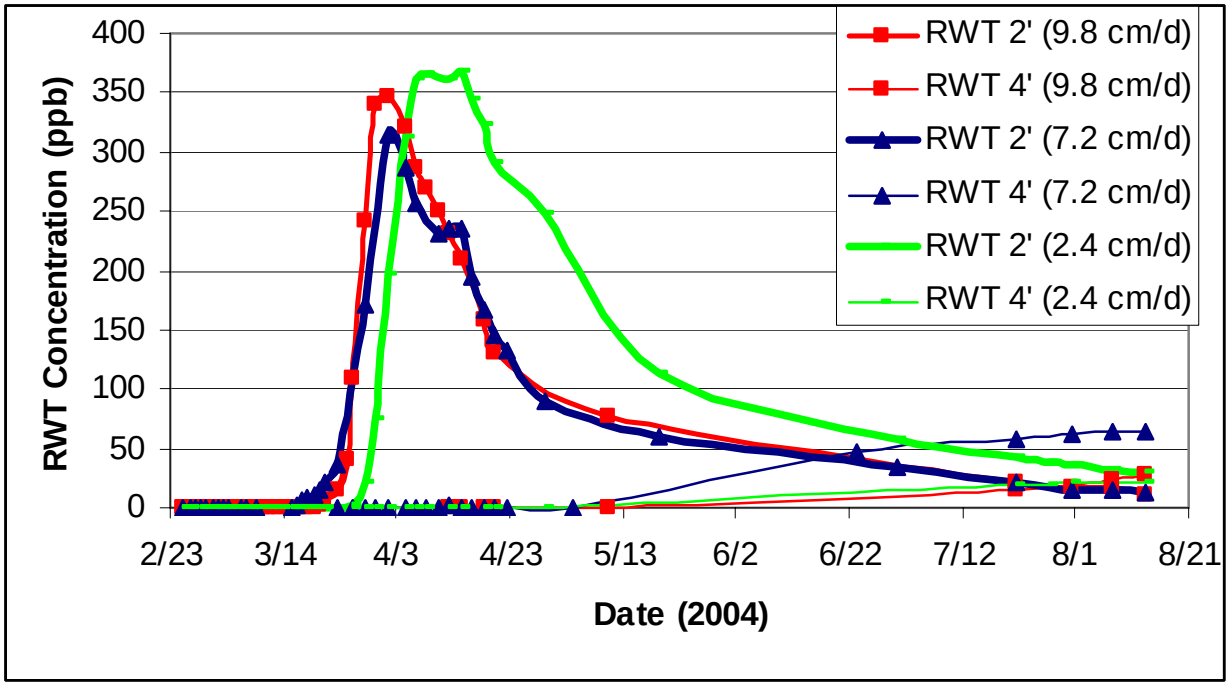


Figure 3-27
Breakthrough Curves of Three Test Cells[18]

[21] Actual delivery rate given in parentheses; average RWT concentration applied = 2,335 μg/L

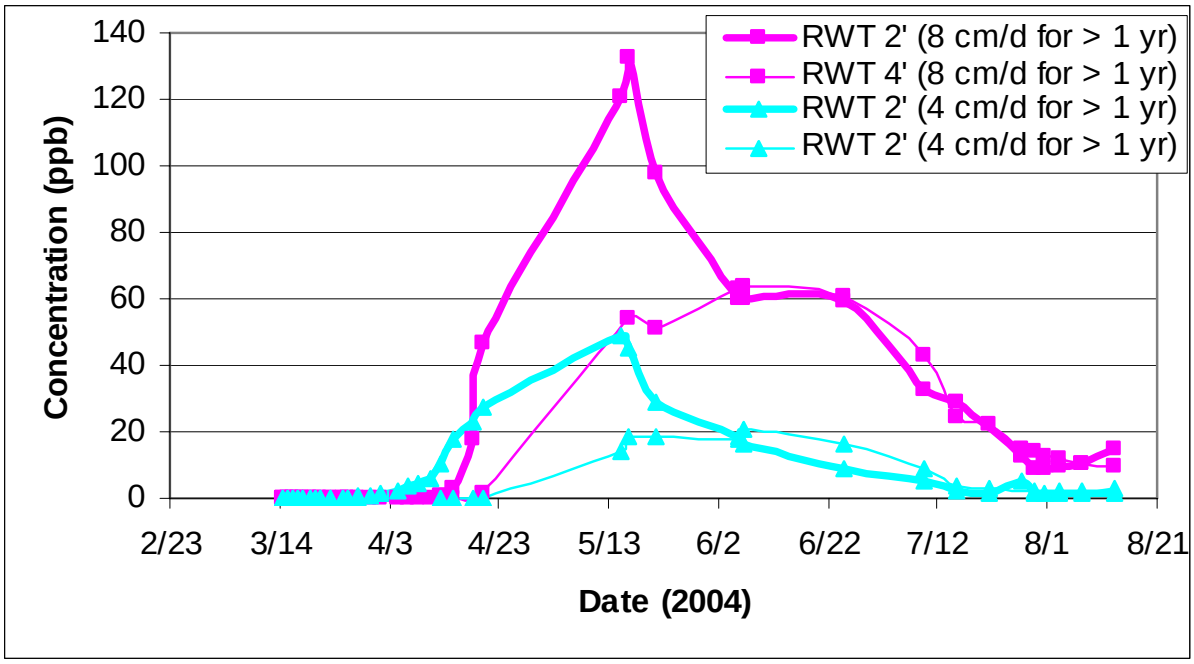


Figure 3-28
Breakthrough Curves of Two Test Cells That Have Received Clean Water Since May 2003[22]

Two additional test cells from the companion study received clean water (City of Golden tap water) spiked with tracer. These cells had been installed more than one year before the start of the tracer test (May 2003). They had been receiving clean water applied to the infiltrative surface at design HLR of 4 cm/d or 8 cm/d. The breakthrough curves of these two clean water test cells (Figure 3-28) show a peak concentration at 60 cm below the infiltrative surface of 133 µg/L and 48 µg/L for the cells receiving design HLR of 8 cm/d and 4 cm/d, respectively.

These peak concentrations occurred around 80 days after the start of tracer addition. At 120 cm below the infiltrative surface, peak concentrations seen were 64 and 21 µg/L for the 8 cm/d and 4 cm/d design HLR cells, respectively. These peak concentrations occurred around 102 days after the start of tracer addition (about 22 days after peaking at 60 cm). The peak concentrations in the test cells receiving clean water for more than one year were both lower in concentration (50% or less) and occurred later (30–45 days) than the peak concentrations seen in the test cells receiving clean water for less than one month, suggesting a difference in sorption capacity between the test cells based on duration of clean water application.

3.3.4 Soil Solution Characterization

Treatment efficiency of the soil was assessed through soil solution characterization using vadose zone lysimeters. Results from the lysimeter sampling are provided in this section.

[22] Actual delivery rate given in parentheses; average RWT concentration applied = 2,335 µg/L

3.3.4.1 Lysimeter Performance

All 30 lysimeters installed as part of this project have yielded soil solution samples. However, three of the six lysimeters at 240 cm below the infiltrative surface had insufficient sample volumes for analyses. After switching to an extended sampling schedule, two of these 240 cm lysimeters yielded sufficient sample volume for analyses. In addition, one 120 cm lysimeter has not consistently yielded sufficient sample volume for analyses, frequently yielding no appreciable volume.

At the same locations and depths of some of the lysimeters, subsurface temperature probes were installed at 60, 120, and 240 cm below the infiltrative surface (6, 6, and 1 probe, respectively). The results of periodic monitoring of subsurface temperatures revealed temperature ranges of 8.9 to 18.3 ^{o}C (48 to 65 ^{o}F) at the 60 cm depth, 8.9 to 16.7 ^{o}C (48 to 62 ^{o}F) at the 120 cm depth, and 8.9 to 14.4 ^{o}C (48 to 58 ^{o}F) at the 240 cm depth. There was a general seasonal trend with subsurface temperatures observed in April 2004 (minimum value of range) increasing over the summer (maximum value of range observed at 60 and 120 cm in August 2004, but later at 240 cm depth). The relatively uniform subsurface temperatures are attributed to the time of monitoring (April to October) when temperatures are moderate to warm.

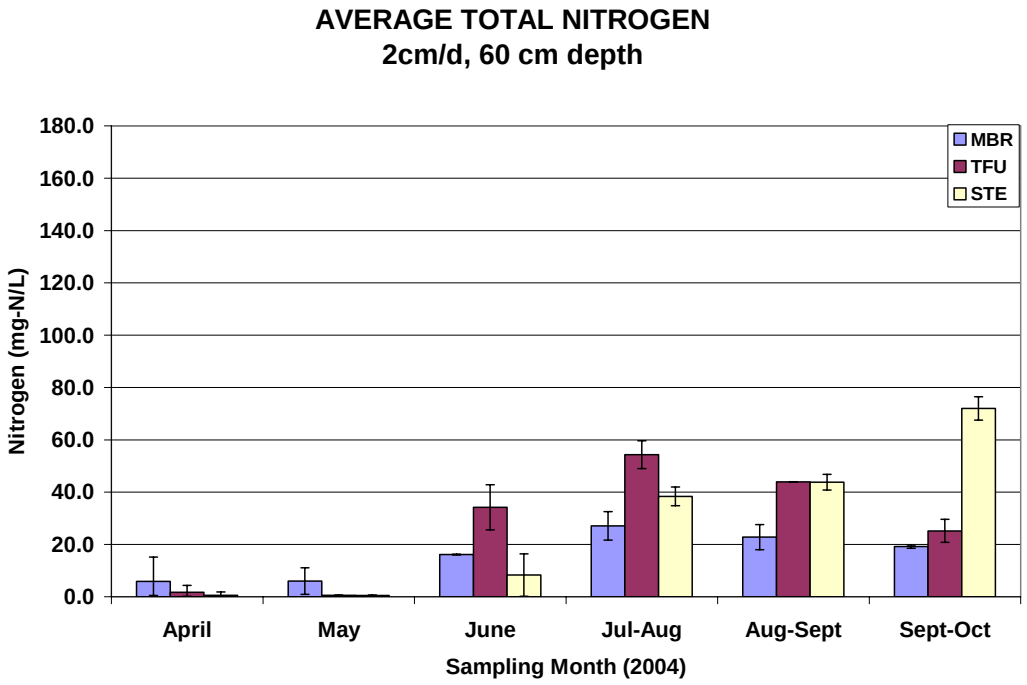
3.3.4.2 Water Quality Characteristics

This section presents the water quality results from seven months of soil solution sampling (six sample rounds). As discussed previously (Section 2.5.1.2.3), during the three-month tracer test conducted in July 2004, interference with several analysis methods was observed due to bromide. Specifically, elevated bromide concentrations interfered with DOC, COD, total nitrogen, and nitrate analyses. Interpretation and data impacts are presented in the following discussions where applicable. See Section 3.4.2 for discussion of purification efficiency (percent removals).

3.3.4.2.1 Nitrogen Compounds

The average total nitrogen applied to the test cells was 62.4 mg-N/L for STE, 57.3 mg-N/L for TFU during the first 51 days of operation, 39.5 mg-N/L for TFU for more than 51 days of operation, and 26.5 mg-N/L for MBR. Little-to-no total nitrogen was found in the soil solution during the first month of operation, except for the test cells receiving MBR effluent. Figure 3-29 and Figure 3-30 depict the average of duplicate test cells for each sample round in the 60 and 120 cm lysimeters. Cells receiving MBR effluent, however, received an already nitrified effluent from the first day of effluent delivery to the test cells. Since nitrate is mobile in soil, it is not surprising that total nitrogen (nitrate) was detected in the test cells receiving MBR effluent. This is demonstrated to the greatest extent in the 60 cm lysimeters receiving MBR at the higher design loading rate (8 cm/d).

After approximately two months of operation, increasing total nitrogen concentrations were detected in the STE and TFU test cell lysimeters. A steady increase of total nitrogen was observed with time in the STE test cells at both 60 cm and 120 cm below the infiltrative surface.



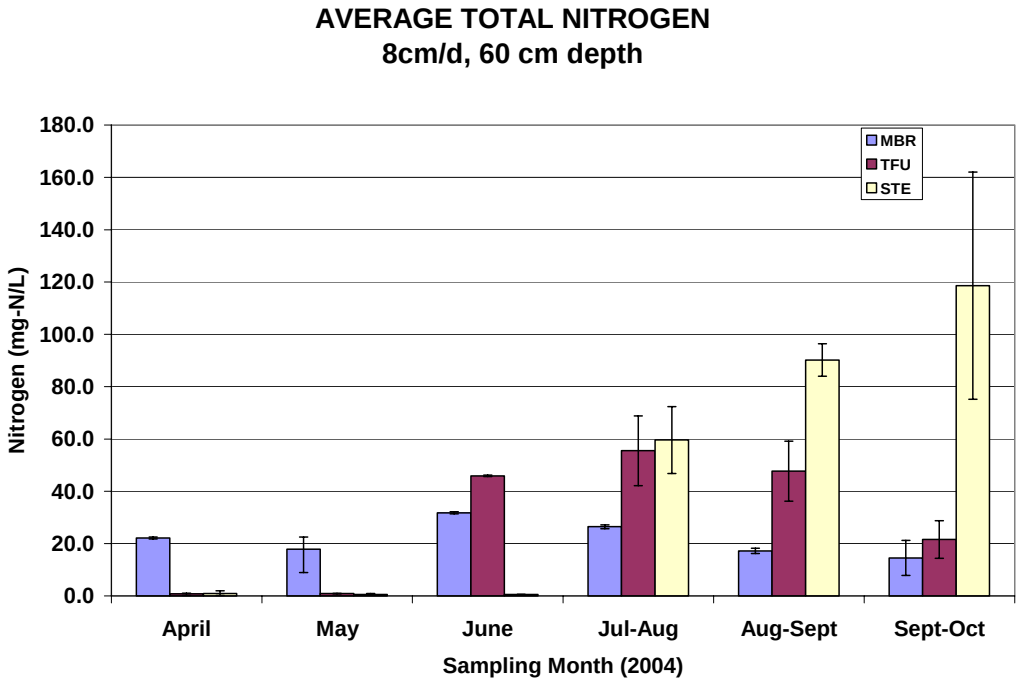


Figure 3-29
Average Total Nitrogen Measured in 60 cm Lysimeter Samples Taken From Test Cells Dosed at Design HLR of 2 or 8 cm/d [23]

[23] Error bars indicate minimum and maximum sample values; no error bar indicates only one sample shown

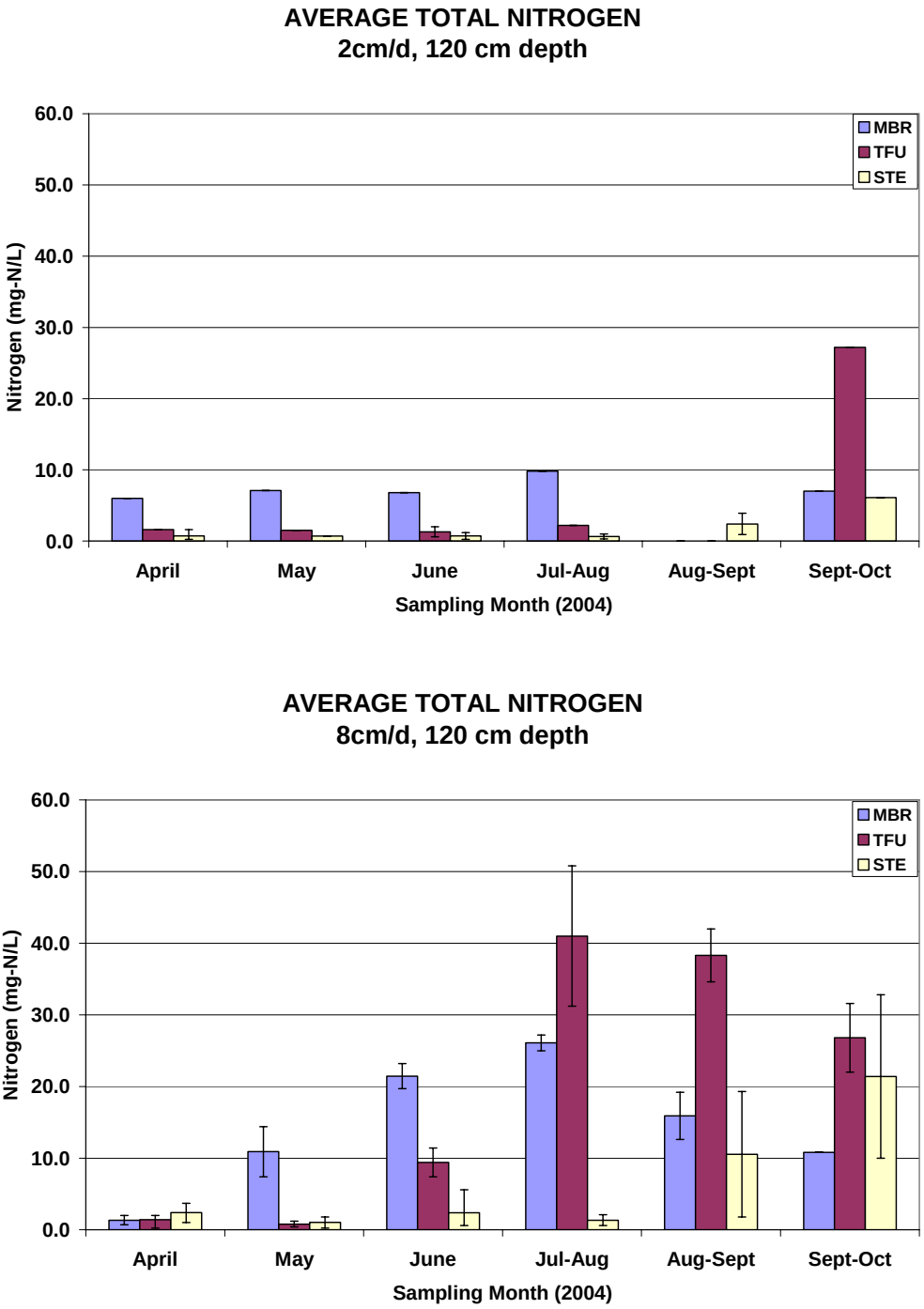


Figure 3-30
Average Total Nitrogen Measured in 120 cm Lysimeter Samples Taken From Test Cells Dosed at Design HLR of 2 or 8 cm/d [24]

[24] Error bars indicate minimum and maximum sample values; no error bar indicates only one sample shown

The increased levels of total nitrogen observed in the STE test cells may be attributed to cycling of nitrogen retained in the biozone (Grady *et al.* 1999). As the microbes in the biozone die and decay, the cell material (with nitrogen) is released and becomes mobile in the soil pore water. Ultimately, some of the nitrogen is converted to ammonium. Bioassimilation of nitrogen is typically considered a nitrogen removal process. The bacterial cells are harvested and removed from the wastewater treatment process (Converse 1999). However, these cells are not removed in the soil treatment unit of an OWS and may have contributed to the increased nitrogen levels observed in the STE test cells at 60 cm below the infiltrative surface.

Alternatively, the positively charged NH_4^+ can be strongly adsorbed onto mineral surfaces causing retardation in the soil. It is possible that release of the NH_4^+ occurred through cation exchange; however, the concentration of NH_4^+ in effluent typically overcomes the higher strength cation bonds (Al, Ca, Mg) (Tackett 2004).

With time, the total nitrogen concentrations at 60 and 120 cm below the infiltrative surface are expected to come to a steady state. Continued monitoring is required to evaluate this trend. A general trend in the TFU test cells of increasing total nitrogen concentrations during the first four months of operation, followed by a decrease, is attributed to nitrogen removal of the TFU.

Overall, independent of effluent quality, the test cells with the higher loading rates resulted in higher concentrations of total nitrogen in the soil solution (at both 60 and 120 cm). As previously mentioned, bromide interfered with the total nitrogen analysis method (see Table 2-8). No attempt was made to "correct" the total nitrogen data based on bromide interference during analysis due to the limited number of affected sampling points (approximately 30 of 140 samples) and the remaining uncertainty of the data if it had been corrected. Rather, the total nitrogen results in July and August are expected to be 10 to 20% low.

Figure 3-31 and Figure 3-32 depict the average nitrate concentrations (of the duplicate test cells for each sample round) in the 60 and 120 cm lysimeters. The average nitrate applied to the test cells was 2.5 mg-N/L for STE, 1.4 mg-N/L for TFU during the first 51 days of operation, 16.0 mg-N/L for TFU for more than 51 days of operation, and 21.3 mg-N/L for MBR. Nitrate was observed the earliest (first month of operation) in the test cells receiving the nitrified MBR effluent. However, as noted previously, the MBR began receiving STE in January/February 2004 while the TFU began receiving STE in April 2004, explaining why nitrate was observed within the first month of operation for the MBR test cells, but not for several months for the TFU test cells.

In general, the observed nitrate trends are similar to the total nitrogen trends. The most significant impact due to bromide interference was seen in nitrate analyses, where less than 25% of the nitrate was recovered at bromide concentrations in the sample as low as 100 mg-Br/L. In these cases, nitrate data were removed from the dataset. This screened dataset is sporadic and limited (37 of 144 samples removed from the dataset) making the interpretation of the nitrate trends and estimation of nitrogen balances difficult. Continued monitoring is required. Future sample analysis for nitrate will be conducted using methods that are not sensitive to bromide concentration.

Over the six months of operation, ammonium was not detected at appreciable amounts in any of the lysimeter samples. The majority of the results were below the detection limit (0.02 mg-N/L ammonium). A few results were just at the detection limit.

The average ammonium applied to the test cells was 58.0 mg-N/L for STE, 51.8 mg-N/L for TFU during the first 51 days of operation, 11.2 mg-N/L for TFU for more than 51 days of operation, and 0.7 mg-N/L for MBR. There was no interference with ammonium analyses due to bromide. However, the impact to the nitrate and total nitrogen analysis prevents assessment of nitrogen balances for several individual conditions.

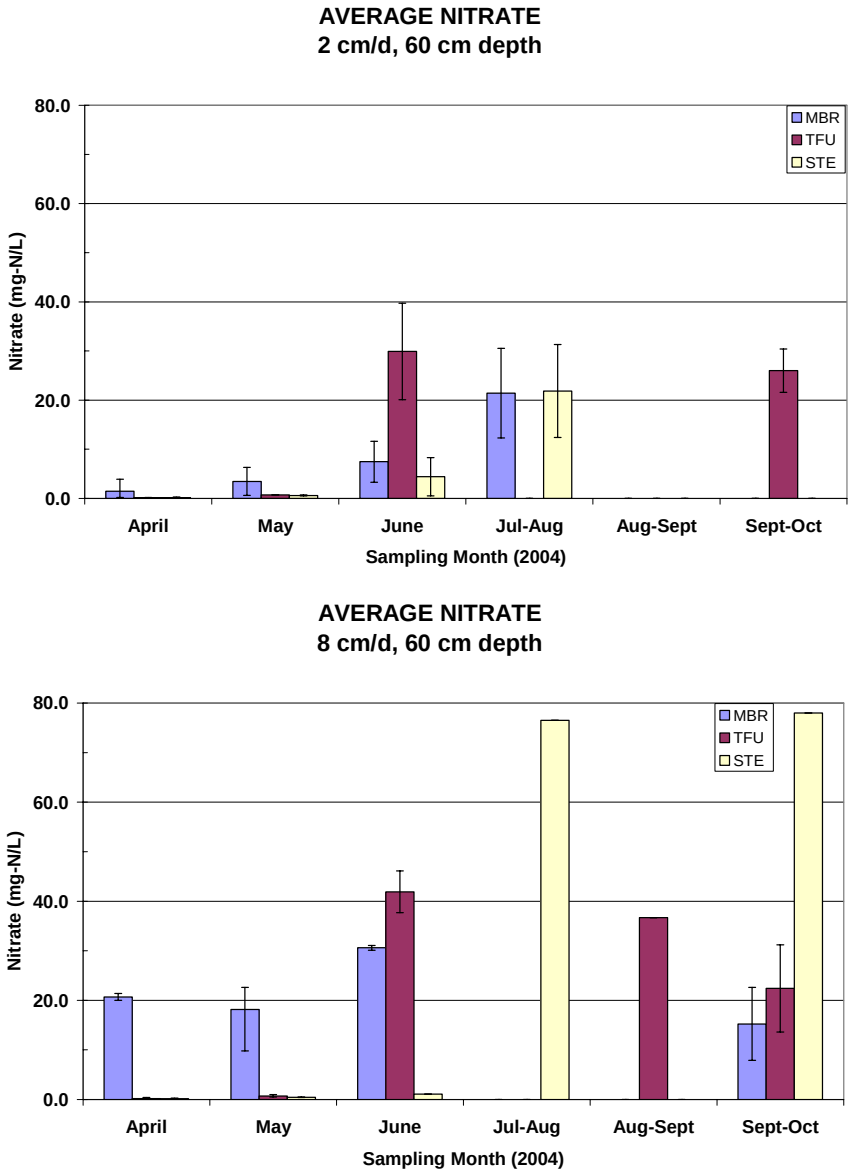


Figure 3-31
Average Nitrate Measured in 60 cm Lysimeter Samples Taken From Test Cells Dosed at Design HLR of 2 or 8 cm/d[25]

[25] Error bars indicate minimum and maximum sample values; no error bar indicates only one sample shown

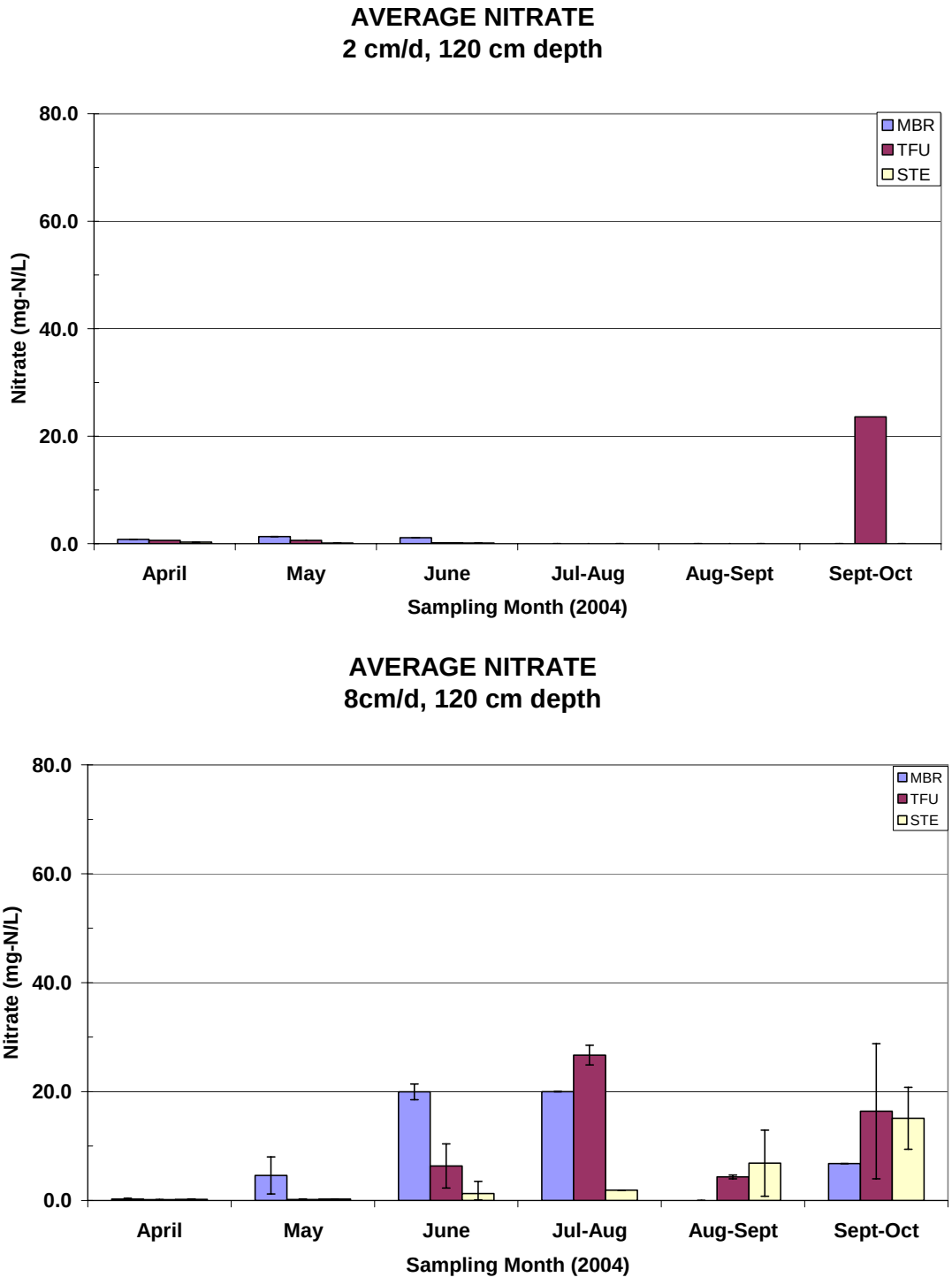


Figure 3-32
Average Nitrate Measured in 120 cm Lysimeter Samples Taken From Test Cells Dosed at Design HLR of 2 or 8 cm/d [26]

[26] Error bars indicate minimum and maximum sample values; no error bar indicates only one sample shown

3.3.4.2.2 Organic Carbon

Organic carbon in the soil solution was analyzed to determine its fate in the vadose zone and to assist in the understanding of nitrogen cycle trends in the soil. Since the nominal pore size of the lysimeters is 0.2 µm (<0.45 µm filter is normally used for measuring DOC), it is assumed that the measurement of TOC represents only the dissolved carbon portion, or DOC. The average DOC applied to the test cells was 34.2 mg-C/L for STE, 9.7 mg-C/L for TFU during the first 51 days of operation, 11.5 mg-C/L for TFU for more than 51 days of operation, and 6.2 mg-C/L for MBR. The DOC concentrations in the 120 cm lysimeters for the test cells at the 8 cm/d design loading rate were higher than those in the cells at the 2 cm/d design loading rate (see Figure 3-33 and Figure 3-34).

As previously mentioned, bromide interfered with the DOC analysis method (see Table 2-8). While there was no impact to the effluent DOC data, the impacts to the lysimeter data were significant. Due to higher DOC concentrations and the use of an instrument with a higher detection range for the effluent characterization, no data points were removed. However, there was no predictable correlation established between bromide concentration in the sample and percent DOC recovery for the instrument with a lower DOC detection range (0.5 to 10 mg-C/L) that was used for lysimeter samples. Samples with more than 100 mg-Br/L (36 of 100 total samples) were removed from the dataset.

3.3.4.2.3 Phosphorus

Over six months of effluent application, analyses of soil solution demonstrated nearly complete removal of phosphorus after effluent treatment in only 60 cm of the vadose zone (that is, at 60 cm lysimeter). Nearly all of the analytical results were below or just slightly above the 0.06 mg-PO_4/L detection limit. All soil solution results were below 0.5 mg-PO_4/L, as compared with total phosphorus average concentrations in the effluents, which were 20.9 mg-PO_4/L (STE), 18.9 mg-PO_4/L (TFU) and 19.5 mg-PO_4/L (MBR). There was no interference with phosphorus analyses due to bromide.

3.3.4.2.4 pH and Alkalinity

The average alkalinity applied to the test cells was 272 mg-$CaCO_3$/L for STE, 228 mg-$CaCO_3$/L for TFU during the first 51 days of operation, 53.6 mg-$CaCO_3$/L for TFU for more than 51 days of operation, and 30.4 mg-$CaCO_3$/L for MBR. Results of the soil solution analyses for pH and alkalinity from the STE-dosed test cells show pH values consistently between 7.3 and 8.4. Alkalinity values had some variability ranging from 50 to 230 mg-$CaCO_3$/L. There was no apparent trend for pH or alkalinity with lysimeter depth, operation time, or HLR.

TFU test cell lysimeter samples were similar to STE lysimeter samples with a pH between 7.1 and 8.2 and an alkalinity between 50 and 180 mg-$CaCO_3$/L. MBR test cell lysimeter samples had pHs between 6.8 and 8.2 and alkalinities between 40 and 120 mg-$CaCO_3$/L. As with STE, there was no apparent trend for pH or alkalinity for TFU or MBR lysimeters with depth, operation time, or HLR. Higher alkalinity values in both TFU and MBR lysimeter samples compared to the effluent dosed to these cells is attributed to the soil adding alkalinity to the soil pore water.

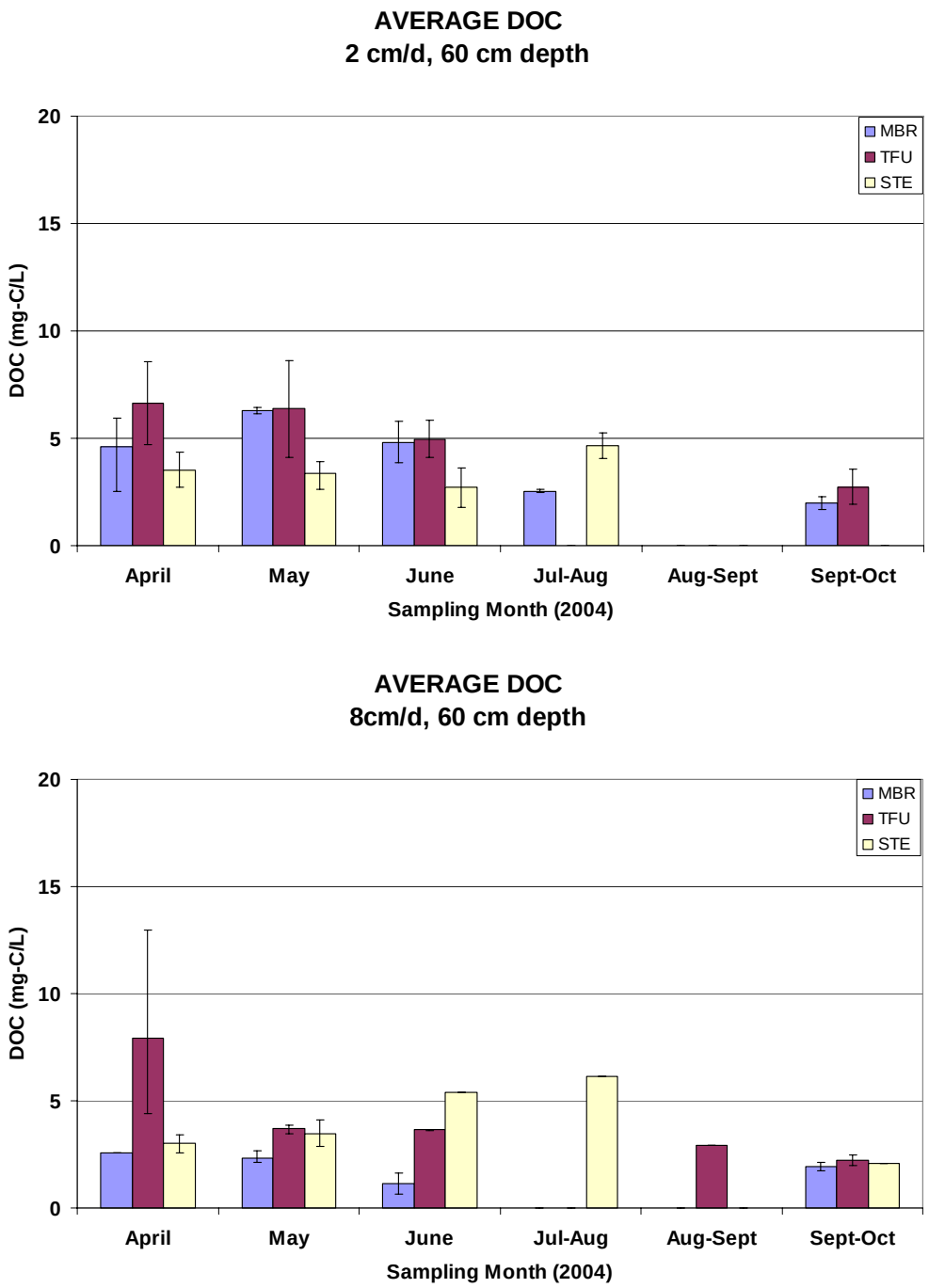


Figure 3-33
Average DOC Concentrations in 60 cm Lysimeter Samples Taken From Test Cells Dosed at Design HLR of 2 or 8 cm/d [27]

[27] Error bars indicate minimum and maximum sample values; no error bar indicates only one sample shown

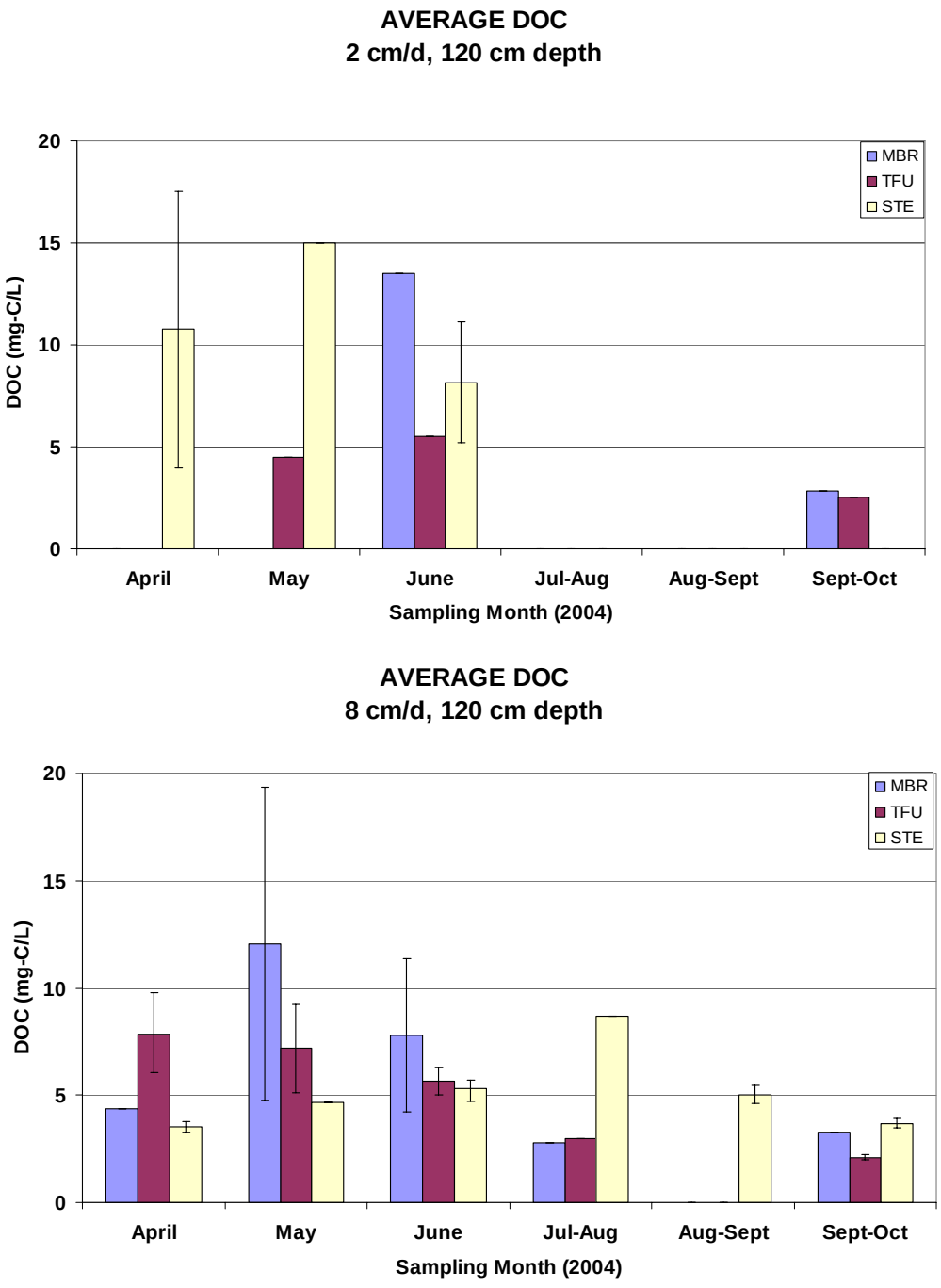


Figure 3-34
Average DOC Concentrations in 120 cm Lysimeter Samples Taken From Test Cells Dosed at Design HLR of 2 or 8 cm/d [28]

[28] Error bars indicate minimum and maximum sample values; no error bar indicates only one sample shown

3.3.5 Soil Profile Characterization

The soils below the infiltrative surface within the companion study site were characterized after 13 months of STE application by coring through several individual test cells (Appendix C). Of the 12 locations sampled, two locations received STE at a design-loading rate of 8 cm/d and are relevant to this study. Duplicate cores were collected 15 cm apart from each of the 12 locations (designated as "north" or "south" based on the relative position of the duplicate cores). An operational history of these two test cells is presented in Table 3-8. Additional information on the companion study is presented in Appendix B. The remaining 10 locations are specific to the companion study to evaluate the effects of infiltrative surface architecture and HLR on soil hydraulic performance and purification. The results presented here reflect 13 months of STE delivery while the NDWRCDP had received effluent for six months at the time of this writing. A similar characterization is planned for the soil profile of the current test cells, which are still in operation as of the writing, and the results will be presented elsewhere.

Table 3-8
Summary of Operational History for Test Cells Cored During Soil Vadose Zone Characterization (July 2004)

Test Cell Attribute	Test Cell Identification	
	T1C10	**T2C8**
Effluent quality	STE	STE
Infiltrative surface architecture	Open	Open
Design HLR	8 cm/d	8 cm/d
Actual HLR (median)	6.9 cm/d	6.3 cm/d
Effluent start date	May 5, 2003	May 5, 2003
Date test cell cored	July 27, 2004	July 27, 2004
Days of effluent loading at time of coring	449 days	449 days
Baseline IR	50.2 cm/d	41.8 cm/d
12 mo. IR	2.6 cm/d	4.0 cm/d
IR_t/IR_o (t = 12 mo.)	0.05	0.09
Days to continuous effluent ponding	<30 days	36 days
Depth of effluent ponding at time of coring	17.1 cm	17.3 cm
Cumulative volume processed at time of coring	14775 L (3903 gallons)	14925 L (3943 gallons)

3.3.5.1 Physical Properties

The results of soil core analyses revealed the soil pH to be in the range of 4.6 to 7.0 with a median value of 6.1. The color was described as reddish brown to dark reddish brown (Munsell color 3/2 5YR to 4/4 5YR) with the biozone a notable dark brown/black color (top 1 to 2 cm of soil from the infiltrative surface). As expected, the water content was highest within the biozone (0 to 2 cm). In the underlying soil, higher water contents near the biozone (2 to 17 cm) gradually declined until they remained consistent to depths up to 1.5 m below the infiltrative surface. The average volumetric water content (duplicate cores collected from two locations, T1C10 and T2C8) with depth is presented in Figure 3-35.

The water content (by dry weight) near the infiltrative surface (4 to 9 cm) was approximately 23% and averaged approximately 13% at depths greater than 10 cm. These water contents equate to a volumetric water content (volume of water / volume of soil) of greater than 50% at depths less than 10 cm and less than 30% at depths greater than 10 cm (Figure 3-35). The average water content for background sample cores was about 5% by dry weight, or 12% volumetric water content.

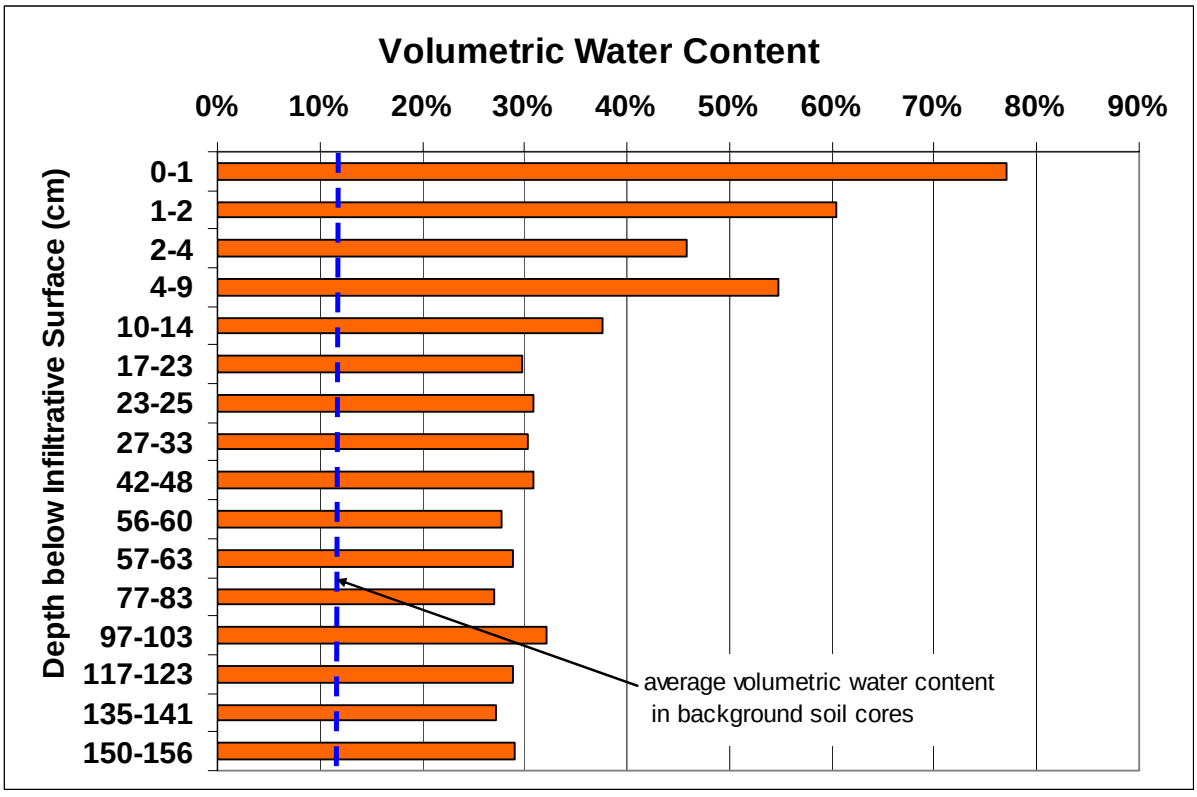


Figure 3-35
Average Volumetric Water Content in Soil Cores Taken From the Companion Study Test Cells That Were Dosed With STE for 13 Months at Design HLR of 8 cm/d (Assuming a Moist Bulk Density of 2.34 as Measured at the Mines Park Test Site)

3.3.5.2 Chemical Properties

The soil profile with depth was evaluated for exchangeable ammonium and total and available phosphorus concentrations. The top 4 cm (0 to 4 cm below the infiltrative surface) were not analyzed for exchangeable ammonium, available phosphorus, or total phosphorus, since there was inadequate soil sample left after analyses for other parameters. This upper layer of soil, where a biozone forms (observed from 0 to 2 cm), may be responsible for retaining a disproportionately large amount of these nutrients. However, nutrient transformation and retention extends below this upper layer. Also, due to subsurface complexities, the core samples taken may not be completely representative of the variability throughout the entire vadose zone.

Background ammonium concentrations in this soil averaged 5.2 mg-N/kg dry soil. Because the positively-charged ammonium ion can be strongly adsorbed onto mineral surfaces, its movement in the soil is retarded. After 13 months of effluent application, increased ammonium soil concentrations were observed to approximately 50 cm below the infiltrative surface (Figure 3-36). Maximum ammonium concentrations of more than 500 mg-N/kg dry soil were observed in the interval immediately below the biozone (4 to 9 cm below the infiltrative surface).

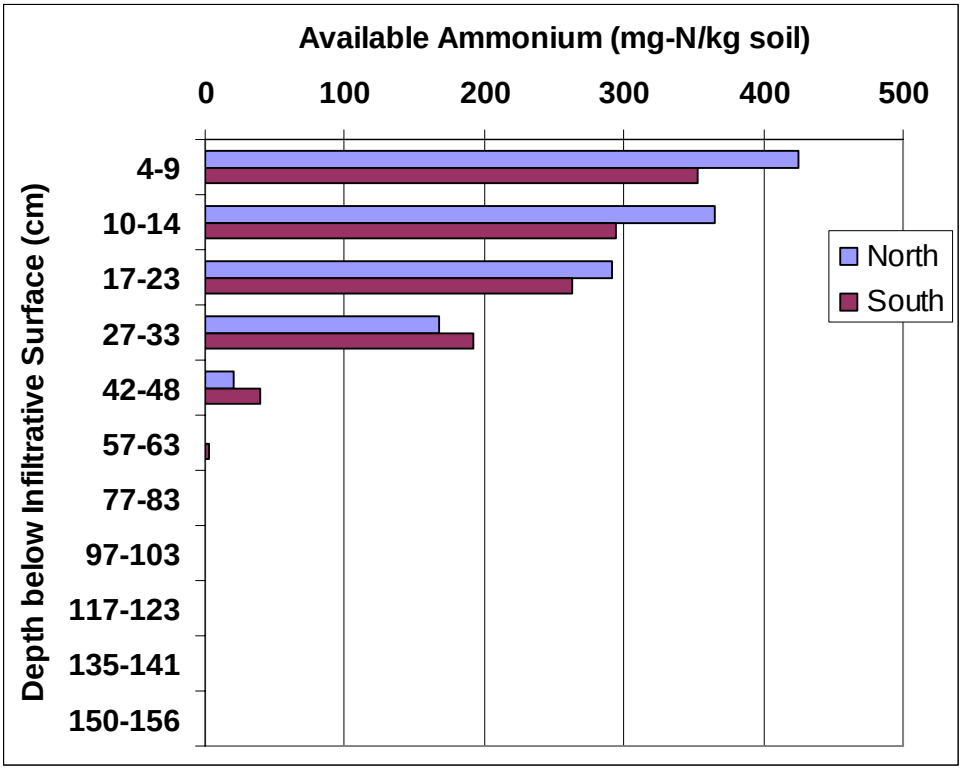


Figure 3-36
Average Ammonium Concentration in Soil Cores Taken From the Companion Study Test Cells That Were Dosed With STE for 13 Months at Design HLR of 8 cm/d[29]

[29] Duplicate cores (north and south) were collected from two test cells. No bar indicates concentrations less than 2.6 mg-N/kg dry soil

Based on the ammonium soil concentrations and the loading rate, nearly 15% (or approximately 126 g-N) of the applied effluent ammonium at a HLR of 8 cm/d was retained in the top 50 cm of soil below the infiltrative surface. Below 50 cm, ammonium soil concentrations ranged from 0.15 to 2.6 mg-N/kg dry soil (note, due to scale on Figure 3-36, the measured concentrations between 50 and 156 cm below the infiltrative surface are not shown). The absence of ammonium in lysimeter samples supports that the applied effluent ammonium is not mobile and may be present but firmly sorbed in the soil.

Similar trends were observed for available phosphorus, with increased available phosphorus concentrations observed to approximately 35 cm below the infiltrative surface (Figure 3-37). Background available phosphorus soil concentrations averaged 4.4 mg-P/kg soil. A maximum concentration of nearly 30 mg-P/kg soil was observed at 4 to 9 cm depth. As expected, the total phosphorus soil concentrations show a similar trend as the available phosphorus. Based on available phosphorus soil concentrations, only about 3% (or 10 g-P) of the applied effluent phosphorus was retained in the top 50 cm of soil below the infiltrative surface. However, the absence of phosphorus in the lysimeter samples suggests that the applied effluent phosphorus is not mobile and may be present in the soil in firmly sorbed or precipitate forms.

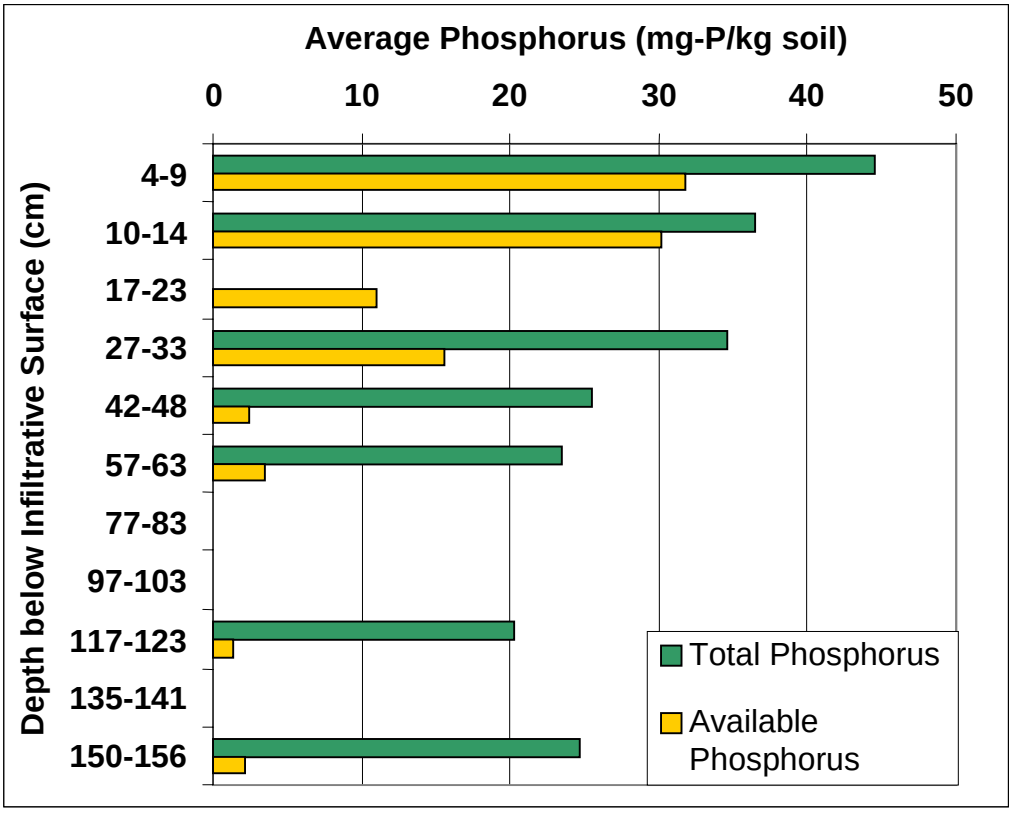


Figure 3-37
Distribution of Available and Total Phosphorus in Soil Cores Taken From the Companion Study Test Cells That Were Dosed With STE for 13 Months at Design HLR of 8 cm/d

3.3.5.3 Biological Properties

The companion study test cell cores were extracted and analyses of the extract were conducted for heterotrophic plate count (HPC) bacteria, fecal coliform bacteria, *E. coli*, and MS-2 and PRD-1 bacteriophages (see Appendix C for soil core sample analysis materials and methods). Collection of soil cores and extraction of these cores is the only feasible way to obtain information on the fate of bacteria in these systems, due to the constraints of using porous cup suction lysimeters and the limitation posed by their small pore sizes.

Figure 3-38 presents results from one test cell (T1C10) loaded at a design HLR of 8 cm/d with STE (actual HLR=6.9 cm/d). Fecal coliform bacteria and *E. coli* were detected at low levels in the top 10 cm of soil below the infiltrative surface. The sporadic detection of fecal coliform bacteria is likely due to variability of bacteria in the soil compounded by the small soil sample size (for example a few grams of soil from one location within the test cell). Where no bar is present, no bacteria were detected. Selected cores were analyzed for bacteria at depths greater than 10 cm. However, results were consistent indicating no detectable fecal coliform bacteria or *E. coli* and high levels of heterotrophic bacteria (all HPCs were too numerous to count; greater than 1.0×10^{23} cfu per gram of dry soil). Samples for T1C10 were not analyzed greater than 10 cm below the infiltrative surface. No bacteria have been detected in lysimeter samples (nominal pore size of 0.2 microns).

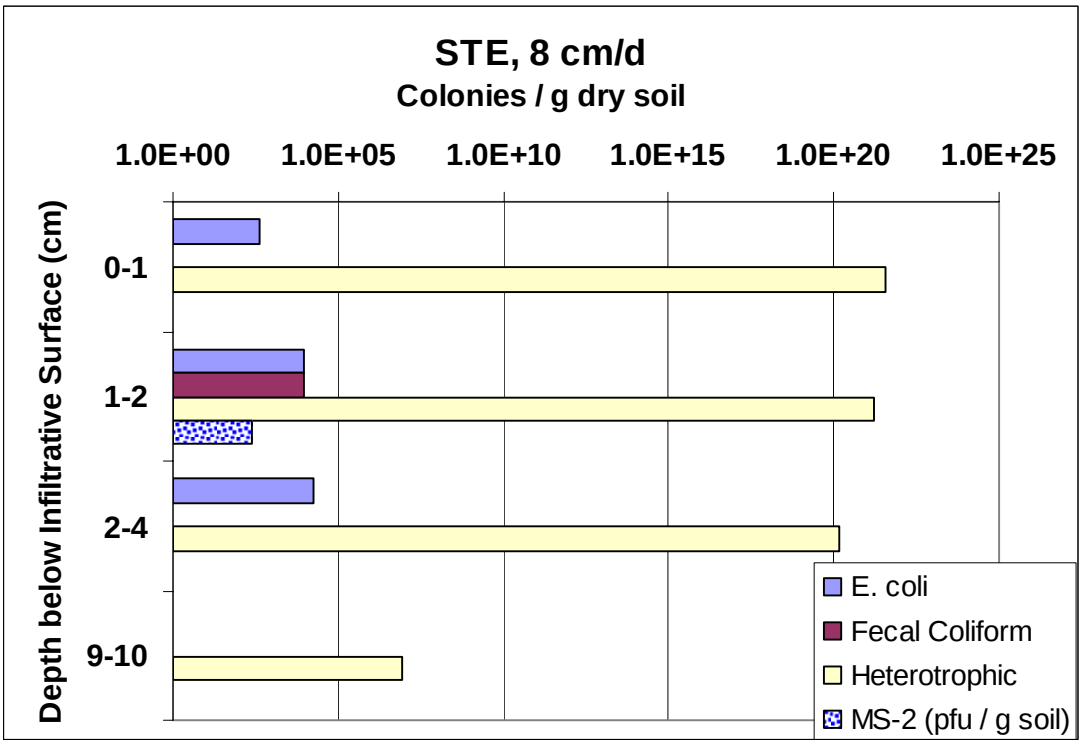


Figure 3-38
Extracted Values (cfu/g Dry Soil) of HPC Bacteria, Fecal Coliform Bacteria, *E. coli*, and MS-2 Bacteriophage in Soil Core Taken From a Companion Study Test Cell Loaded With STE at Design HLR of 8 cm/d for 13 Months[30]

[30] The absence of a bar represents a non-detect

Two background cores were taken in an area adjacent to the test cells that had not received any effluent. Cores from the background site were analyzed for HPC, fecal coliform bacteria, *E. coli*, and MS-2 and PRD-1. No detectable levels of fecal coliform bacteria, *E. coli*, or either of the bacteriophages were measured in the background soil sample extracts. HPC in soil samples from depths of 30, 60, and 120 cm in duplicate cores from the same area were too numerous to count (more than 1.0×10^{23} cfu per gram of dry soil). HPC analyses were not conducted within 30 cm of the infiltrative surface due to a problem with the assay and reagents used for analysis. Therefore, background levels of HPC from 0 to 30 cm below the infiltrative surface are unknown, but likely to be very high.

In an attempt to better understand the location and mechanisms of organic carbon and inorganic nutrient removal during effluent treatment in soil, soil biomass activity measurements were conducted. Phospholipid extraction proved to be a viable tool to quantify total viable biomass in soil systems impacted by effluent infiltration (Appendix C). This assay quantifies all viable microbial cells that are potentially, but not necessarily, active in soil. Background biomass concentrations in the soil from the Mines Park Test Site before application of wastewater effluents were monitored on two occasions: October 2002 (in the area of the companion study) and October 2003 (in the area of this study) (Figure 3-39 and Figure 3-40). Both sampling events resulted in similar, low biomass concentrations, of approximately 1 to 15 nmol phosphate per g dry soil in the first 2 m of soil. Slightly higher microbial presence is shown in the immediate vicinity of the depth of the soil infiltrative surface (60 to 90 cm bgs) compared to deeper soils (greater than 90 cm).

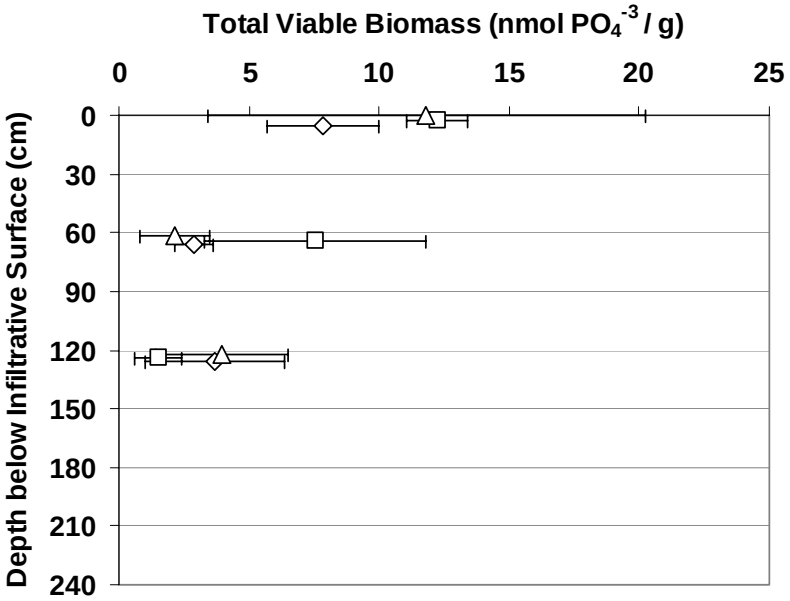


Figure 3-39
Total Viable Soil Biomass Depth Profiles at the Site Before Application of Effluent (10/2002) to the Companion Study Test Cells [31]

[31] Bars are +/- one standard deviation. Three replicate samples collected from locations across the study area.

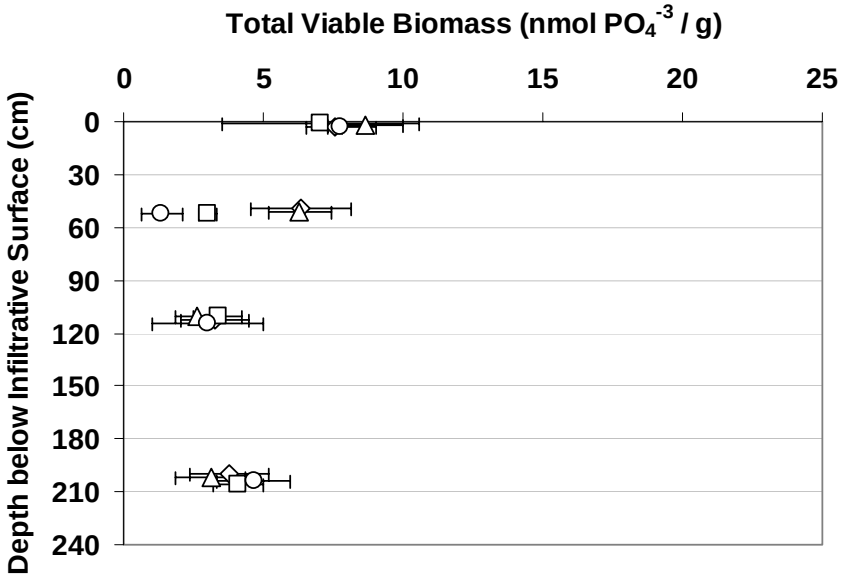


Figure 3-40
Total Viable Soil Biomass Depth Profiles at the Site Before Application of Effluent (10/2003) to this NDWRCDP Study Test Cells[32]

A significant growth of soil biomass was observed as a result of STE infiltration in the upper 50 cm of the soil profile of the test cells. Figure 3-41 shows the soil biomass depth profile revealed in samples collected during July 2004 compared to the average soil biomass concentration of the background cores collected in 2003. Total viable biomass increased by a factor of approximately 10 near the infiltrative surface and decreased exponentially with depth. The biomass depth profile is an indication that most of the biological processes causing organic and inorganic nutrient removal are occurring in the first 50 cm of the vadose zone of the test cells.

[32] Bars are +/- one standard deviation. Three replicate samples collected from locations across the study area.

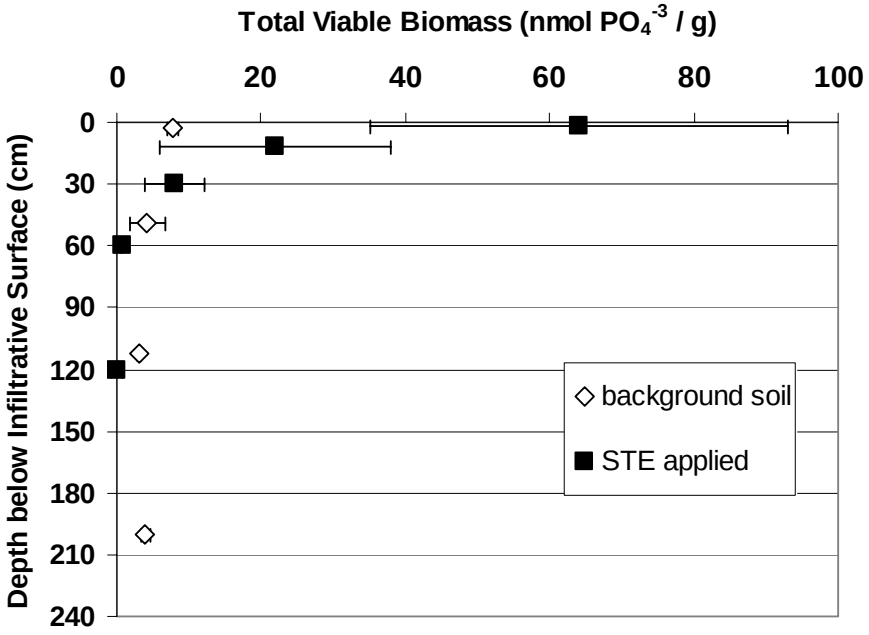


Figure 3-41
Total Viable Biomass in Background Soil and After Application of STE at 8 cm/d for 13 Months on an Open Infiltrative Surface[33]

The test cells (STE at a design HLR of 8 cm/d) and the control cells were analyzed at 0–1, 1–2, 2–4, and 9–10 cm below the infiltrative surface for labile polysaccharide and humic substances. Some additional analyses were also performed at 24 to 25 and 59 to 60 cm below the infiltrative surface for certain cells. Three graphs (Figure 3-42, Figure 3-43, and Figure 3-44) compare the labile polysaccharides, fulvic acid, and humic acid with depth below the infiltrative surface. There appeared to be a trend of decreasing concentration of fulvic acid, humic acid, and polysaccharides with depth below the infiltrative surface in both the test cells receiving STE at 8 cm/d for 13 months and the control cells receiving tap water at 8 cm/d for 13 months. When comparing the test cell results to background soil samples (taken in soil cores collected away from the test cells), one can see concentrations that are comparable with those of the 8 cm/d cells (see graphs).

[33] +/- one standard deviation

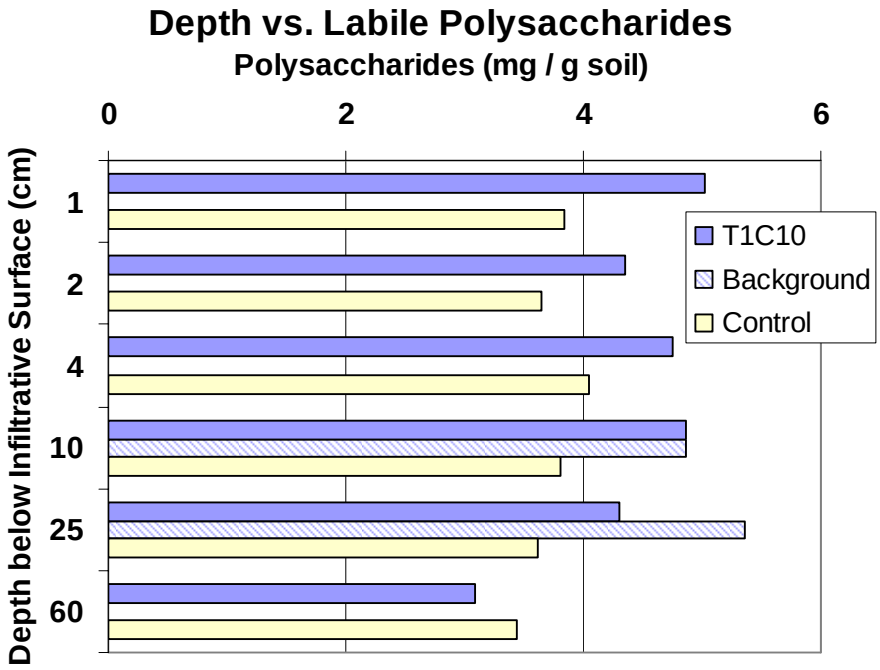


Figure 3-42
Depth Versus Labile Polysaccharides for Companion Study Test Cells Dosed With STE at a Design HLR of 8 cm/d for 13 Months

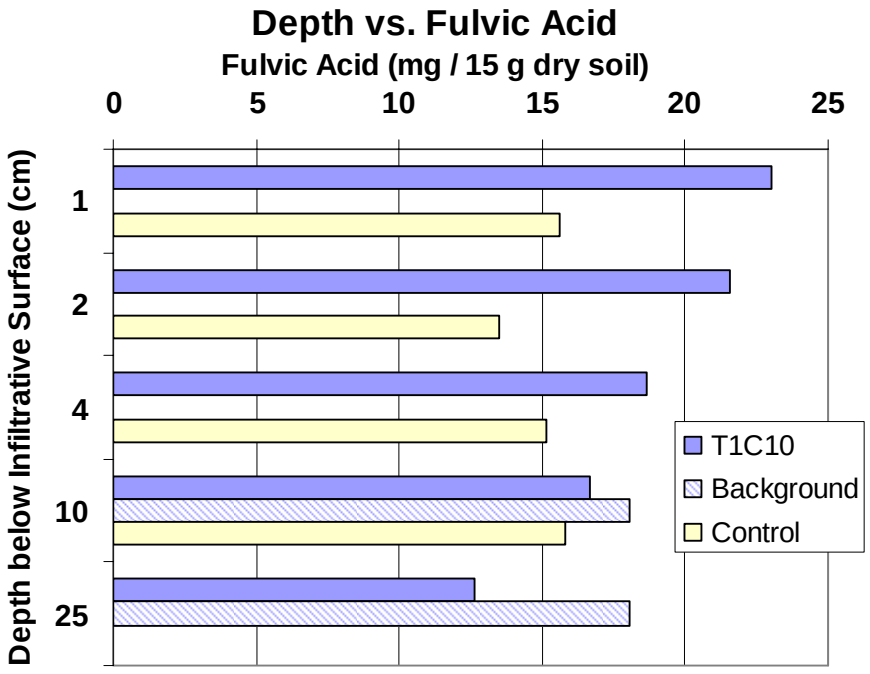


Figure 3-43
Depth Versus Fulvic Acid Content for Companion Study Test Cells Dosed With STE at a Design HLR of 8 cm/d for 13 Months

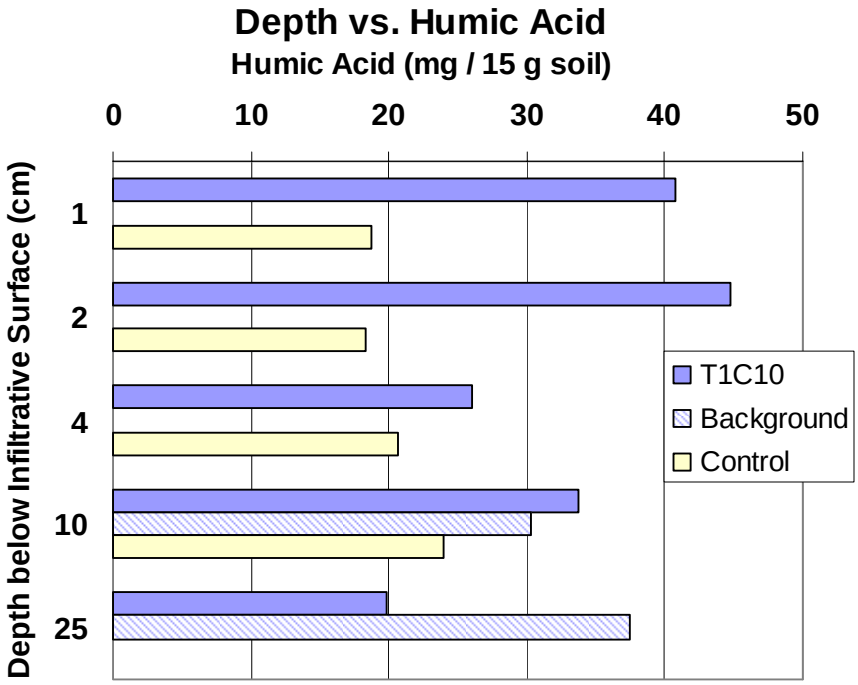


Figure 3-44
Depth Versus Humic Acid Content for Companion Study Test Cells Dosed With STE at a Design HLR of 8 cm/d for 13 Months

3.4 Discussion

This section provides discussion relevant to the changes observed in the infiltrative surface zone of the test cells over the course of this study.

3.4.1 Biozone Formation and Soil Clogging

It is known that wastewater-induced biozone formation in a soil involves the accumulation of pore-filling agents at and immediately below the soil infiltrative surface. At this location, wastewater effluent enters the soil pore network (Figure 3-45) and the reduction in pore size yields a loss in permeability. This loss in permeability affects the hydraulics of the infiltrative surface and the underlying soil profile. It is also known that the rate and extent of biozone development is dependent on several factors, such as soil morphology (Jones and Taylor 1964; Healy and Laak 1974; Bouma 1975), wastewater composition and loading rate (Laak 1970; Siegrist 1987; Duncan *et al.* 1994), and application mode and continuity of use (Siegrist 1987; Hargett *et al.* 1982). This project attempts to examine further, the impact of wastewater effluent quality and loading rate on biozone development. This section will focus on the infiltration rate behavior observed and travel times in the vadose zone below the infiltrative surface. This section will also present some initial modeling in order to predict hydraulic performance of these test cells with the different effluents applied at different design HLRs.

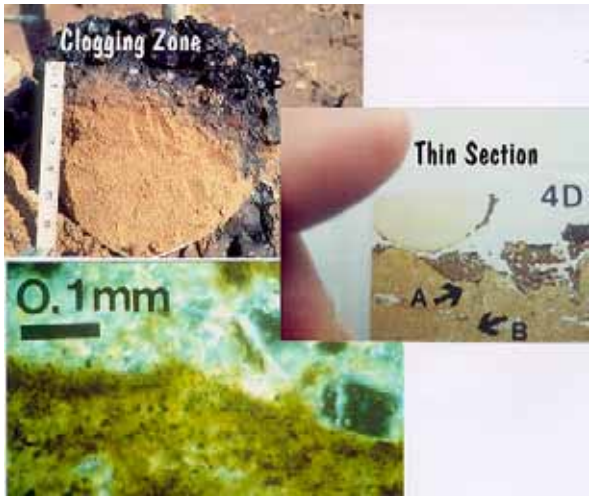


Figure 3-45
Porous Media Showing Layer of Blackened, Organically Rich Wastewater-Induced Biozone (Formerly Referred to as "Clogging Zone") Formation at the Infiltrative Surface (From Siegrist 1986)

3.4.1.1 Infiltration Rate Behavior

The most significant changes in infiltration rate behavior, revealed during the initial six months of operation of the soil test cells, was observed in those cells loaded with STE at a design HLR of 8 cm/d (note that 8 cm/d is four-times the Jefferson County regulatory prescribed design rate for the Ascalon sandy loam soil). These test cells have exhibited continuous ponding, while test cells dosed with STE at a design HLR of 2 cm/d or those dosed with TFU or MBR effluents at design HLRs of 2 or 8 cm/d have only shown sporadic/incipient ponding.

Over time, the application of effluent solids, as measured by TSS, and total BOD (ultimate cBOD + nBOD) can contribute to pore filling at the infiltrative surface and the concurrent establishment of a biozone. This biozone is a biogeochemically reactive zone that can provide more rapid and extensive treatment of the constituents in the applied effluent (for example, by enhanced sorption, nitrification, and biological decay). As the biozone develops, the infiltration rate for wastewater effluent will decline from the baseline rates determined with clean water prior to effluent application. A certain degree of biozone development and the associated infiltration rate loss will improve the treatment of wastewater by causing an unsaturated flow regime below the biozone and improved reaction rates and extents. However, excessive pore-filling can lead to eventual system failure if the biozone becomes essentially impermeable and wastewater can no longer infiltrate (Siegrist 1987; Siegrist and Boyle 1987). This section discusses results from this study as well as information obtained from coring of test cells operated in the companion study at the site.

Figure 3-46 presents the average percent reduction of infiltration rate observed in the test cells after six months of effluent application compared to the baseline rates determined with clean water during pre-startup.

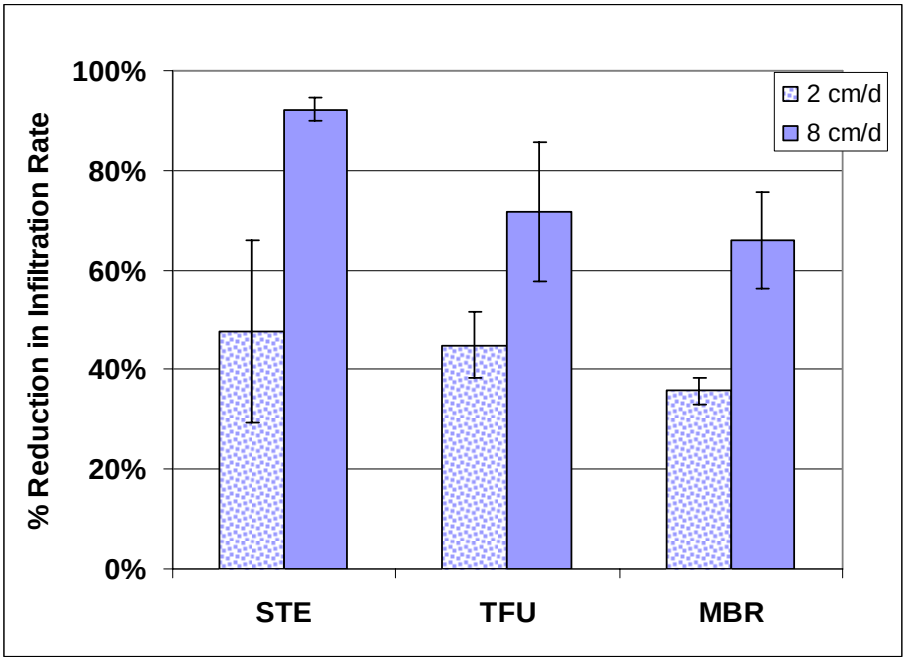
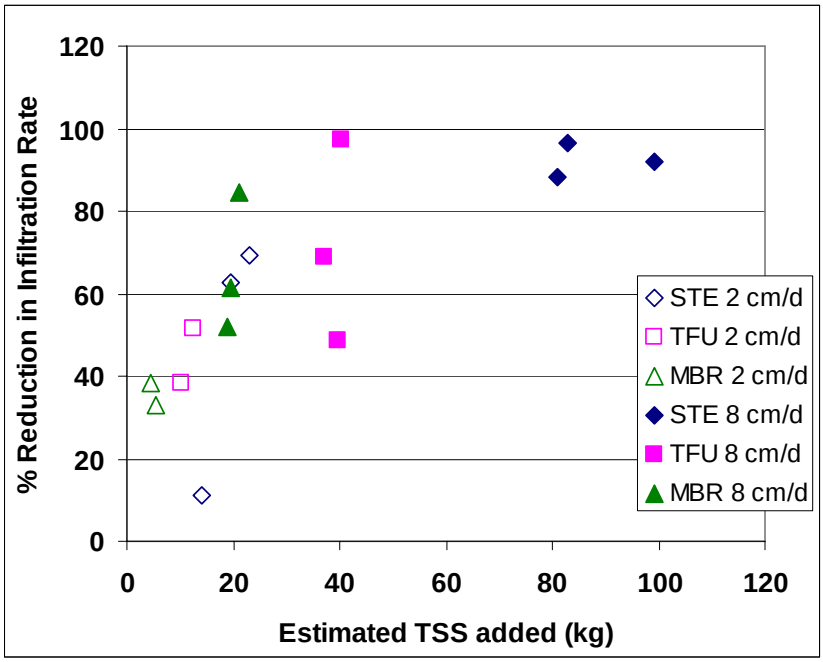


Figure 3-46
Average Percent Reduction in Infiltration Rate, Comparing Six-Month Infiltration Rate to Pre-Startup (Clean Water) Infiltration Rates [34]

The percent reduction in infiltration rate from baseline to six months of effluent delivery was calculated for each test cell. These individual percent reductions were then averaged (excluding the two test cells where an increase in infiltration rate was measured) for each effluent type and delivery rate. Figure 3-46 shows that the most significant reduction of infiltration rate occurred in STE test cells loaded at a design HLR of 8 cm/d. In fact, the 8 cm/d design HLR test cells (regardless of effluent type) had higher reductions in infiltration rate compared to 2 cm/d design HLR test cells.

Similarly, the total mass (kg) of TSS and cBOD added to each test cell was estimated for each individual test cell (three replicates for each loading rate and each effluent type). The estimation was based on actual delivery volume for each test cell and average effluent concentrations for TSS and cBOD for each effluent. Figure 3-47 shows the comparison of the percent reduction in infiltration rate for each individual test cell as a function of how much TSS and cBOD that test cell had received after six months of effluent dosing. This figure allows a comparison of triplicate test cells, and it compares HLR and effluent type.

[34] Error bars indicate +\- one standard error



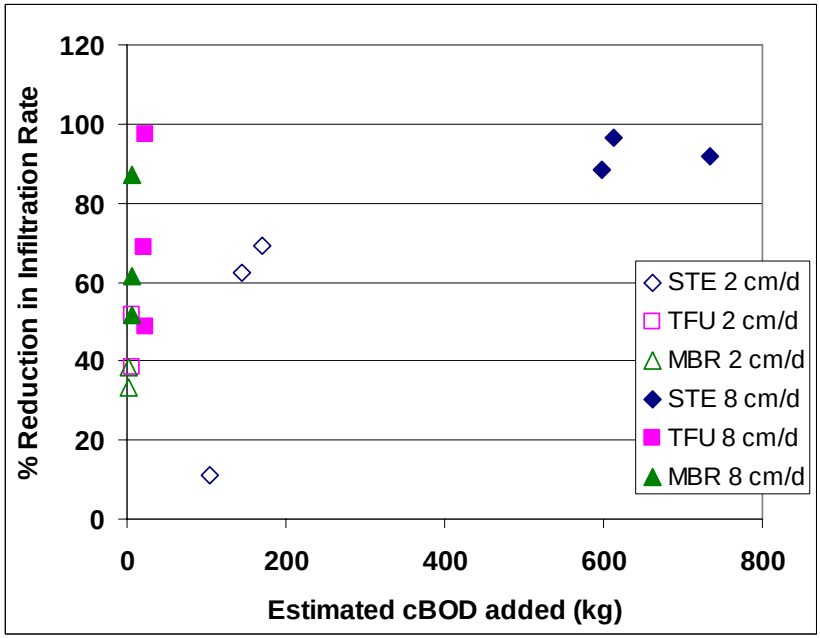


Figure 3-47
Percent Reduction of Infiltration Rate as a Function of Total Mass of TSS and cBOD (kg) Added to the Test Cell (Excluding Rates That Increased Between Baseline and Six Months)

In general, the replicate test cells are clustered together and a general trend of increased reduction in infiltration rate with increased mass of TSS and cBOD added is observed. Two 8 cm/d design HLR test cells, one TFU and one MBR, do not appear to be performing similarly to their replicate cells (top data point for TFU at 8 cm/d and top data point for MBR at 8 cm/d), indicating higher reduction in infiltration rate at a similar TSS and cBOD mass added. For

example, an infiltration rate reduction of 97% at a total TSS mass added of 40 kg for the TFU test cell. This TFU test cell (TBC1) had a baseline infiltration rate of less than half of the average infiltration rate observed across the site (25.2 cm/d compared to average baseline infiltration rate of 62.9 cm/d). The higher reduction in the infiltration rate for this test cell compared to the other replicates is likely influenced by the soil conditions. Alternatively, the MBR test cell (TCC4) had a baseline infiltration rate higher than the average infiltration rate observed across the site (70.0 cm/d). The cause for the higher reduction in infiltration rate for this test cell compared to the replicates is unknown.

3.4.1.2 Shallow Vadose Zone Travel Times

Through the addition of bromide as a conservative tracer during the two tracer tests, the travel times for effluent movement through the vadose zone can be estimated for those test cells instrumented with lysimeters. Travel times in this soil system appeared to increase with time of operation, presumably due to biozone formation and flow regime effects. This is particularly apparent in test cells dosed at a design HLR of 8 cm/d, with the most significant changes revealed in test cells loaded with STE (Figure 3-24). TFU cells dosed at 2 cm/d design HLR showed a significant increase in travel times between pre-startup and three months of effluent dosing. Test cells loaded at 2 cm/d design HLR and dosed with STE and MBR effluent did not reach bromide concentrations that were 50% of the influent concentration. This may be due to the method of delivery and to dispersion in the subsurface (Figure 3-25). However, STE and MBR dosed at a design HLR of 2 cm/d are expected to have had a similar trend had samples been collected and analyzed for a longer period after tracer addition.

In this NDWRCDP study, soil test cells are intermittently dosed once each hour for 90 seconds with three different effluents at 2 or 8 cm/d design HLRs for 16 hours each day. Initial travel times during clean water (pre-startup) tracer tests were 18 and 11 days to 60 cm below the infiltrative surface (2 and 8 cm/d design loading rates, respectively) and were 43 and 32 days to 120 cm (2 and 8 cm/d design loading rates, respectively). Bromide results presented in Section 3.3.3.2 and Figure 3-24 demonstrate an increase in vadose zone travel time to the 60 cm lysimeters with increased time of operation. The most dramatic changes occurred in the STE cells loaded at 8 cm/d design HLR. In addition, because 50% breakthrough concentrations were not observed at 60 cm in the MBR and STE test cells dosed at 2 cm/d design HLR (or in any of the 120 cm lysimeters), dispersion is believed to have occurred and the travel times for approximately 50% of the bromide mass to reach 120 cm after three months of effluent delivery are more than 38 days.

During the three-month tracer test for this NDWRCDP study, tracer was also applied to the companion study test cells. A clean water tracer test was also conducted for the companion study test cells in April 2003. This second tracer test (July 2004) for the companion study was conducted after 13 months of STE application. The companion study test cells include three different infiltrative surfaces that are loaded with only one effluent (STE) delivered continuously for 16 hours each day at design HLR of 4 or 8 cm/d (22 or 44 mL/min., respectively) (Appendix B).

Figure 3-48 presents a comparison of the travel times between the clean water tracer test and the 13-month tracer test for the companion study test cells. Note that two test cells shown, T1C10 and T2C8, have been operated at similar conditions to the NDWRCDP study (STE applied to open infiltrative surface at a design HLR of 8 cm/d), albeit these test cells were loaded with STE for 13 months at the time of the second tracer test (July 2004). These test cells appear to be performing similarly, based on the travel time of 50% bromide breakthrough at 60 cm depth, but differences in travel times are observed to 120 cm (Figure 3-24).

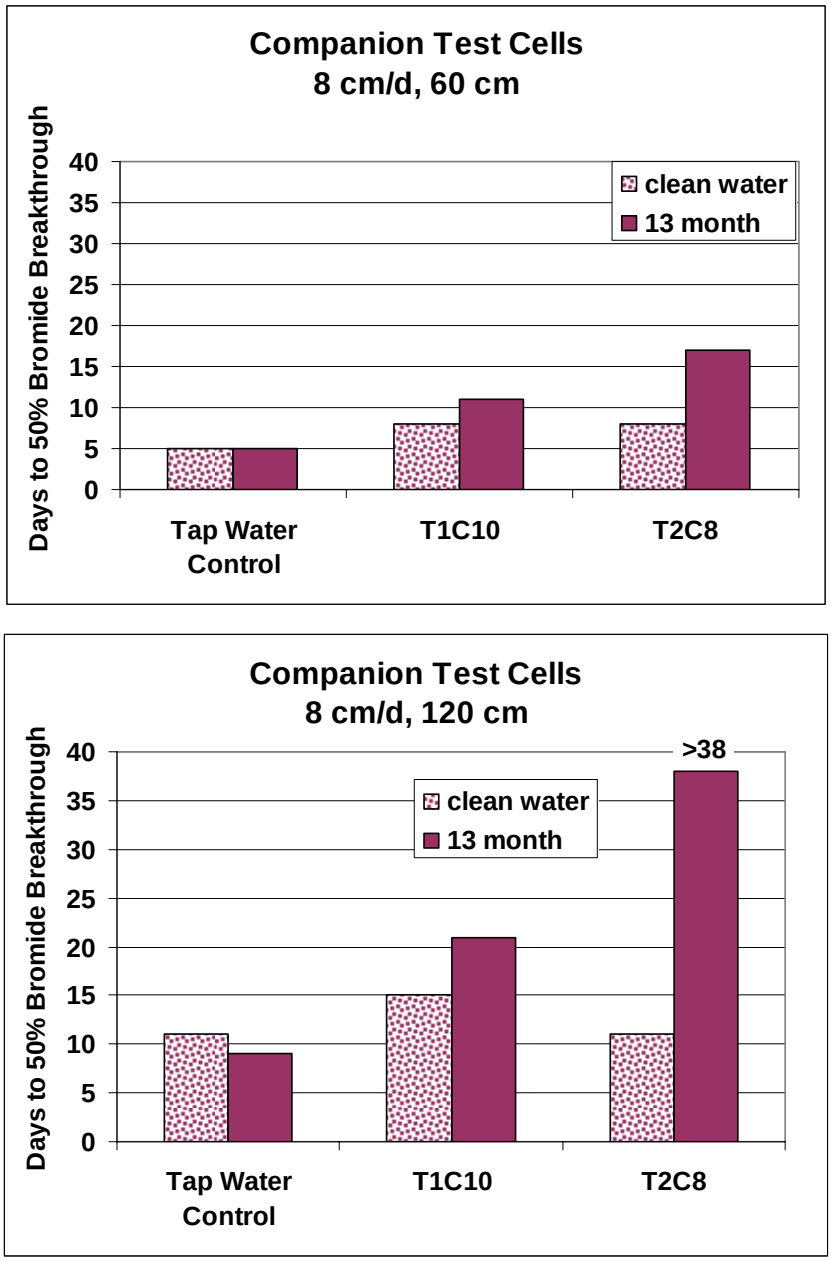


Figure 3-48
Time to 50% Bromide Breakthrough During Clean Water and After 13 Months of STE Application at a Design HLR of 8 cm/d for Companion Study Test Cells

When considering all of the companion test cells after 13 months of effluent delivery (18 total test cells with lysimeters), average travel times were approximately 22 days and 18 days to 60 cm below the infiltrative surface (4 and 8 cm/d design loading rates, respectively). They ranged from 26 to more than 38 days and from 21 to more than 38 days to 120 cm below the infiltrative surface (4 and 8 cm/d design loading rates, respectively). These results indicate that the companion study test cells, loaded at higher or equal rates, had faster travel times for 50% of the bromide mass after 13 months of effluent loading compared to the NDWRCDP test cells after three months of effluent loading at lower or similar design HLRs. Specifically, the hydraulic performance of T1C10 and T2C8 at 13 months is similar or better than the current study's performance (TAC4 and TBC3) after only three months of operation.

Although a three-month tracer test was not conducted on the companion study test cells, an additional tracer test is planned for the NDWRCDP study to enable comparison of all the test conditions at the Mines Park Test Site. It will be interesting to investigate if more dramatic changes will occur in the current study test cells, and to see how the performance (hydraulic and purification) compares to that of the companion study test cells. What is expected to occur is that changes in soil hydraulics will continue in test cells dosed with high-quality effluent in a similar way to that demonstrated in the clean water test cells. Additional evaluation may include a 10 or 25% breakthrough benchmark due to the longer travel times and dispersion that have been observed to date.

Current (or future) differences observed between these two sets of test cells may conceivably be explained by differences in the effluent application methods. This difference is illustrated by the baseline testing at both study areas as shown in Figure 3-49.

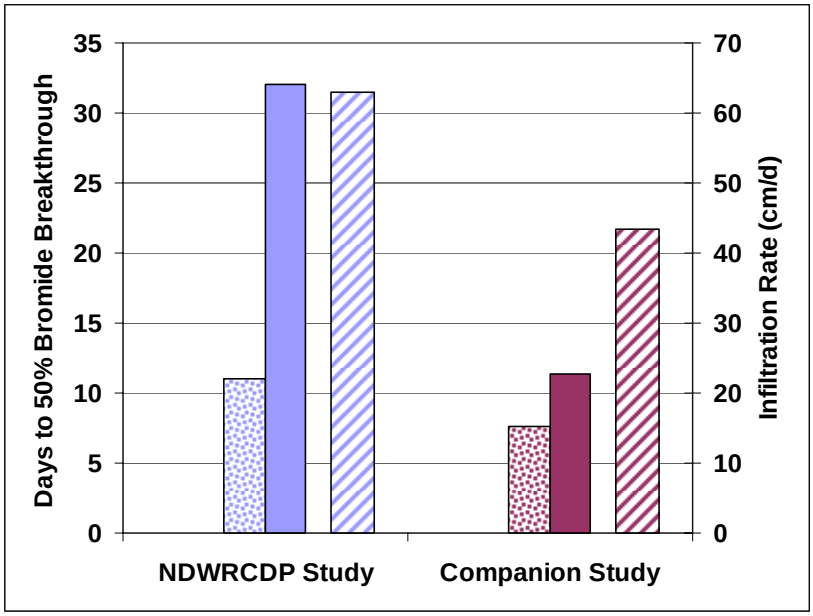


Figure 3-49
Comparison of Baseline Clean Water Characterization[35]

[35] Hatched bar: average travel time to 60 cm lysimeters; Solid bar: average travel time to 120 cm lysimeters; Striped bar: average baseline infiltration rate

During the clean water baseline testing, the average infiltration rate was higher for this NDWRCDP study, but the travel times to 60 and 120 cm below the infiltrative surface were longer (Figure 3-49). In this NDWRCDP study, the test cells are dosed with effluent once each hour. The rate of application of each dose is equivalent to 4.0 or 16.2 cm/s applied to the horizontal infiltrative surface. Since the instantaneous application rate is relatively high compared to the baseline infiltration rate of the clean soil (for example, approximately 0.0007 cm/s), the applied effluent will tend to spread over the entire available soil surface within the test cell as it infiltrates into the vadose zone. This results in an increased use of the test cell infiltrative surface. Therefore, a larger proportion of the test cell infiltrative surface will receive effluent, even before effluent delivery, due to the spreading of the dose as it infiltrates into the soil.

In the companion study, effluent is delivered to the test cells by a continuous trickle that mimics gravity delivery methods used in OWS designs. This application method results in an instantaneous application rate of only about 0.00014 cm/s, which is much less than the baseline infiltration rate of the soil. This application method yields localized infiltration of the effluent near the point of application until the biozone evolves and yields permeability loss. As permeability of the infiltrative surface is reduced, there is concomitant spreading of effluent, infiltration occurs, and an increasing percentage of the available soil infiltrative surface within the test cell is used.

A simple model has been used in the past to estimate travel times in the vadose zone in order to estimate the first-order removal of pollutants (Van Cuyk 2003; Van Cuyk and Siegrist 2004). The model assumes vertical downward plug flow in the vadose zone directly under the infiltrative surface (for example, no evaporative losses, no lateral spreading of water, and no preferential vertical flow). The model can be used to gain a rough estimate of the travel time to a given depth in the vadose zone as estimated by Equation 3.2:

$$t = ((L \times Ne) \times ISU)/q \qquad (3.2)$$

where

t = travel time to a given depth (days)
L = depth (cm)
Ne = effective porosity
ISU = fraction of infiltrative surface utilized (percent of porous media infiltrative surface that actually accepts effluent)
q = HLR (cm/d)

Using this model for the 2 cm/d HLR, and assuming an effective porosity of 0.35 with the entire infiltrative surface utilized (ISU=1), the estimated times for effluent to reach the 60 cm and 120 cm depths are 10.5 and 21 days, respectively. For the 8 cm/d HLR, the times to reach the 60 cm and 120 cm depths are similarly estimated to be 2.6 and 5.2 days, respectively. Average travel times measured at the start of this NDWRCDP study (that is, before effluent addition) were on the order of 11 to 18 days to the 60 cm depth lysimeters at HLRs of 8 cm/d and 2 cm/d, respectively. These measured values (based on the time to 50% bromide breakthrough) are

somewhat higher than estimates based on Equation 3.2. The fact that the measured travel times are longer than the estimated times could be due to processes not being captured in Equation 3.2. For example, evaporative losses of applied effluent as well as dispersion within the soil would cause Equation 3.2 to underestimate the actual solute (for example, Br^-) travel times to depth in the vadose zone. However, in the absence of measured travel times, this simple model provides a conservative estimate of probable travel times.

3.4.1.3 Modeling

Models can be useful tools providing insight into expected outcomes based on a given set of inputs. Evaluation of various scenarios using different expected inputs can aid in design and/or establishing a monitoring program. Research conducted by Siegrist (1987) assessed the comparative infiltration rate loss of different effluent compositions and HLRs. A model was developed to describe the observed infiltration rate loss. This same model was used at the start of this study to evaluate the potential time when reductions in the infiltration rate might be observed. Siegrist (1987) derived an equation to describe the soil clogging due to the biozone development as a function of the cumulative mass density loadings of TSS and tBOD (ultimate cBOD + nBOD):

$$IR_t = 241 \times \{\exp[2.63 - 5.70\,(\text{tBOD}) + 41.08\,(\text{TSS}) - 0.048\,(\text{tBOD} \times \text{TSS})]\} / \{1 + \exp[2.63 - 5.70\,(\text{tBOD}) + 41.08\,(\text{TSS}) - 0.048\,(\text{tBOD} \times \text{TSS})]\} \quad (3.3)$$

where

IR_t = infiltration rate (cm/d) at time, t (days)
tBOD and TSS = cumulative density loadings (kg/m^2 of horizontal surface area) at time, t

To estimate the rate of infiltration rate loss and the anticipated time to ponding (the point at which the infiltration rate of the soil is approximately equal to the HLR), this existing infiltration rate loss model was used. For this model, the total BOD is calculated as the sum of the ultimate cBOD and nitrogenous BOD (nBOD) where cBOD is measured and nBOD was calculated assuming all total Kjeldahl nitrogen (TKN = organic nitrogen plus ammonia) was nitrified. For this study, the model inputs were based on typical effluent qualities from the treatment units (MBR, TFU, and STE) as shown in Table 3-9. Based on the model, initial ponding was expected to occur after operating periods ranging from one month (STE at 8 cm/d) to 2.5 years (MBR at 2 cm/d). The anticipated infiltration rate loss with time is shown in Figure 3-50.

Table 3-9
Typical Effluent Values Used for Model Input

Mass Loading Parameter	STE	TFU	MBR
$cBOD_5$ (mg/L)	200	10	2
TSS (mg/L)	40	10	2
TKN (mg/L)	65	40	15

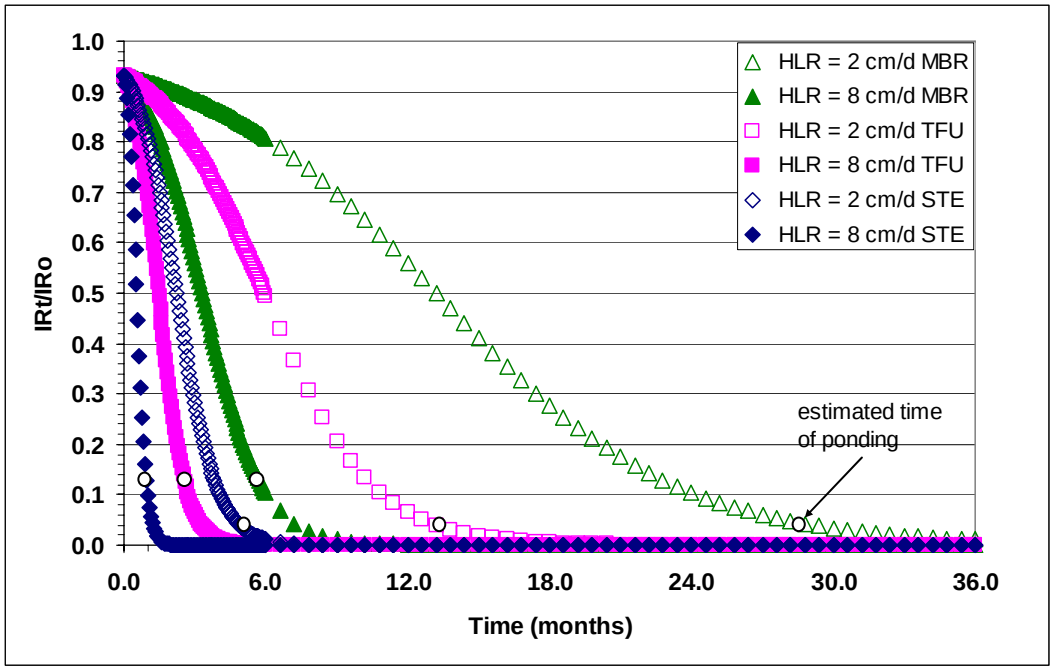


Figure 3-50
Predicted Infiltration Loss With Time as a Function of Effluent Quality (TSS and tBOD) and HLR (Based on Model by Siegrist 1987)

Comparison of field observations to model predictions indicate that the model closely simulated field observations. For example, the model predicted that ponding would ensue after one month of operation for the test cells receiving STE at 8 cm/d design HLR (Figure 3-51).

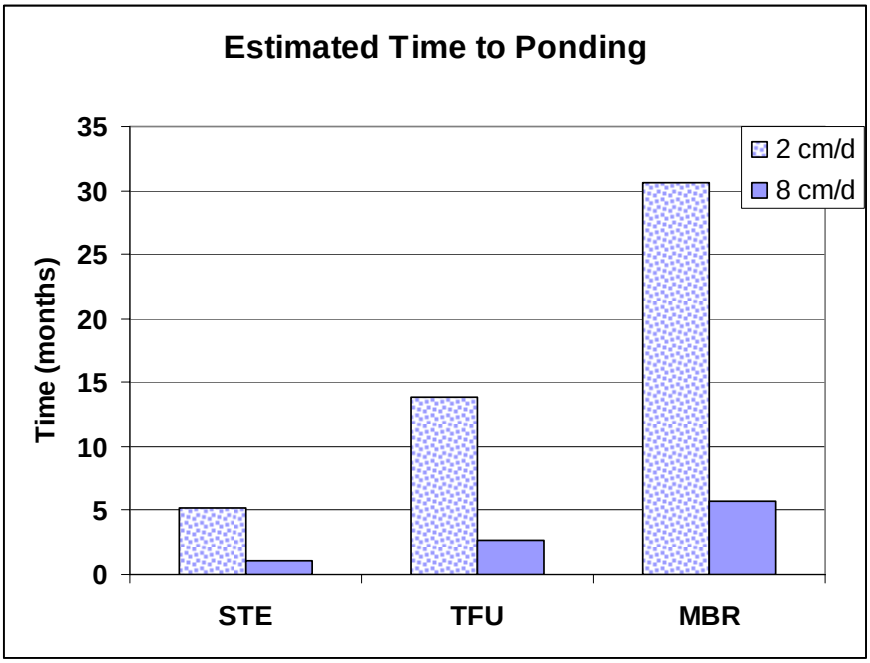


Figure 3-51
Predicted Time to Ponding (HLR = Infiltration Rate) Based on Siegrist (1987)

Ponding was observed in all three STE 8 cm/d design HLR test cells within the first month of effluent delivery (Table 3-6). In addition, after six months of STE delivery at a design HLR of 2 cm/d, intermittent ponding was observed in all three replicate test cells (model predicted 5.2 months). Similarly, the model predicts that the test cells loaded with TFU effluent at 8 cm/d would have become ponded after 2.6 months of operation, and field observations were that two of three TFU test cells had ponded after three months of effluent delivery. All three test cells ponded by six months (Table 3-6). It is projected that test cells loaded with TFU effluent at 2 cm/d will begin ponding after one year of operation. Finally, it is projected that the test cells loaded with MBR effluent at 2 cm/d will exhibit ponding after 2.5 years of operation. Intermittent ponding was observed for the 8 cm/d test cells after six months (model predicted 5.8 months).

3.4.2 Purification Efficiency

The purification efficiency was assessed through nitrogen and phosphorus removal and the presence of bacteriophages and virus. This section provides discussion relevant to the purification efficiency.

3.4.2.1 Conventional Pollutants

The results of soil solution analyses revealed few differences in composition of pore water samples collected from lysimeters at 60 or 120 cm below the infiltrative surface, independent of effluent quality and loading rate. While nitrate was observed earlier in test cells dosed with MBR effluent (due to the nitrification occurring in the unit at the start of effluent delivery to the soil), after six months of operation, STE test cells show the highest concentrations of total nitrogen in lysimeter samples. Because there is no denitrification occurring in the septic tank, these test cells are being loaded with a higher total mass of nitrogen. Both the MBR and TFU removed more than 50% of the total nitrogen found in the STE (after a start-up period of two to three months) (see Figure 2-23).

Figure 3-52 presents total nitrogen removal percentages based on effluent and lysimeter concentrations. The average percent removal was determined by comparing the lysimeter concentration to the corresponding effluent (STE, TFU, or MBR) concentration accounting for estimated travel times in the soil (for example, lysimeter concentration in May compared to the effluent concentration in April).

The average percent concentration removal for total nitrogen in the STE test cells loaded at 2 cm/d design HLR was similar to the TFU and MBR test cells, even though they received a higher total mass of total nitrogen. Test cells loaded at a design HLR of 8 cm/d had slightly lower percent removals. A significant reduction in the percent removal for the 8 cm/d design HLR STE loaded test cells at 60 cm below the infiltrative surface may be attributed to nitrogen cycling within the soil or release of sorbed ammonium as previously discussed. It is expected that with continued monitoring, the average percent removals will reach steady state efficiency.

Purification efficiency of the soil was also evaluated by estimating the total mass removed with soil depth (Dimick 2005). In this case, the mass of constituent applied to the test cell was calculated as the volume of effluent applied to the test cell multiplied by the concentration of constituent (DOC or total nitrogen) in the effluent applied to the soil. Then the mass of

constituent remaining in the soil pore water was determined by multiplying the lysimeter concentration by the volume of effluent moving below the depth location (60, 120, and 240 cm) between soil solution analyses. Travel times of the soil pore water were based on estimated travel times determined from the bromide tracer tests. The delivery volume, effluent concentration, and soil solution concentrations were averaged between measurements. A percentage of mass removed was calculated by comparing the mass in the pore water in each case to the total mass applied over the duration of operation.

Table 3-10 and Table 3-11 summarize the total mass and percent mass removal of DOC and total nitrogen. Similar to the estimates of percent removal based on concentration, a high percentage of the applied mass of total nitrogen was removed in the top 60 cm of soil. All effluents (STE, TFU, and MBR) demonstrated higher percent mass removals of total nitrogen at 120 cm compared to 60 cm. Soil solution samples at the deeper depths (120 cm) tended to have similar or slightly higher DOC concentrations precluding estimation of mass removal (Figure 3-33 and Figure 3-34). A detailed discussion of the mass removal of organic carbon and total nitrogen as well as the complete data can be found in Dimick 2005.

Table 3-10
Mass and Percentage of DOC and Total Nitrogen Removed in 60 cm of Vadose Zone After 196 Days of Operation[36]

Effluent, Design HLR	Lysimeter Identification	Average Actual HLR (cm/d)	Dissolved Organic Carbon		Total Nitrogen	
			Mass Removed (g)	% Mass Removed	Mass Removed (g)	% Mass Removed
STE, 2 cm/d	TBC6-2	1.76	33.2	91	29.3	48
	TCC2-2	1.52	27.8	87	26.3	50
TFU, 2 cm/d	TAC2-2	2.05	6.1	49	14.0	30
	TBC4-2	2.34	10.3	70	15.2	27
MBR, 2 cm/d	TAC3-2	1.72	3.6	52	15.9	46
	TCC1-2	1.57	3.4	53	18.0	56
STE, 8 cm/d	TAC4-2	5.95	114.2	93	80.7	39
	TBC3-2	5.23	97.9	86	32.9	18
TFU, 8 cm/d	TBC1-2	9.74	40.9	67	34.8	15
	TCC6-2	9.51	43.3	72	71.8	31
MBR, 8 cm/d	TAC6-2	6.92	21.8	80	69.4	49
	TCC4-2	7.96	23.7	72	57.0	36

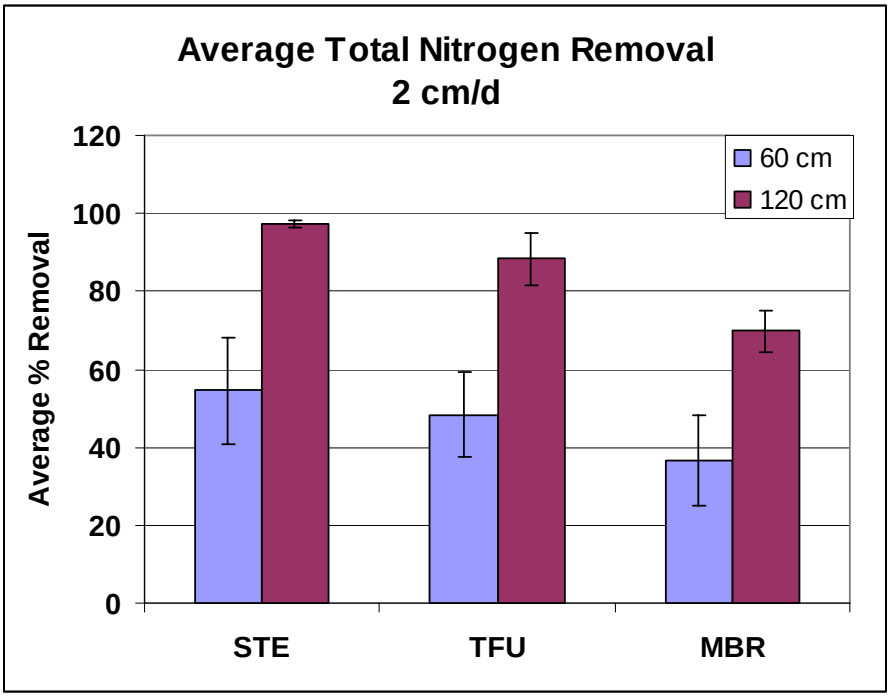
[36] After Dimick 2005

Table 3-11
Mass and Percentage of Total Nitrogen Removed in 120 cm of Vadose Zone After 196 Days of Operation[37]

Effluent, Design HLR	Lysimeter Identification	Average Actual HLR (cm/d)	Total Nitrogen	
			Mass Removed (g)	% Mass Removed
STE, 2 cm/d	TBC6-2	1.76	60.0	99
	TCC2-2	1.52	50.3	95
TFU, 2 cm/d	TAC2-2	2.05	–	–
	TBC4-2	2.34	43.9	78
MBR, 2 cm/d	TAC3-2	1.72	–	–
	TCC1-2	1.57	25.3	78
STE, 8 cm/d	TAC4-2	5.95	179.0	86
	TBC3-2	5.23	176.2	95
TFU, 8 cm/d	TBC1-2	9.74	118.8	51
	TCC6-2	9.51	78.3	34
MBR, 8 cm/d	TAC6-2	6.92	73.2	52
	TCC4-2	7.96	91.0	57

While some total phosphorus was removed in the TFU and MBR treatment units, little-to-no total phosphorus was observed in the lysimeter samples from all of the soil test cells. The average percent removal (comparison of the lysimeter concentration to the corresponding effluent (STE, TFU, or MBR) concentration accounting for estimated travel times in the soil) indicated greater than 99% removal, independent of effluent quality or loading rate.

[37] After Dimick 2005



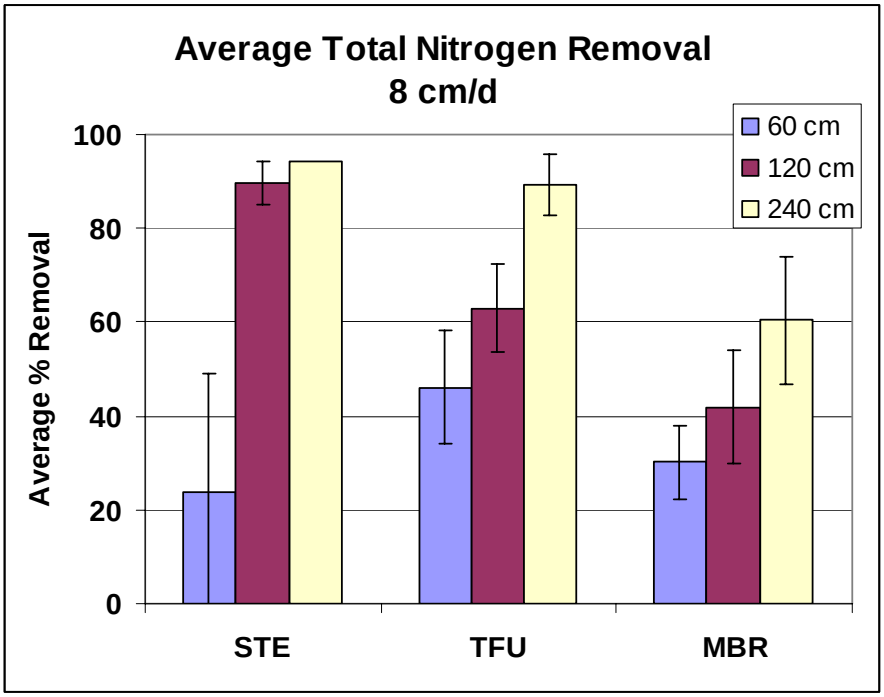


Figure 3-52
Average Percent Removal of Total Nitrogen Through 60, 120, or 240 cm of Vadose Zone [38]

[38] +/- standard error; no error bar indicates only one sample available

3.4.2.2 Bacteria and Virus

Due to the small pore size of the stainless steel suction lysimeters, soil water collected from these samplers would filter out most of any bacteria that might be present in the soil solution drawn into the lysimeter body. However, the lysimeters are appropriate for sampling soil solution for virus, since the lysimeter pore size is larger than a virus particle, and the stainless steel used in the lysimeter fabrication is inert. Sampling and analysis for virus was completed during the surrogate/tracer tests, but there was little breakthrough of the MS-2 and PRD-1 bacteriophages.

Figure 3-53 presents results for MS-2 and PRD-1 breakthrough in all lysimeter samples during the tracer tests completed prior to startup and again after three months of effluent dosing. This figure illustrates that during the clean water tracer test, 31% of 440 total samples analyzed had detectable levels of PRD-1 and 21% of these had detectable levels of MS-2. The average concentration of these samples was 2.8 and 3.0 pfu/mL for PRD-1 and MS-2, respectively.

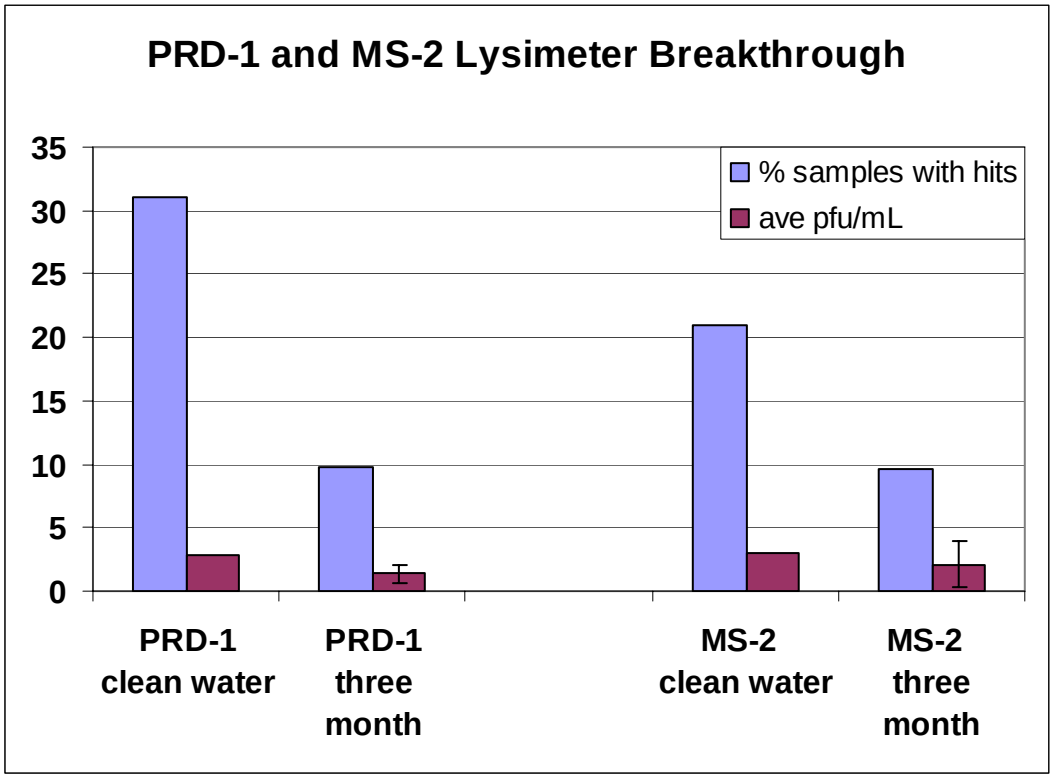


Figure 3-53
MS-2 and PRD-1 Breakthrough Statistics During the Clean Water and Three-Month Tracer Test[39]

[39] Percent of samples with detectable levels of each bacteriophage is shown as average concentration in detectable samples (+/- one standard deviation)

During the second tracer test conducted after three months of effluent loading, only 9.7% of the 392 total samples analyzed had detectable levels of PRD-1 and MS-2. The average concentration of PRD-1 in these samples was 1.3 pfu/mL and 2.1 pfu/mL for MS-2. There appeared to be no differences in virus removal with respect to the type of effluent or the HLR at the time of either tracer test (Figure 3-54).

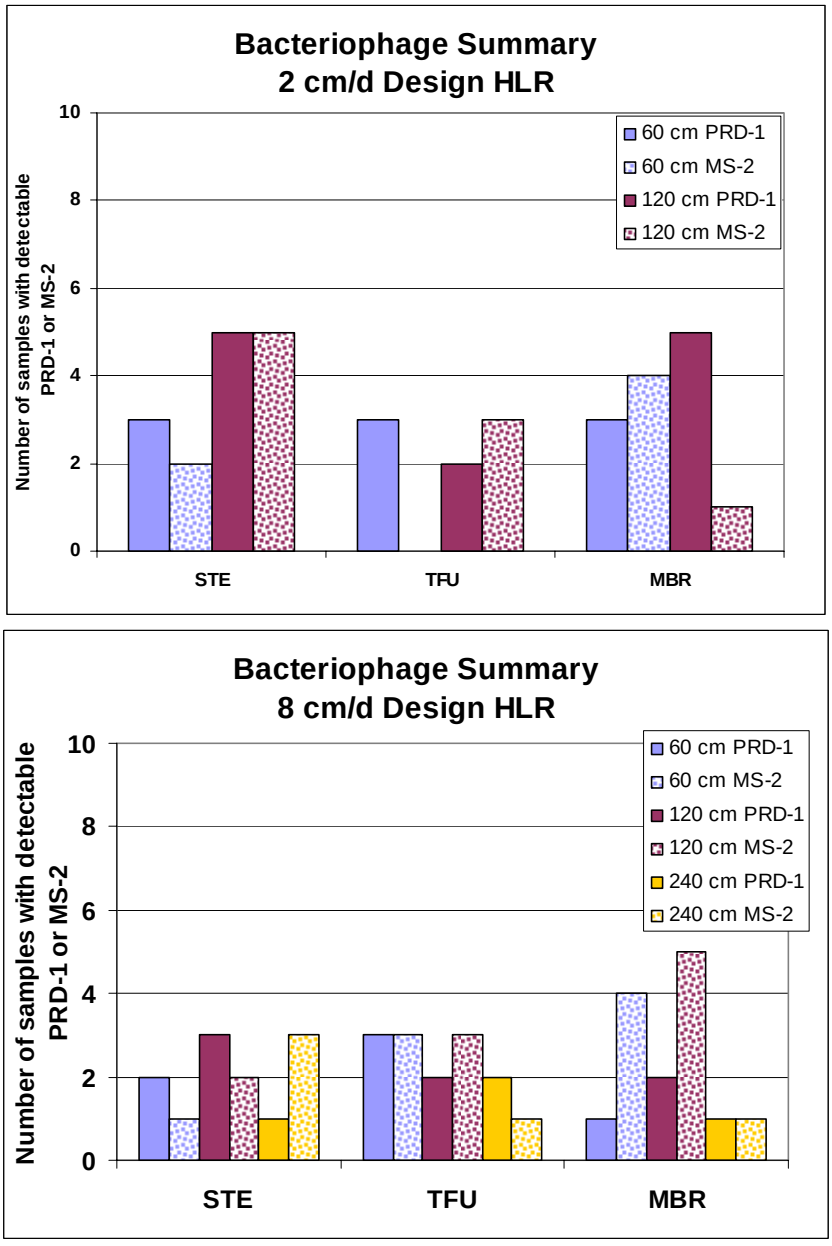


Figure 3-54
Summary of Bacteriophage Breakthrough Based on Detectable Levels During the Three-Month Tracer Test [40]

[40] A total of 392 samples were collected

Based on the sampling and analysis of soil solution below the infiltrative surface of the test cells, very high removal efficiencies for the MS-2 and PRD-1 virus were achieved during three months of effluent infiltration and movement through as little as 60 cm of the vadose zone in Ascalon sandy loam soil loaded with the different effluents at design HLRs of 2 or 8 cm/d. To gain insight into the reasonableness of the total percent or log removal of the added viral surrogates, the following analyses were completed. The bacteriophages were added to the STE on an actual delivery concentration of 1.9×10^4 pfu/mL for MS-2 and 1.4×10^5 pfu/mL for PRD-1 for 21 consecutive days while soil solution sampling and analyses continued for 38 days. Assuming that the virus concentrations in the soil solution pulled into a lysimeter below a test cell are representative of those in the pore water in the vadose zone (vacuum was pulled for a 48-hour sample collection interval), the highest pore water concentration of MS-2 or PRD-1 was equal to 11 or 5 pfu/mL, respectively. These concentrations equate to an estimated total percent removal of approximately 99.9% or a 3-log removal for MS-2 and 99.99% or 4-log removal for PRD-1 during effluent movement through 120 cm of soil.

The US EPA is proposing a risk-based regulatory strategy for all groundwater systems. This Ground Water Rule (GWR) will attempt to reduce public health risk associated with the consumption of waterborne pathogens from fecal contamination (US EPA 2000). The two main goals of this rule are:

- Define and categorize which drinking water wells are susceptible to viral contamination
- Develop regulatory means to protect drinking water wells from viral contamination

This rule will contain a requirement for correction of fecal contamination by eliminating the source of contamination, providing an alternative source water, or providing treatment which achieves at least 99.99 percent (4-log) inactivation or removal of viruses. Twelve-log removal of viruses from "toilet to tap" has been proposed based on risk estimates for two viruses in 10^7 L of drinking water to obtain a risk of infection of 1 in 10,000. If 4-log removal can be achieved during transport in groundwater from the point of recharge of an OWS to a drinking water well (as will be required by the GWR), an additional 8-log reduction must occur in the OWS from toilet to groundwater (for example, virus removal in a septic tank or textile filter plus removal during unsaturated flow through 60 to 120 cm of vadose zone soil).

Virus transport distance and transport times estimated by models have been found to be highly sensitive to the choice of attachment and inactivation rate coefficients (Yates 1995; Navigato 1999). These parameters and the processes that control attachment and inactivation are not readily available or well understood. Limited information is available in systems with wastewater effluent impacts. Previous work has been conducted to address the issue of virus fate and transport. Information on the attachment and inactivation/die-off behavior has been gathered in the laboratory using material and temperatures representative of field conditions following established methods (Loveland *et al.* 1996; Harvey 1997; Navigato 1999; Van Cuyk *et al.* 2001a; Van Cuyk 2003; Van Cuyk and Siegrist 2004). In these studies, the same two bacteriophages, MS-2 and PRD-1, were used as surrogates for human pathogenic enteric viruses.

The removal of viruses in wastewater effluent (non-disinfected) released into the subsurface may depend almost completely upon the permanent attachment of viruses to subsurface solids and/or their inactivation due to strong inter-surface forces occurring during reversible or intermittent attachment. Models used to predict the transport of virus must include the loss of virus from soil solution or groundwater due to the attachment based on the physical and chemical properties of the soil and the groundwater. The effects of temperature and pH, and the presence of organic carbon on attachment have been studied at the laboratory scale and may be important factors in virus removal. In addition, the removal of viruses in the infiltrative surface zone was investigated by Van Cuyk (2003) to understand the effects of effluent type, style of effluent delivery (micro dosing or larger daily doses), and soil type.

Inactivation is assumed to be a first-order kinetic process described by the following equation (Gerba *et al.* 1991; Powelson and Gerba 1994):

$$C = C_o\, e^{-k_i t} \tag{3.4}$$

where

C = concentration at time, t (pfu/mL)
C_o = initial concentration (pfu/mL)
k_i = inactivation rate (time^{-1}).

The literature suggests that little temperature-induced inactivation will be observed at temperatures below about 10 ^{o}C. The results of monitoring of subsurface temperatures in this study revealed temperature ranges of 8.9 to 18.3 ^{o}C at 60 cm with slightly lower temperatures at deeper depths.

As Table 3-12 presents, the range of inactivation rates reported in the literature spans two orders of magnitude. In Figure 3-55, the time required to achieve 1- or 4-log removal of viruses is shown as a function of inactivation rate (six rates were used: 10, 1, 0.1, 0.01, 0.001 day^{-1}). Depending on the rate of inactivation, 4-log removal of viruses by inactivation can be rapid (many hours to days) or may take years.

A few assumptions were made to calculate k_i values for virus removal in the soil test cells studied in this field experiment. A consistent influent concentration of 1.9×10^4 pfu/mL for MS-2 and 1.4×10^5 pfu/mL for PRD-1 were assumed with delivery of this concentration for 21 consecutive days. For this calculation, an HLR of 2 cm/d and a travel time to 60 cm below the infiltrative surface of 20 days was used.

With an average virus concentration of 2 pfu/mL for MS-2 and 1 for PRD-1 in the pore water sample at 60 cm (see Figure 3-53) k_i values of 0.46 day^{-1} and 0.59 day^{-1} were calculated for MS-2 and PRD-1, respectively. These virus removal rates fall within the range of k_i values observed by other investigators who have studied virus transport in vadose zone soils (see Table 3-12; Reddy *et al.* 1981; Van Cuyk 2003).

Table 3-12
Calculated Inactivation Rates (k_i) for Viruses Observed and Reported by Other Investigators Under Various Conditions

Researcher	Porous Media	Temp (C)	Rate, k_i (day^{-1})	Virus Used	Comments
Bertucci *et al.* (1974)	No	12	2.21	Echovirus 11	Anaerobic digestion
Bertucci *et al.* (1974)	No	12	2.53	MS-1	Anaerobic digestion
Larkin *et al.* (1976)	Yes		0.1	Poliovirus	Soil flooded with inoculated, non-chlorinated secondary effluent
Navigato (1999)	No	5	0.022	PRD-1	Contaminated groundwater
Navigato (1999)	No	5	0.056	PRD-1	Radiolabeled phage
Navigato (1999)	No	5	0.083	MS-2	Contaminated groundwater
Navigato (1999)	No	5	0.093	MS-2	Radiolabeled phage
Powelson *et al.* (1990)	No	4	0.041	MS-2	Groundwater
Reddy *et al.* (1981)	Yes and No		0.04–3.69	Many	Compilation of data
Schijven *et al.* (1999)	No		0.12	PRD-1	Groundwater
Schijven *et al.* (1999)	No		0.030	MS-2	Groundwater
Van Cuyk *et al.* (2001a); Van Cuyk (2003)	Medium sand	18	0.26–1	PRD-1	Wastewater, too little MS-2 breakthrough to measure
Yates (1995)	No	4	0.018–0.15	MS-2	Groundwater
Yahya *et al.* (1993)	No	7	0-0.092	MS-2, PRD-1	Groundwater

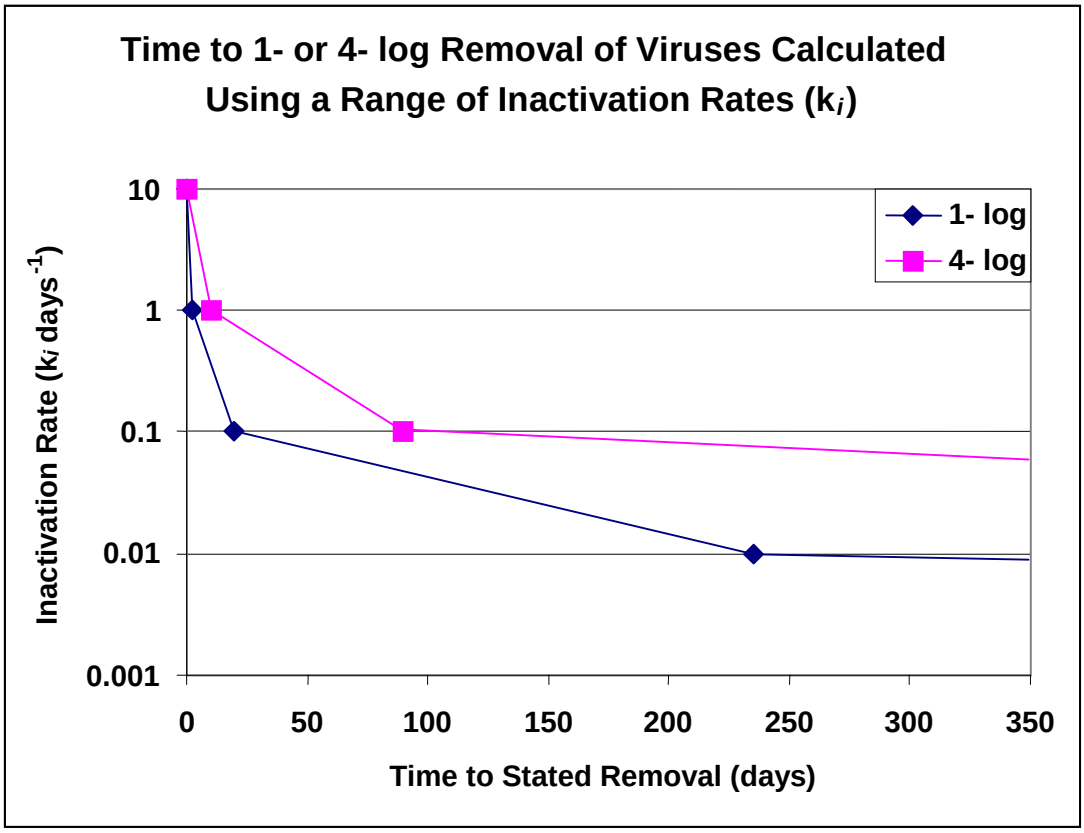


Figure 3-55
Time to 1-or 4- Log Removal of Virus Calculated Using a Range of Inactivation Rate (k_i) Values

4 OWS PERFORMANCE

4.1 Conceptual Model of OWS Performance

In the past, OWSs were viewed as temporary and short-term solutions for wastewater management in many areas. They were to be used only until centralized sewerage became available. Today, decentralized wastewater management involving OWSs—used individually or in clusters—is viewed as a necessary and appropriate component of a sustainable wastewater infrastructure (US EPA 1997). To fulfill this role, OWSs must provide reliable treatment to a degree that protects public health and water quality. OWSs can include various unit operations as illustrated in Figure 4-1. The top portion of this schematic portrays the most commonly implemented OWS, which uses a septic tank unit followed by application of the STE to a subsurface soil treatment unit before recharge to groundwater under the site.

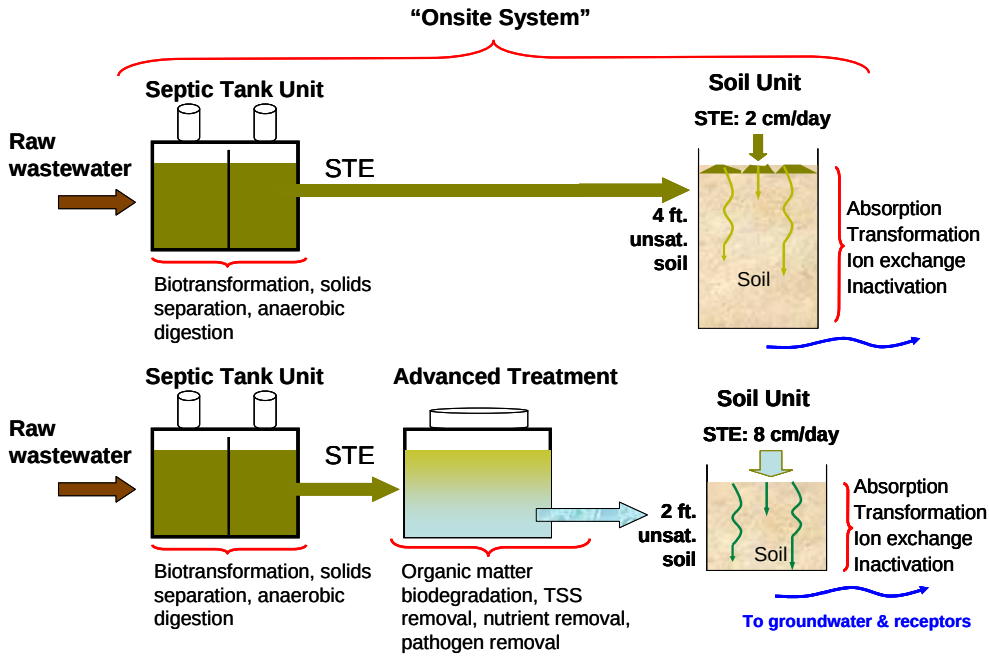


Figure 4-1
Schematic of Typical OWS Treatment Trains[41]

[41] Top figure represents a conventional treatment train; bottom figure represents treatment train with an advanced treatment unit

Figure 4-1 also presents another type of OWS design. In this OWS, there is an additional tank-based unit in the overall treatment train and the generation of higher-quality effluent that is then applied to the soil. This type of OWS is being promoted and used to accomplish two primary goals:

1. Enhance the protection of public health and environmental quality—usually based on a reduction in nutrients and pathogens released in treated wastewater effluents—through engineering controls rather than natural systems (for example, treatment in soil and groundwater)

2. Provide adequate treatment in areas of special concern, such as where there are already high nitrogen levels in water resources (for example, due to agricultural runoff) or in locations where site conditions are challenging for a conventional OWS (for example, where there is limited soil depth or small lot sizes). In these areas, the application of higher-quality or "pretreated" effluent may occur at higher HLRs or in conditions where shallower depths of unsaturated zone exist.

Additional goals may include improved serviceability, operation, and maintenance.

The principal risks that OWSs are designed to mitigate are:

- Direct human exposure to partially-treated wastewater caused by hydraulic failures, which result in wastewater backing up into the dwelling or seeping to the ground surface
- Contamination of groundwater by nitrates or pathogens that are not effectively treated in the OWS or assimilated in the subsurface before the groundwater is used for drinking water
- Pollution of surface waters by nutrients or pathogens, which can pose a risk of infection or perturbation of an ecosystem

More recently, human health and ecological risks associated with emerging organic chemicals such as pharmaceuticals and personal care products are also growing in concern. Ideally, an OWS should perform reliably and achieve the desired risk management goals over a design life, which can be 10 to 20 years or more.

4.2 Comparing OWS and Performance

A growing and potentially vexing question concerns the degree of purification that could be reliably and effectively accomplished in a tank-based treatment unit (for example, sand filter, textile media biofilter) versus a natural soil treatment unit. Tank-based treatment units can be engineered allowing process control and monitoring to optimize and assure system performance. However, these desirable attributes may require a more complex unit operation with more electro-mechanical components, energy and chemical consumption, and more operation and maintenance. In contrast, soil treatment units are inherently complex due to the varied processes that contribute to wastewater purification, which require suitable site and environmental conditions to perform properly. But properly designed and implemented, soil treatment units can

be robust in performance, require minimal operation and maintenance, and have long service lives.

A comparative evaluation of the performance of the three treatment trains studied in this research (septic tank with soil treatment; septic tank, TFU, and soil treatment; septic tank, MBR, and soil treatment unit) can be made relative to various aspects of hydraulic function, purification efficiency, and operation and maintenance requirements. Other metrics, such as service life and cost, are also important.

With respect to hydraulic function, additional tank-based treatment to produce an effluent of a higher quality than typical STE can retard biozone development and enable application of higher daily loading rates to soil, and concomitantly smaller soil treatment units (assuming purification is reliably achieved over the service life of the system). However, the magnitude of the increases in effluent application rates enabled by higher effluent quality is limited by the hydraulic properties of the natural soil. Even relatively clean water can cause reduction in soil acceptance rates and potential hydraulic failure of the soil treatment unit. In a companion study, application of City of Golden tap water at 4 and 8 cm/d design HLRs stimulated soil clogging and led to continuous ponding of the 8 cm/d test cells after one year (Ascalon sandy loam soil) (Figure 4-2).

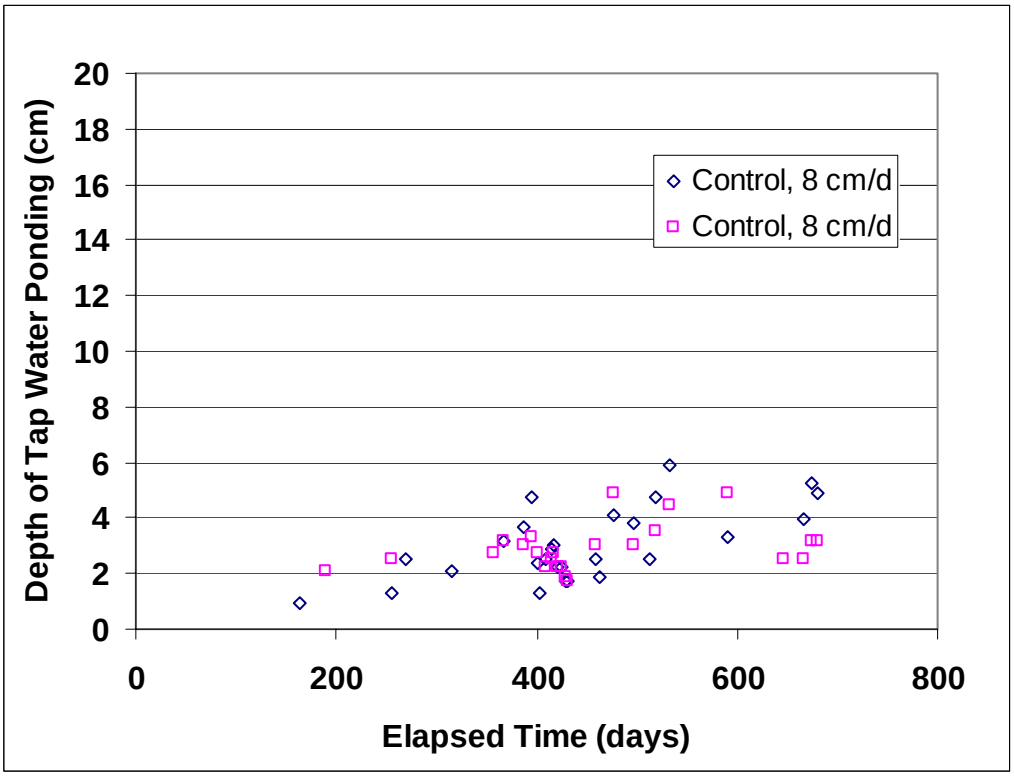


Figure 4-2
Continuous Ponding of Tap Water on Control Cells Indicating a Reduction in Soil Infiltration Rate Due to Delivery of Tap Water at 8 cm/d [42]

[42] Soil K_{sat} 41.8 cm/d

After two years, the test cells receiving tap water at a 4 cm/d design HLR are not continuously ponded. While limited data are available to support a firm recommendation, it seems reasonable to limit the daily design HLR for a given soil treatment unit to a small percentage of the soil's K_{sat} (for example, design daily HLR = 3 to 5% of the K_{sat}).

With respect to the complete treatment train purification, adding a tank-based treatment unit (for example, TFU or MBR) to process STE before discharge to soil can increase the purification achieved prior to soil treatment. In this research, the three treatment units examined achieved different treatment efficiencies for organic matter, solids, nutrients, and bacteria. The relative efficiency ranking was: STE << TFU << MBR. The treatment efficiency was determined by comparison of the TFU and MBR effluents to the STE and the percent removal calculated on a concentration basis (see Section 2.5.3.2). However, the relative ranking for operational complexity, operation and maintenance requirements, energy use, and cost followed a similar pattern: STE << TFU << MBR. Due to the short duration of the performance evaluation completed, it is difficult to estimate the service life of the OWSs employing TFUs or MBRs or their long-term operation and maintenance requirements and life-cycle costs.

A key question is: How will a higher-quality effluent applied to soil affect the quality of the water that exits a soil treatment unit (for example, soil pore water 60 or 120 cm deep in the vadose zone)? The percentage of the total cumulative mass removed of each treatment train was estimated by comparing the estimated total mass applied to the treatment train (that is the mass of DOC or total nitrogen in STE) to that of the total mass estimated to be leaving the treatment train as measured in lysimeter samples. As illustrated in Figure 4-3, the removal efficiencies for DOC and total nitrogen, by either soil treatment alone, or with TFU- or MBR-soil treatment, indicate a trend of higher mass removal given by: septic tank-MBR-soil treatment > septic tank-TFU-soil treatment > septic tank-soil treatment at 60 cm below the infiltrative surface. For total nitrogen, an increase in the vadose zone increased the mass removal, but the difference in mass removal efficiency between treatment trains is smaller. Soil solution samples at 120 cm tended to have similar or slightly higher DOC concentrations than the 60 cm soil solution samples. These results are unexpected but possibly could be due to the occurrence or alteration of naturally occurring organic carbon in the soil (Dimick 2005).

The purification performance of the three treatment trains can be compared by examining the concentrations of potential pollutants and pathogens in the subsurface discharging from the system (such as, the soil pore water at 60 or 120 cm below the soil infiltrative surface). Table 4-1 presents soil pore water concentrations for DOC, total nitrogen, and nitrate as measured in lysimeter samples collected during months five and six of operation. As shown in this table, the treatment train purification for DOC and nitrogen generally follows a trend of higher performance that is given by: septic tank-MBR-soil treatment > septic tank-TFU-soil treatment > septic tank-soil treatment. The treatment trains including a TFU or MBR generally perform better with respect to purification of these constituents, and they are less affected by HLR than the system based on only STE and soil treatment (Table 4-1). In addition, the comparative performance of treatment trains with a TFU or MBR is relatively better than soil treatment alone with 60 cm of soil. However, increasing the vadose zone soil depth (for example, from 60 to 120 cm) tends to shrink the differences in treatment train performance between the three system types.

Removal of virus in soil treatment when higher-quality effluents are applied at higher HLRs or with less soil depth is an important consideration. The results of this research revealed that the soil's ability to remove virus was quite high. The soil's virus removal ability was also insensitive to whether the natural soil had received STE, TFU effluent, or MBR effluent at either 2 or 8 cm/d design HLRs. Table 4-2 presents average concentrations of viruses measured in pore water samples over 38 days of sample collection during the three-month tracer test.

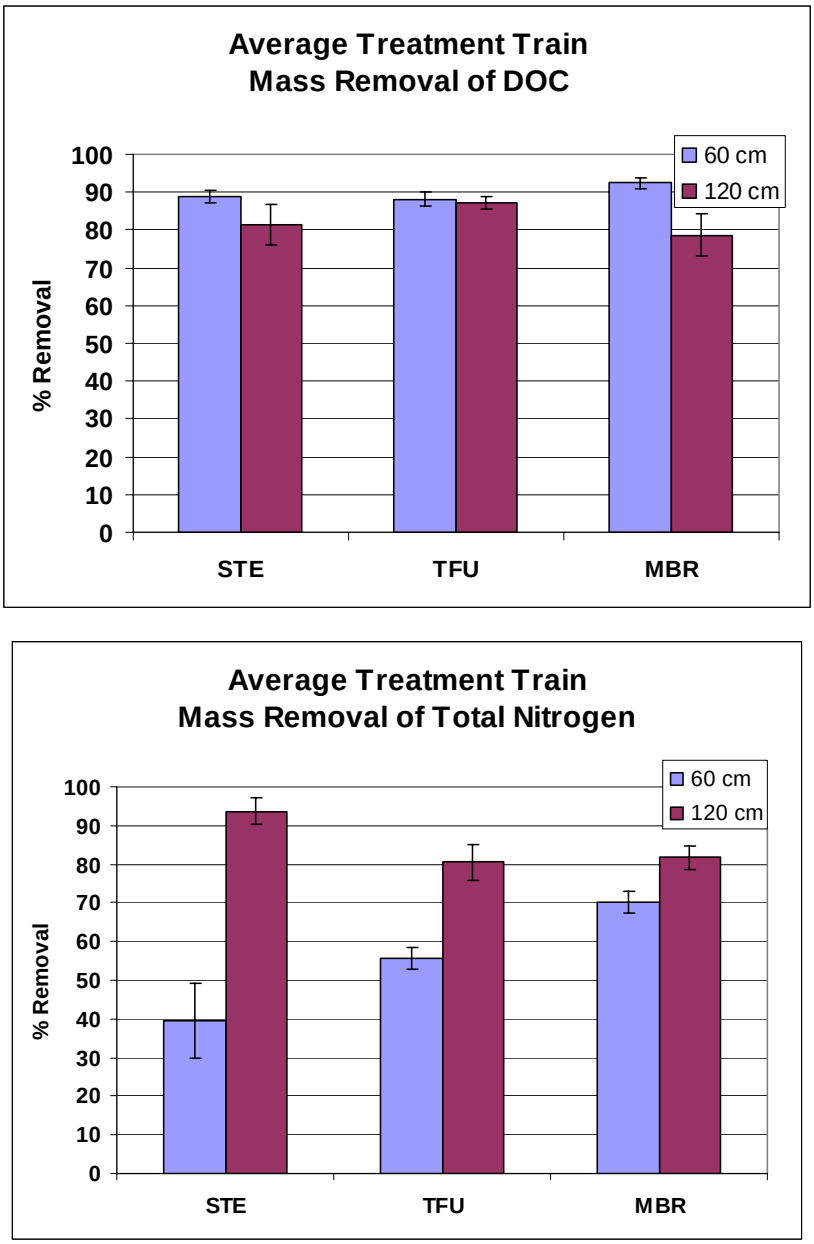


Figure 4-3
Average Percent Concentration Removals of DOC and Total Nitrogen for Treatment Trains: Septic Tank-Soil Treatment, Septic Tank-TFU-Soil Treatment, and Septic Tank-MBR-Soil Treatment[43]

[43] Average of results for both 2 and 8 cm/d design HLRs

Table 4-1
Concentration of DOC, Total Nitrogen, and Nitrate in Soil Pore Water Samples Collected From Test Cells Dosed With Septic Tank, TFU, or MBR Effluents at 2 or 8 cm/d Design HLRs

Lysimeter Depth and Parameter	Design HLR = 8 cm/d			Design HLR = 2 cm/d		
60 cm	**MBR**	**TFU**	**STE**	**MBR**	**TFU**	**STE**
DOC in mg-C/L	4.1 (2.7)	4.2 (3.6)	10.5 (9.4)	3.6 (1.9)	7.5 (5.9)	14.1 (7.9)
Total nitrogen in mg-N/L	18 (2.2)	34.5 (18.7)	85.1 (10.7)	20.8 (4.1)	34.4 (11.1)	57.5 (16.7)
Nitrate nitrogen in mg-N/L	8.9 (9.5)	22.1 (14)	34 (9.4)	5.5 (4.0)	7.8 (9.4)	15.7 (4.6)
120 cm						
DOC in mg-C/L	4.1 (1.3)	6.9 (5.7)	4.3 (0.9)	7.4 (4.4)	5.7 (2.9)	9.7 (3.5)
Total nitrogen in mg-N/L	14.2 (4.4)	32.5 (8.3)	15.7 (13.0)	10.8 (6.2)	14.8 (11.6)	3.2 (2.3)
Nitrate nitrogen in mg-N/L	3.8 (2.6)	10.2 (11.9)	10.7 (7.8)	5.5 (7.5)	10.1 (11.8)	0.7 (0.5)

Averages of samples collected during months five and six; duplicate test cells combined; standard deviation in parentheses

Table 4-2
Average Concentration of Viruses in Soil Pore Water Samples as Measured During the Three-Month Tracer Test

Lysimeter Depth and Virus	Design HLR = 8 cm/d			Design HLR = 2 cm/d		
60 cm	**MBR**	**TFU**	**STE**	**MBR**	**TFU**	**STE**
MS-2 (pfu/mL)	<3	<3	<3	<3	<3	<3
PRD-1 (pfu/mL)	<3	<3	<3	<3	<3	<3
120 cm						
MS-2 (pfu/mL)	<3	<3	<3	<3	<3	<3
PRD-1 (pfu/mL)	<3	<3	<3	<3	<3	<3

Average of 38 days of sample collection

While all test cells received MS-2 and PRD-1 at actual concentrations of 1.9×10^4 and 1.4×10^5 pfu/mL, respectively, there are no differences in average values measured with respect to effluent type, HLR, or soil depth. The results of bromide tracer tests, infiltration rate measurements, and modeling reveal that some degree of soil clogging and biozone formation is occurring in the Ascalon sandy loam soil. The clogging and biozone formation is occurring even with higher-quality effluents applied, and viruses are effectively removed (removal in soil of about 6-logs). These results refute concerns that virus removal in soils receiving high-quality effluents might be diminished due to the absence of a classic biozone resulting from the low levels of tBOD and TSS applied.

These results suggest that for a site with only 60 cm of vadose zone, addition of a treatment unit such as a TFU or MBR would reduce the pollutant load discharged to groundwater while having no negative impact on removal of virus. The relative purification benefits of higher-quality treatment prior to soil treatment are diminished with greater vadose zone depths. In the case of a system employing a highly-advanced treatment process like the MBR, even with only 60 cm of soil, a HLR of 8 cm/d (approximately four times the regulatory prescribed design HLR) can still provide comparable or improved performance compared to the other two systems studied. However, it is important to caveat these statements since the performance evaluation completed in this research is based on a relatively short-term period of observation. Longer-term performance data is needed to solidify the findings of the work.

5 CONCLUSIONS AND RECOMMENDATIONS

5.1 Conclusions

This research investigated the field performance of OWSs employing engineered treatment units followed by soil treatment, to enable higher HLRs and/or less unsaturated soil depth. The primary objectives of the research were:

- Delineate the effluent quality with respect to chemicals and pathogens over time from three treatment units that produce effluents of differing quality (septic tank, septic tank with TFU, and septic tank with MBR)
- Determine the effects of higher effluent quality on soil clogging and biozone development during effluent infiltration and percolation in soil
- Determine the treatment efficiency for selected chemicals and pathogens achieved by tank-based treatment units and soil treatment unit operations

The research was completed through controlled field experimentation at the Mines Park Test Site on the CSM campus. The experimentation was designed to build on previous and ongoing research concerning the hydraulic and purification performance of OWSs. The project approach was conceived to provide the requisite experimental control at the pilot-scale, while representing full-scale operations, to enhance the understanding of design and performance relationships related to engineered treatment units and OWSs. The conclusions derived from the research are:

- During this project, a major field experiment was established and operations were initiated, yielding an array of treatment unit operation and performance data over a period of six months (April to October 2004). This research duration has provided valuable insight concerning the startup and early operation and performance of an OWS. A longer period of monitoring and assessment is needed to develop longer-term data and provide greater insight relevant to full-scale system operation.
- The effluents generated by the septic tank, TFU, and MBR, after a period of start-up operations, were generally consistent in quality for each unit, but constituent concentrations varied among treatment units.

- The three treatment units achieved different treatment efficiencies for organic matter, solids, nutrients, and bacteria. The relative efficiency ranking was: STE << TFU << MBR. The relative ranking for operational complexity, operation and maintenance requirements, energy use, and cost followed a similar pattern: STE << TFU << MBR. Due to the short duration of the performance evaluation completed, it is difficult to estimate the service life of the OWSs employing TFUs or MBRs or their long-term operation and maintenance requirements and life-cycle costs.
- Addition of an engineered treatment unit to produce an effluent of a higher quality than typical STE can retard soil clogging and biozone development. Engineered treatment units can also enable application of higher daily loading rates to soil and concomitantly smaller soil treatment units (assuming purification is reliably achieved over the service life of the system). However, the magnitude of the increase in the higher-quality effluent application rates is likely limited by the hydraulic properties of the natural soil. It may be reasonable to limit the daily design HLR for a given soil treatment unit, to a small percentage of the soil's K_{sat} (for example, design daily HLR = 3 to 5% of the K_{sat}).
- The treatment train purification for DOC and total nitrogen follows a trend of higher performance that is given by: septic tank-MBR-soil treatment > septic tank-TFU-soil treatment > septic tank-soil treatment. The treatment trains including a TFU or MBR generally perform better with respect to purification and are less affected by HLR than the treatment train based on only a septic tank and soil treatment. In addition, their performance is relatively better than soil treatment alone with 60 cm of soil. However, increasing the vadose zone soil depth (for example, from 60 to 120 cm) tends to shrink the differences in performance between the three treatment trains.
- The ability of an Ascalon sandy loam soil to remove viruses was quite high and insensitive to whether the natural soil had received STE, TFU effluent, or MBR effluent at either 2 or 8 cm/d design HLRs. The results of bromide tracer tests, infiltration rate measurements, and modeling reveal that some degree of soil clogging and biozone formation is occurring in the soil, even with higher quality effluents applied. As a result, viruses are effectively removed (removal in soil of about 6-logs). These results refute concerns that virus removal in soils receiving high-quality effluents might be diminished due to the absence of a classic biozone resulting from the low levels of tBOD and TSS applied.
- Coring of companion study test cells occurred and is presented here to serve as a method check for planned future coring of the NDWRCDP study test cells (target for 18 to 24 months of operation) as well as a comparison to this study. The coring results from the 8 cm/d STE loaded test cells showed increased water content, ammonium, total phosphorus, fecal coliform, *E. Coli* bacteria, and total viable biomass. This increase occurred in the 30 cm of soil closest to the infiltrative surface with the greatest increases in the top 0 to 10 cm. Future characterization of the NDWRCDP test cells will allow for a better understanding of how the infiltrative surface of more mature systems dosed with different quality effluents compare.

5.2 Recommendations

Based on the research completed to date, the following recommendations are made:

- Continued operation and monitoring of the tank-based treatment units and soil treatment component should proceed as long as possible to yield longer-term performance data.
- Completion of an 18- to 24-month coring event and a terminal characterization event of the soil test cells is needed to investigate the biozone morphology, biogeochemistry, and vadose zone properties with depth.
- A surrogate/tracer test should be completed to evaluate the removal of emerging organic chemicals and virus in the tank-based unit operations (septic tank, TFU, MBR) and the soil treatment units to demonstrate the treatment train purification efficiencies for these wastewater constituents.
- Analytical and numerical modeling of the unit operations, including the TFU, MBR, and soil treatment units, should be completed to enhance the ability to describe and predict system performance as a function of design and environmental conditions.
- Additional experimentation with other types of engineered treatment units and other environmental settings should be carried out to corroborate and extend the findings of the research completed at the Mines Park Test Site in Golden, Colorado.

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7 ACRONYMS AND ABBREVIATIONS

7.1 Acronyms

BOD	Biochemical oxygen demand
BOD_5	Biochemical oxygen demand, 5-day test
cBOD	Carbonaceous biochemical oxygen demand
CEC	Cation exchange capacity
CFD	Cumulative frequency distribution
COD	Chemical oxygen demand
CSM	Colorado School of Mines
DOC	Dissolved organic carbon
FC	Fecal coliform bacteria
GWR	Ground Water Rule
HPC	Heterotrophic plate count
HPI	Hydrophilic fraction of bulk organic carbon
HPO-A	Hydrophobic acids fraction of bulk organic carbon
HPO-N	Hydrophobic neutral fraction of bulk organic carbon
HLR	Hydraulic loading rate
ID	Inner diameter
ISA	Infiltrative surface architecture
ISU	Infiltrative surface utilized

K_{sat}	Saturated hydraulic conductivity
MBR	Membrane bioreactor
MLSS	Mixed liquor suspended solids
MS-2	Coliphage used as a surrogate for human enteric virus
NDWRCDP	National Decentralized Water Resources Capacity Development Project
nBOD	Nitrogenous biochemical oxygen demand
OWS	Onsite wastewater system
PETG	Polyethylenterephthalate
PMB	Porous media biofilter
PRD-1	Bacteriophage with *Salmonella* host used as a surrogate for human enteric virus
PBS	Phosphate buffered saline
QA	Quality assurance
QC	Quality control
RPD	Relative percent difference
RSV	Re-circulating splitter valve
RWT	Rhodamine-WT
SEC	Size exclusion chromatography
STE	Septic tank effluent
SUVA	Specific ultraviolet absorbance
tBOD	Total biochemical oxygen demand (cBOD + nBOD)
TKN	Total Kjeldahl nitrogen
TFU	Textile filter unit
TOC	Total organic carbon

TS	Total solids
TSS	Total suspended solids
USDA	United States Department of Agriculture
US EPA	United States Environmental Protection Agency
UV	Ultraviolet
UVA	Ultraviolet absorbance

7.2 Abbreviations and Units

bgs	below ground surface
C	Carbon (or concentration at time *t* in Equation 3.4)
°C	Celsius
C_o	initial concentration
C_j	concentration of virus in column outflow
Ca	calcium
$CaCO_3$	calcium carbonate
cfu	colony forming unit
cm	centimeters
CV	coefficient of variation
d	day
in.	inch
°F	Fahrenheit
ft	feet
g	gram
gal	gallon

gpd	gallons per day
h	hour
HCl	hydrochloric acid
hp	horsepower
IR_o	initial infiltration rate (baseline infiltration rate)
IR_t	infiltration rate at time *t*
K	potassium
k_i	inactivation rate for viruses
KBr	Potassium bromide
kg	kilogram
L	liter (or length in Equation 3.2)
m	meter
meq	milliequivalent
Mg	magnesium
mg	milligram
mL	milliliter
mS	millisiemens
N	nitrogen
n	number of samples
Ne	effective porosity
NH_4	ammonium
NO_3	nitrate
ng	nanogram

nm	nanometer
P	phosphorus
pfu	plaque forming unit
PO_4	phosphate
ppb	parts per billion
ppm	parts per million
psi	pounds per square inch
q	hydraulic loading rate
s	second
SD	standard deviation
t	time (or travel time in Equation 3.2)
µg	micrograms
µm	micron
V_j	volume of column outflow
V_t	total volume of dose

A SITE CHARACTERISTICS AND SOIL CONDITIONS

A.1 Mines Park Test Site Photographs

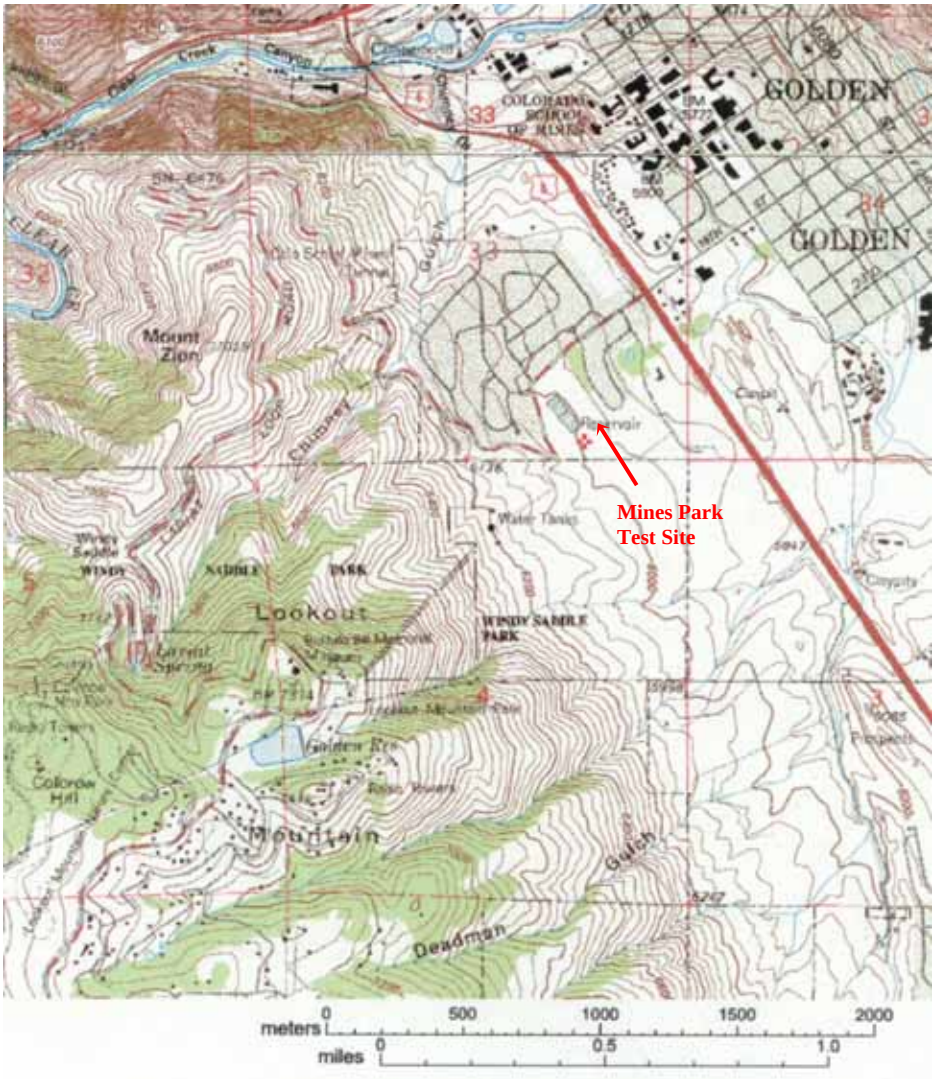


Figure 7-1
Topographic Map of the Area Surrounding the Mines Park Test Site

Figure 7-2
Photograph of Backhoe Test Pit 1, Looking Southwest

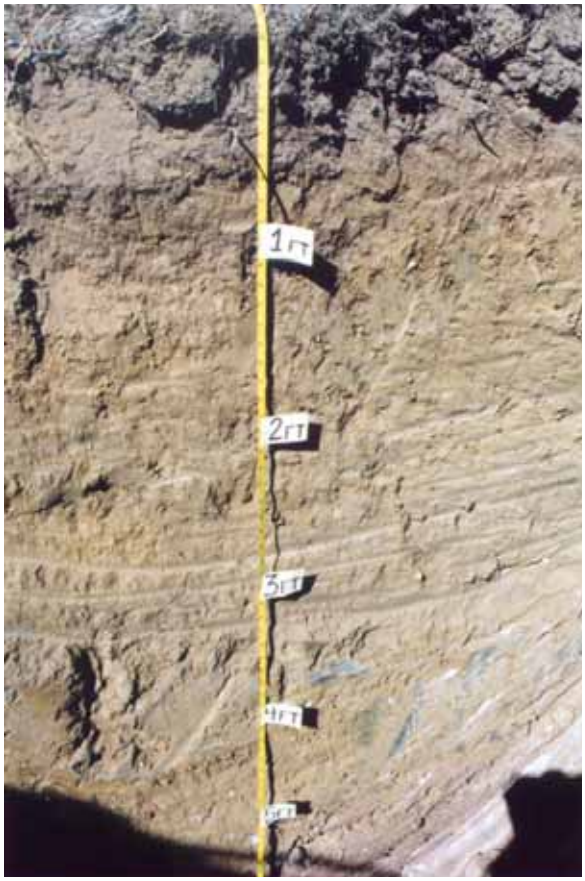


Figure 7-3
Photograph of Soil Profile at Backhoe Test Pit 1 (West Side of Pit)

Figure 7-4
Photograph of Backhoe Test Pit #2, Looking Northeast

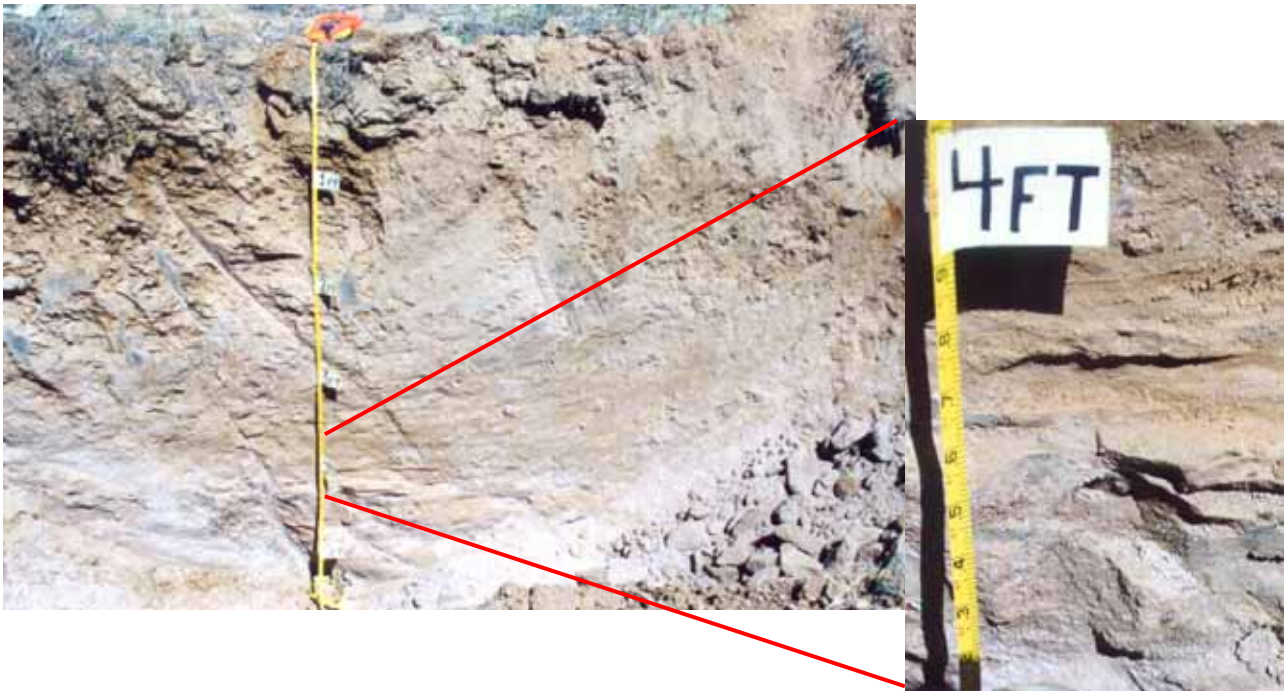


Figure 7-5
Photograph of Soil Profile at Backhoe Test Pit 2 (North Side of Pit) With Close-Up of Transition Zone

A.2 Summarized Soil Profile Descriptions

Table A-1
Backhoe Test Pit 1, Soil Profile Description

Colorado School of Mines
Environmental Science and Engineering Division
1500 Illinois Street
Golden, Colorado 80401-1887
Phone: 303-274-3427 Fax: 303-273-3413

Project: WRTS Site Evaluation — Ground Surface Elevation:
Site/Location: Backhoe Test Pit #1 (BTP1) — Described by: Kathryn Lowe
Date: 09-May-02 — Page 1 of 1

Depth (bgs)	Soil Texture	Structure / Consistency	Color	Soil Hydrologic Parameters	Remarks
1	Sandy clay loam	Unconsolidated, tight, limited macropores	10YR3/3	dry	No mottling observed in pit
2	Sandy clay loam	Unconsolidated, tight, limited macropores	7YR4/4	dry	
3	Sandy clay loam	Unconsolidated, tight, limited macropores	7YR4/4	dry	
4	Sandy clay loam	Unconsolidated, tight, limited macropores	7YR4/4	dry	
5	Sandy clay loam	Unconsolidated, tight, limited macropores	7YR4/4	dry	
5.5	Not applicable	Poorly consolidated, very friable	2.5YR5/4	dry	Clear interface between soils; weathered igneous rock (conglomerate)
6	Not applicable	Poorly consolidated, very friable	2.5YR5/4	dry	weathered igneous rock (conglomerate)
7	Not applicable	Poorly consolidated, very friable	2.5YR5/4	dry	weathered igneous rock (conglomerate)
8	Not applicable	Poorly consolidated, very friable	2.5YR5/4	dry	weathered igneous rock (conglomerate)

Depth in feet below ground surface (bgs)

Table A-2
Backhoe Test Pit 2, Soil Profile Description

Colorado School of Mines
Environmental Science and Engineering Division
1500 Illinois Street
Golden, Colorado 80401-1887
Phone: 303-274-3427 Fax: 303-273-3413

Project: WRTS Site Evaluation — Ground Surface Elevation:
Site/Location: Backhoe Test Pit #2 (BTP2) — Described by: Kathryn Lowe
Date: 09-May-02 — Page 1 of 1

Depth (bgs)	Soil Texture	Structure / Consistency	Color	Soil Hydrologic Parameters	Remarks
0.5	Sandy clay loam	Unconsolidated, tight, limited macropores	10YR5/6	dry	No mottling observed in pit
1	Sandy loam	Highly fractured with bedding planes ~0.5 to 5.5 inches thick	5YR6/4	dry	weathered siltstone
2	Sandy loam	Highly fractured with bedding planes ~0.5 to 5.5 inches thick	5YR6/4	dry	mottling along root zones in top 2ft of pit; weathered siltstone
3	Sandy loam	Highly fractured with bedding planes ~0.5 to 5.5 inches thick	5YR6/4	dry	weathered siltstone
4	Sandy loam	Highly fractured with bedding planes ~0.5 to 5.5 inches thick	5YR6/4	dry	weathered siltstone
4.5	Sandy loam	Fractured with bedding planes ~2 to 5 inches thick	7.5YR5/6	dry	Clear interface between soils, but not contiguous across pit; weathered siltstone / weathered igneous rock (conglomerate)
5	Not applicable	Fractured with bedding planes ~2 to 5 inches thick	7.5YR5/6	dry	weathered siltstone / weathered igneous rock (conglomerate)
6	Not applicable	Fractured with bedding planes ~2 to 5 inches thick		dry	weathered igneous bedrock (poorly sorted, subangular, <3% schist, 40% silica, 55% feldspar (Kspar), and <2% hornblend)
					Total depth of test pit, 6 ft bgs - bedrock

Table A-3
Borehole 1, Soil Profile Description

Colorado School of Mines
Environmental Science and Engineering Division
1500 Illinois Street
Golden, Colorado 80401-1887
Phone: 303-274-3427 Fax: 303-273-3413

Project: WRTS Site Evaluation
Ground Surface Elevation:
Site/Location: Borehole 01 (BH01)
Described by: Kathryn Lowe
Date: 16-Apr-02
Page 1 of 1

Depth (bgs)	Sample Type	Color	Soil Hydrologic Parameters	Soil Texture	Structure / Consistency / Morphology
2	core	10YR5/6	dry	very fine-medium gravelly sandy clay loam	loose, subangular
4	core	10YR6/3	dry	fine-medium gravelly sandy loam	loose, subangular
6	core	10YR7/4	dry	very fine-medium gravelly sandy loam	friable, poorly sorted, subangular; weathered igneous
8	core	10YR7/4	dry	very fine-medium gravelly sandy loam	friable, poorly sorted, subangular; weathered igneous
10	cuttings	5YR4/4	dry	sandy loam	mottled, poorly sorted, subangular
12	core	5YR4/4 & 5GY7/	dry	sandy loam	mottled, poorly sorted, subangular
14	cuttings	2.5YR3/4	dry	sandy loam	poorly sorted, subangular
16	cuttings	2.5YR4/4	wet	sandy loam	poorly sorted, subangular

Table A-4
Borehole 2, Soil Profile Description

Colorado School of Mines
Environmental Science and Engineering Division
1500 Illinois Street
Golden, Colorado 80401-1887
Phone: 303-274-3427 Fax: 303-273-3413

Project: WRTS Site Evaluation
Ground Surface Elevation:
Site/Location: Borehole 02 (BH02)
Described by: Kathryn Lowe
Date: 16-Apr-02
Page 1 of 1

Depth (bgs)	Sample Type	Color	Soil Hydrologic Parameters	Soil Texture	Structure / Consistency / Morphology
2	core	5YR4/4	dry	fine gravelly sandy clay loam	loose
4	core	5YR4/4	dry	fine gravelly sandy loam	loose, poorly sorted, subangular
6	core	5YR4/4 & 10Y7/	dry	fine gravelly sandy loam	mottled (predominately grey with some red), friable, poorly sorted, subangular; weathered igneous
8	core	5YR4/3	dry	sandy loam	loose, poorly sorted, subangular
10	cuttings	5YR4/3	dry	sandy loam	poorly sorted, subangular
12	cuttings	7.5YR7/2	dry	sandy loam	poorly sorted, subrounded
14	cuttings	7.5YR7/2	dry	sandy loam	poorly sorted, subrounded
16	cuttings	2.5YR6/3	dry	sandy loam	rounded to subrounded
18	cuttings	5YR5/4	dry	sandy loam	subangular

Table A-5
Borehole 3, Soil Profile Description

Colorado School of Mines
Environmental Science and Engineering Division
1500 Illinois Street
Golden, Colorado 80401-1887
Phone: 303-274-3427 Fax: 303-273-3413

Project: WRTS Site Evaluation
Ground Surface Elevation: _____
Site/Location: Borehole 03 (BH03)
Described by: Kathryn Lowe / Brett Chambers
Date: 16-Apr-02
Page 1 of 1

Depth (bgs)	Sample Type	Color	Soil Hydrologic Parameters	Soil Texture	Structure / Consistency / Morphology
2	core	7.5YR4/6	dry	sandy clay loam	loose
4	core	10YR5/4	dry	sandy loam	loose
6	core	10YR5/4	dry	fine gravelly sandy loam	tight, well sorted, subangular
8	core	7.5YR4/4	dry	medium gravelly sandy loam	friable, poorly sorted, subangular (clay layer at 7.75 ft - 7.5YR4/6); weathered igneous
10	core	2.5YR4/4	moist	fine gravelly sandy loam	friable, poorly sorted, subangular; weathered igneous
12	core	10YR4/4	dry	medium gravelly sandy loam	loose, poorly sorted, subangular
15	cuttings	5YR4/4	dry	sandy loam	poorly sorted, subangular
16	cuttings	5YR5/4	dry	sandy loam	poorly sorted, subangular
18	cuttings	2.5YR4/6	dry	sandy loam	poorly sorted, subangular
20	cuttings	5YR5/4	dry	sandy loam	poorly sorted, subangular
22	cuttings	2.5YR5/6	dry	sandy loam	poorly sorted, subangular

Table A-6
Borehole 4, Soil Profile Description

Colorado School of Mines
Environmental Science and Engineering Division
1500 Illinois Street
Golden, Colorado 80401-1887
Phone: 303-274-3427 Fax: 303-273-3413

Project: WRTS Site Evaluation
Ground Surface Elevation: _____
Site/Location: Borehole 04 (BH04)
Described by: Kathryn Lowe / Brett Chambers
Date: 17-Apr-02
Page 1 of 1

Depth (bgs)	Sample Type	Color	Soil Hydrologic Parameters	Soil Texture	Structure / Consistency / Morphology
2	core	10YR4/6	dry	fine gravelly sandy clay loam	tight
4	core	10YR5/6	dry	sandy clay loam	
6	core	10YR5/6	dry	fine gravelly sandy clay loam	
8	core	10YR5/8	dry	sandy loam	loose, poorly sorted, subangular
10	core	10YR4/6	moist	sandy loam	loose, poorly sorted, subangular
11.5	core	7.5YR5/6	moist	sandy loam	loose, poorly sorted, subangular
14	core	7.5YR4/6	moist	sandy loam	loose, poorly sorted, subangular
16	core	5YR4/4	dry	fine gravelly sandy loam	friable, poorly sorted, subangular; weathered igneous
18	cuttings	7.5YR4/4	dry	fine gravelly sandy loam	poorly sorted, subangular
20	cuttings	5YR4/4	dry	sandy loam	poorly sorted, subangular
22	cuttings	5YR4/4	dry	sandy loam	poorly sorted, subangular

Table A-7
Borehole 5, Soil Profile Description

Colorado School of Mines
Environmental Science and Engineering Division
1500 Illinois Street
Golden, Colorado 80401-1887
Phone: 303-274-3427 Fax: 303-273-3413

Project: WRTS Site Evaluation
Ground Surface Elevation:
Site/Location: Borehole 05 (BH05)
Described by: Kathryn Lowe / Brett Chambers
Date: 17-Apr-02
Page 1 of 1

Depth (bgs)	Sample Type	Color	Soil Hydrologic Parameters	Soil Texture	Structure / Consistency / Morphology
2	core	5YR5/6	dry	fine-medium gravelly sandy clay loam	poorly sorted
4	core	7.5YR5/6	dry	sandy clay loam	poorly sorted
6	core	7.5YR6/6	dry	sandy loam	loose, poorly sorted, subangular
7	core	2.5YR5/4	dry	sandy loam	loose, poorly sorted, subangular
8	cuttings	2.5YR6/4	dry	sandy loam	poorly sorted, subangular
10	cuttings	7.5YR7/3	dry	sandy loam	poorly sorted, subangular

Table A-8
Borehole 6, Soil Profile Description

Colorado School of Mines
Environmental Science and Engineering Division
1500 Illinois Street
Golden, Colorado 80401-1887
Phone: 303-274-3427 Fax: 303-273-3413

Project: WRTS Site Evaluation
Ground Surface Elevation:
Site/Location: Borehole 06 (BH06)
Described by: Kathryn Lowe / Brett Chambers
Date: 17-Apr-02
Page 1 of 1

Depth (bgs)	Sample Type	Color	Soil Hydrologic Parameters	Soil Texture	Structure / Consistency / Morphology
2	core	10YR5/6	dry	medium gravelly sandy loam	poorly sorted, subangular
4	cuttings	10YR6/4	dry	fine gravelly sandy loam	poorly sorted, subangular
6	cuttings	10YR6/6	dry	fine gravelly sandy loam	poorly sorted, subangular
8	cuttings	10YR7/4	dry	fine gravelly sandy loam	poorly sorted, subrounded
10	cuttings	7.5YR6/4	dry	sandy loam	subrounded

Table A-9
Borehole 8, Soil Profile Description

Colorado School of Mines
Environmental Science and Engineering Division
1500 Illinois Street
Golden, Colorado 80401-1887
Phone: 303-274-3427 Fax: 303-273-3413

Project: WRTS Site Evaluation — Ground Surface Elevation: _____
Site/Location: Borehole 08 (BH08) — Described by: Kathryn Lowe / Brett Chambers
Date: 18-Apr-02 — Page 1 of 1

Depth (bgs)	Sample Type	Color	Soil Hydrologic Parameters	Soil Texture	Structure / Consistency / Morphology
2	core	5YR5/6	dry	fine-medium gravelly sandy clay loam	loose; some weathered igneous
4	core	5YR5/6	dry	fine-medium gravelly sandy loam	loose; some weathered igneous
6	core	5YR4/4	dry	very medium gravelly sandy clay loam	poorly sorted, subangular
7.5	core	7.5YR6/4	dry	fine gravelly sandy loam	mottled, poorly sorted, subangular
9	cuttings	5YR7/3	dry	sandy loam	well sorted, subangular
10	cuttings	5YR6/3	dry	sandy loam	well sorted, subangular
12	cuttings	2.5YR6/3	dry	sandy loam	subrounded
14	cuttings	5YR5/4	dry	sandy loam	poorly sorted, subrounded
16	cuttings	7.5YR5/4	dry	sandy loam	subrounded
18	cuttings	5YR5/4	dry	sandy loam	subrounded
20	cuttings	5YR6/3	dry	sandy loam	well sorted, subrounded

Table A-10
Borehole 9, Soil Profile Description

Colorado School of Mines
Environmental Science and Engineering Division
1500 Illinois Street
Golden, Colorado 80401-1887
Phone: 303-274-3427 Fax: 303-273-3413

Project: WRTS Site Evaluation — Ground Surface Elevation: _____
Site/Location: Borehole 09 (BH09) — Described by: Kathryn Lowe
Date: 18-Apr-02 — Page 1 of 1

Depth (bgs)	Sample Type	Color	Soil Hydrologic Parameters	Soil Texture	Structure / Consistency / Morphology
2	core	5YR4/3	dry	sandy clay loam	loose
4	core	7.5YR5/6	dry	sandy loam	loose
6	cuttings	7.5YR5/4	dry	sandy loam	poorly sorted, subangular
8	cuttings	5YR4/3	dry	sandy loam	poorly sorted, subrounded
10	cuttings	5YR3/3	dry	loam	poorly sorted, subangular
12	cuttings	7.5YR5/4	dry	sandy loam	poorly sorted, subrounded
14	cuttings	5YR5/4	dry	sandy loam	poorly sorted, subrounded
16	cuttings	5YR4/4	dry	sandy loam	poorly sorted, subrounded
18	cuttings	5YR4/4	dry	sandy loam	poorly sorted, subrounded
20	cuttings	5YR4/4	dry	sandy loam	poorly sorted, subrounded

A.3 Soil Analyses Data

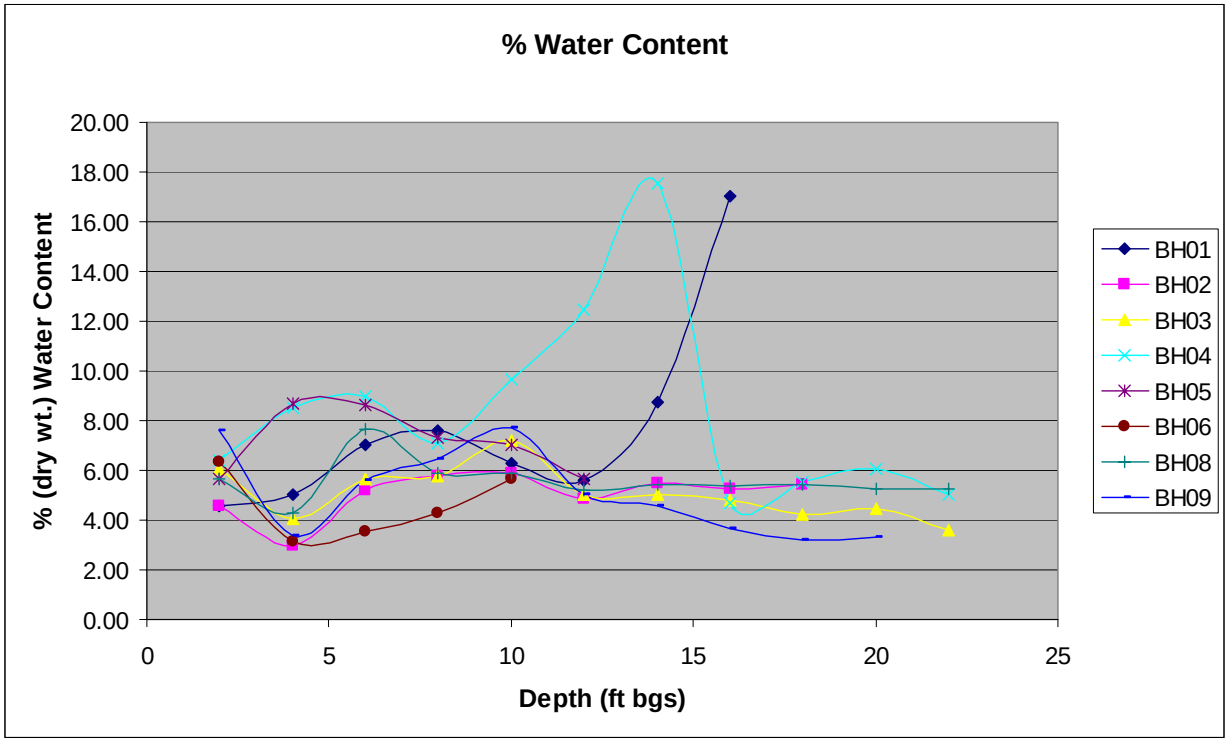


Figure 7-6
Soil Water Content With Depth

Table A-11
Summary of Water Content Results

Depth (ft bgs)	% (Dry wt.) Water Content							
	BH01	BH02	BH03	BH04	BH05	BH06	BH08	BH09
2	4.6	4.6	6.0	6.4	5.6	6.3	5.7	7.6
4	5.0	3.0	4.0	8.5	8.7	3.1	4.3	3.4
6	7.0	5.2	5.7	9.0	8.6	3.5	7.6	5.6
8	7.6	5.8	5.8	7.1	7.0	4.3	5.9	6.4
10	6.3	5.9	7.2	9.7	5.6	5.7	5.2	7.7
12	5.6	4.9	5.0	12.5	–	–	5.4	5.0
14	8.8	5.5	–	17.5	–	–	5.4	4.6
16	17.0	5.3	4.8	4.7	–	–	5.4	3.7
18	–	5.4	4.2	5.6	–	–	5.3	3.2
20	–	–	4.4	6.1	–	–	5.3	3.3
22	–	–	3.6	5.0	–	–	–	–

No samples collected from BH07, which was located within 3 ft of BH05
Depth in feet below ground surface (ft bgs)
– indicates no sample collected at that interval

Table A-12
Summary of Organic Matter Results

Depth (ft bgs)	% Organic Matter							
	BH01	BH02	BH03	BH04	BH05	BH06	BH08	BH09
2	0.7	1.1	1.4	1.0	0.8	1.3	0.8	1.2
4	0.3	0.5	0.6	0.8	0.6	0.6	0.3	0.3
6	0.3	0.4	0.4	0.6	0.5	0.5	0.5	0.2
8	0.3	0.3	0.4	0.5	0.5	0.3	0.2	0.2
10	0.3	0.2	0.1	0.7	0.1	0.8	0.1	0.4

Samples collected below 10 ft were not submitted for analysis
No samples collected from BH07, which was located within 3 ft of BH05

Table A-13
Summary of Cation Exchange Capacity Results

Depth (ft bgs)	Cation Exchange Capacity (meq / 100 gram dry soil)							
	BH01	BH02	BH03	BH04	BH05	BH06	BH08	BH09
2	8.6	8.9	12.1	11.3	11.2	5.3	7.7	1.8
4	5.7	6.9	8.8	13.1	22.1	4.0	6.6	3.9
6	4.9	6.6	9.1	13.6	8.2	4.3	21.6	3.5
8	3.6	2.5	6.4	15.0	5.7	2.7	5.2	6.3
10	13.4	4.1	3.4	10.8	2.9	6.6	12.8	13.4

Samples collected below 10 ft were not submitted for analysis
No samples collected from BH07, which was located within 3 ft of BH05
meq = milliequivalent

Table A-14
Summary of Percent Sand/Silt/Clay Results

Depth (ft bgs)	% Sand / % Silt / % Clay (dry wt.)							
	BH01	BH02	BH03	BH04	BH05	BH06	BH08	BH09
2	64/ 14/ 22	70/ 9/ 21	48/ 23/ 29	59/ 17/ 24	63/ 8/ 29	69/ 12/ 19	69/ 9/ 22	58/ 13/ 29
4	72/ 12/ 16	75/ 7/ 18	68/ 16/ 16	60/ 12/ 28	62/ 14/ 24	76/ 10/ 14	69/ 12/ 19	77/ 10/ 13
6	73/ 13/ 14	66/ 18/ 16	70/ 13/ 17	61/ 15/ 24	69/ 13/ 18	75/ 11/ 14	59/ 17/ 24	74/ 15/ 11
8	75/ 14/ 11	77/ 11/ 12	75/ 10/ 15	69/ 13/ 18	69/ 14/ 17	74/ 13/ 13	74/ 13/ 13	66/ 21/ 13
10	72/ 14/ 14	70/ 15/ 15	68/ 17/ 15	63/ 17/ 20	58/ 26/ 16	73/ 11/ 16	73/ 14/ 13	45/ 38/ 17

Samples collected below 10 ft were not submitted for analysis
No samples collected from BH07, which was located within 3 ft of BH05

Table A-15
Summary of Percent Sand Fraction Based on Dry Sieve Analysis

	Count	Min.	Max.	Ave.	Median	Std. Dev.	CV	Var.
% > 2 mm (coarse sand)	17	8.7%	51.7%	23.9%	22.8%	0.097	0.404	0.0093
% Sand	17	46.3%	84.9%	73.3%	74.6%	0.089	0.122	0.0080
% Silt & Clay	17	1.3%	9.2%	2.8%	2.3%	0.018	0.639	0.0003

Soil sample intervals included in summary statistics are: BH01, 6 and 8 ft below ground surface (bgs); BH02, BH03, and BH04, 2, 4, 6, and 8 ft bgs; and BH05, BH08, and BH09, 4 ft bgs

Table A-16
Summary Statistics for Sand Grain Size Distribution Based on Dry Sieve Analysis

	Count	Minimum	Maximum	Average	Median	Standard Deviation	CV	Var.
d10	17	0.075	0.21	0.16	0.17	0.039	0.252	0.002
d25	17	0.160	0.45	0.30	0.28	0.082	0.278	0.007
d30	17	0.180	0.68	0.37	0.32	0.128	0.351	0.016
d50	17	0.370	1.10	0.70	0.68	0.215	0.309	0.046
d60	17	0.500	1.20	0.88	0.90	0.214	0.244	0.046
d75	17	0.810	1.60	1.21	1.10	0.227	0.188	0.051

Soil sample intervals included in summary statistics are: BH01, 6 and 8 ft below ground surface (bgs); BH02, BH03, and BH04, 2, 4, 6, and 8 ft bgs; and BH05, BH08, and BH09, 4 ft bgs

Table A-17
Sand Grain Size Distribution by Dry Sieve Analysis, BH03 at 2 ft bgs

Colorado School of Mines
Environmental Science and Engineering Division
1500 Illinois Street
Golden, Colorado 80401-1887
Phone: 303-274-3427 Fax: 303-273-3413

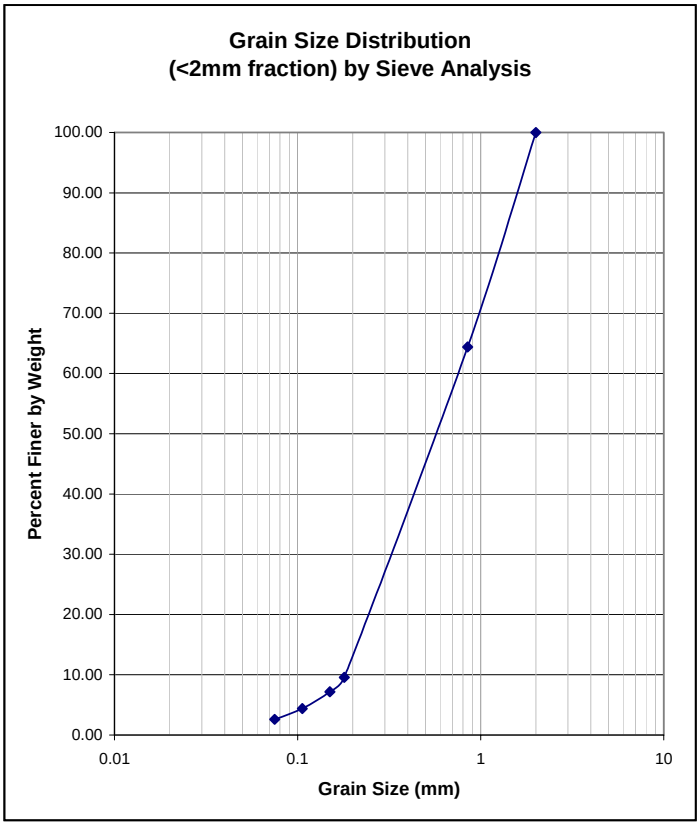
Project: WRTS Site Evaluation
Location: BH03, 2ft bgs
Date: 14-May-02
Analyses by: Brett Chambers

USDA Classification

Size (mm)	Fraction	Mass (g)	Percent
>2.0	Gravel		
2.0-1.0	Very Coarse Sand		
1.0-0.5	Coarse Sand		
0.5-0.25	Medium Sand		
0.25-0.1	Fine Sand		
0.1-0.05	Very Fine Sand		
0.05-0.002	Silt		
<0.002	Clay		
	Total		
> 2mm	Coarse sand		
2-0.05	Sand		
<0.05	Silt + Clay		

Unified Classification

Sieve No.	Size (mm)	Fraction	Mass (g)	Percent
#10	>2.0	Coarse Sand	15.03	12.8%
#20	2.0-0.85	Medium Sand	36.36	31.0%
#80	0.85-0.18	Fine Sand	56.08	47.8%
#100	0.18-0.15	Fine Sand	2.42	2.1%
#140	0.15-0.106	Fine Sand	2.87	2.4%
#200	0.106-0.075	Fine Sand	1.82	1.6%
-	< 0.075	Silt + Clay	2.66	2.3%
-	-	-	-	-
		Total	117.24	100.0%
	> 2mm	Coarse sand	15.03	12.8%
	2 - 0.075	Sand	99.55	84.9%
	<0.075	Silt + Clay	2.66	2.3%



Summary Statistics

d10	0.18
d25	0.28
d30	0.32
d50	0.58
d60	0.74
d75	1.10

Table A-18
Sand Grain Size Distribution by Dry Sieve Analysis, BH03 at 4 ft bgs

Colorado School of Mines
Environmental Science and Engineering Division
1500 Illinois Street
Golden, Colorado 80401-1887
Phone: 303-274-3427 Fax: 303-273-3413

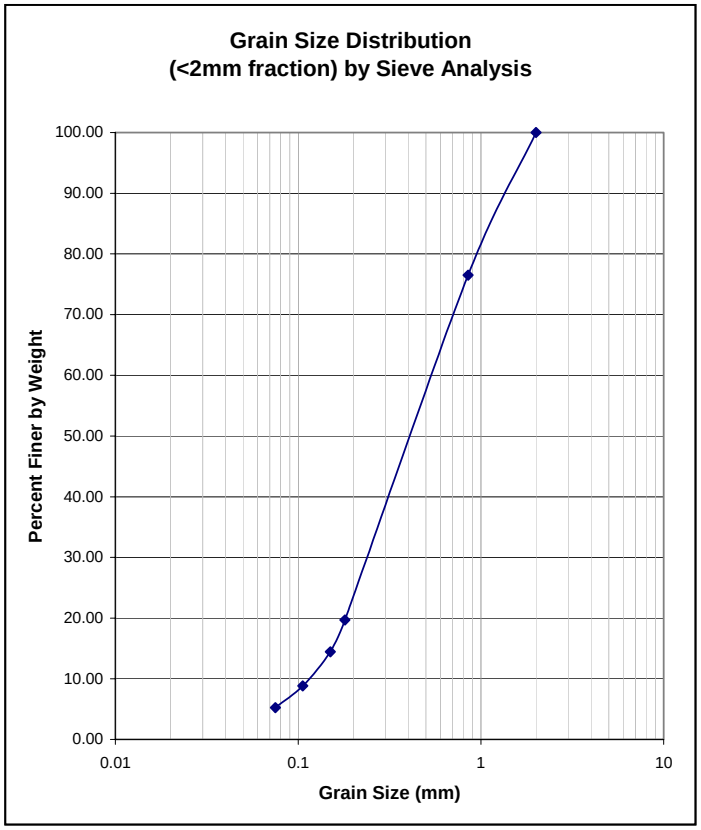
Project: WRTS Site Evaluation
Location: BH03, 4ft bgs
Date: 14-May-02
Analyses by: Brett Chambers

USDA Classification

Size (mm)	Fraction	Mass (g)	Percent
>2.0	Gravel		
2.0-1.0	Very Coarse Sand		
1.0-0.5	Coarse Sand		
0.5-0.25	Medium Sand		
0.25-0.1	Fine Sand		
0.1-0.05	Very Fine Sand		
0.05-0.002	Silt		
<0.002	Clay		
	Total		
> 2mm	Coarse sand		
2-0.05	Sand		
<0.05	Silt + Clay		

Unified Classification

Sieve No.	Size (mm)	Fraction	Mass (g)	Percent
#10	>2.0	Coarse Sand	20.29	15.4%
#20	2.0-0.85	Medium Sand	26.17	19.9%
#80	0.85-0.18	Fine Sand	63.22	48.0%
#100	0.18-0.15	Fine Sand	5.87	4.5%
#140	0.15-0.106	Fine Sand	6.23	4.7%
#200	0.106-0.075	Fine Sand	3.99	3.0%
-	< 0.075	Silt + Clay	5.85	4.4%
-	-	-	-	-
		Total	131.62	100.0%
	> 2mm	Coarse sand	20.29	15.4%
	2 - 0.075	Sand	105.48	80.1%
	<0.075	Silt + Clay	5.85	4.4%



Summary Statistics

d10	0.11
d25	0.20
d30	0.23
d50	0.40
d60	0.52
d75	0.81

Table A-19
Sand Grain Size Distribution by Dry Sieve Analysis, BH03 at 6 ft bgs

Colorado School of Mines
Environmental Science and Engineering Division
1500 Illinois Street
Golden, Colorado 80401-1887
Phone: 303-274-3427 Fax: 303-273-3413

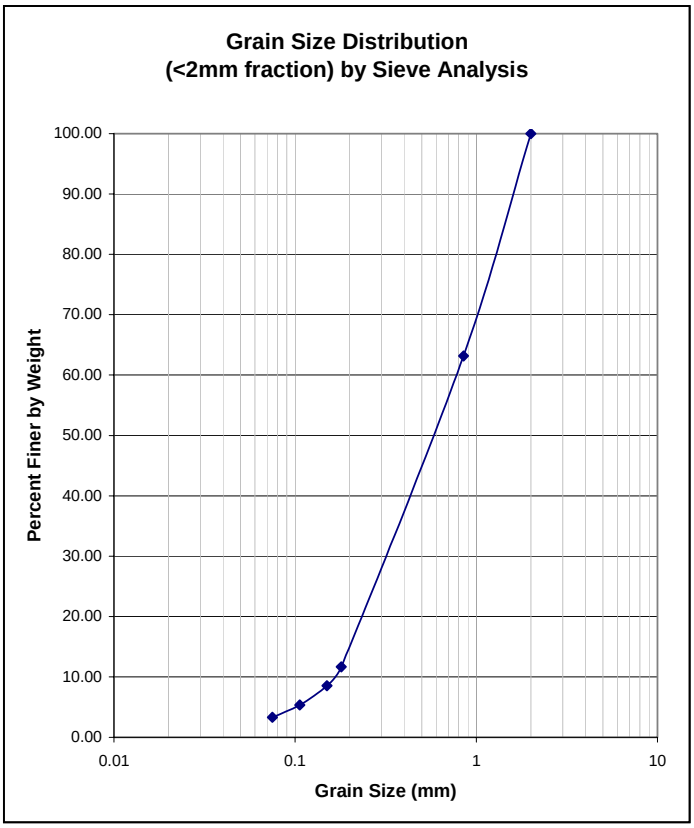
Project: WRTS Site Evaluation
Location: BH03, 6ft bgs
Date: 14-May-02
Analyses by: Brett Chambers

USDA Classification

Size (mm)	Fraction	Mass (g)	Percent
>2.0	Gravel		
2.0-1.0	Very Coarse Sand		
1.0-0.5	Coarse Sand		
0.5-0.25	Medium Sand		
0.25-0.1	Fine Sand		
0.1-0.05	Very Fine Sand		
0.05-0.002	Silt		
<0.002	Clay		
	Total		
> 2mm	Coarse sand		
2-0.05	Sand		
<0.05	Silt + Clay		

Unified Classification

Sieve No.	Size (mm)	Fraction	Mass (g)	Percent
#10	>2.0	Coarse Sand	29.97	22.8%
#20	2.0-0.85	Medium Sand	37.27	28.4%
#80	0.85-0.18	Fine Sand	52.11	39.7%
#100	0.18-0.15	Fine Sand	3.17	2.4%
#140	0.15-0.106	Fine Sand	3.23	2.5%
#200	0.106-0.075	Fine Sand	2.07	1.6%
-	< 0.075	Silt + Clay	3.35	2.6%
-	-	-	-	-
		Total	131.17	100.0%
	> 2mm	Coarse sand	29.97	22.8%
	2 - 0.075	Sand	97.85	74.6%
	<0.075	Silt + Clay	3.35	2.6%



Summary Statistics

d10	0.17
d25	0.27
d30	0.31
d50	0.58
d60	0.77
d75	1.10

Table A-20
Sand Grain Size Distribution by Dry Sieve Analysis, BH03 at 8 ft bgs

Colorado School of Mines
Environmental Science and Engineering Division
1500 Illinois Street
Golden, Colorado 80401-1887
Phone: 303-274-3427 Fax: 303-273-3413

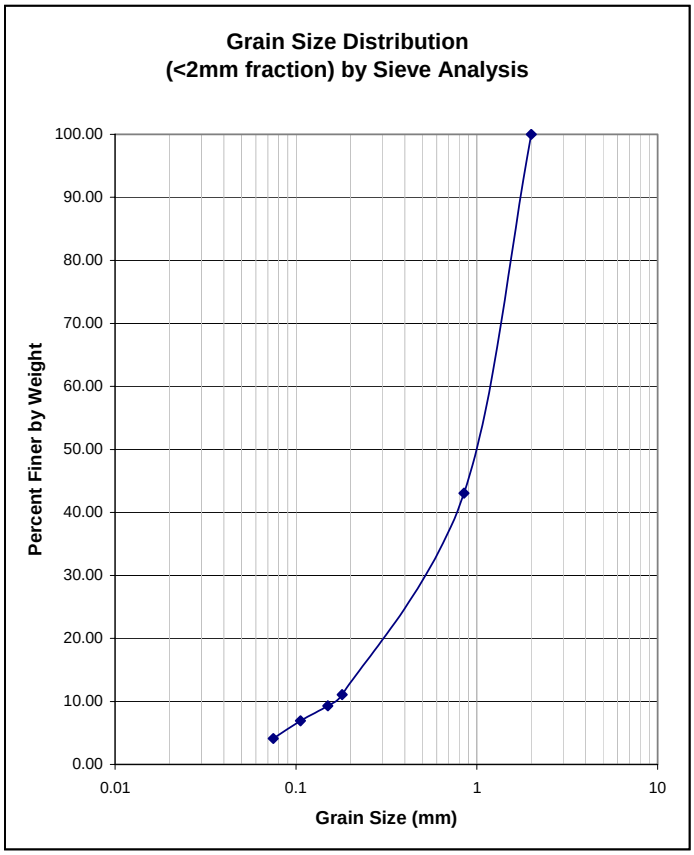
Project: WRTS Site Evaluation
Location: BH03, 8ft bgs
Date: 14-May-02
Analyses by: Brett Chambers

USDA Classification

Size (mm)	Fraction	Mass (g)	Percent
>2.0	Gravel		
2.0-1.0	Very Coarse Sand		
1.0-0.5	Coarse Sand		
0.5-0.25	Medium Sand		
0.25-0.1	Fine Sand		
0.1-0.05	Very Fine Sand		
0.05-0.002	Silt		
<0.002	Clay		
	Total		
> 2mm	Coarse sand		
2-0.05	Sand		
<0.05	Silt + Clay		

Unified Classification

Sieve No.	Size (mm)	Fraction	Mass (g)	Percent
#10	>2.0	Coarse Sand	62.43	51.7%
#20	2.0-0.85	Medium Sand	33.19	27.5%
#80	0.85-0.18	Fine Sand	18.63	15.4%
#100	0.18-0.15	Fine Sand	1.04	0.9%
#140	0.15-0.106	Fine Sand	1.38	1.1%
#200	0.106-0.075	Fine Sand	1.64	1.4%
-	< 0.075	Silt + Clay	2.41	2.0%
-	-	-	-	-
		Total	120.72	100.0%
	> 2mm	Coarse sand	62.43	51.7%
	2 - 0.075	Sand	55.88	46.3%
	<0.075	Silt + Clay	2.41	2.0%



Summary Statistics

d10	0.16
d25	0.40
d30	0.51
d50	1.00
d60	1.10
d75	1.30

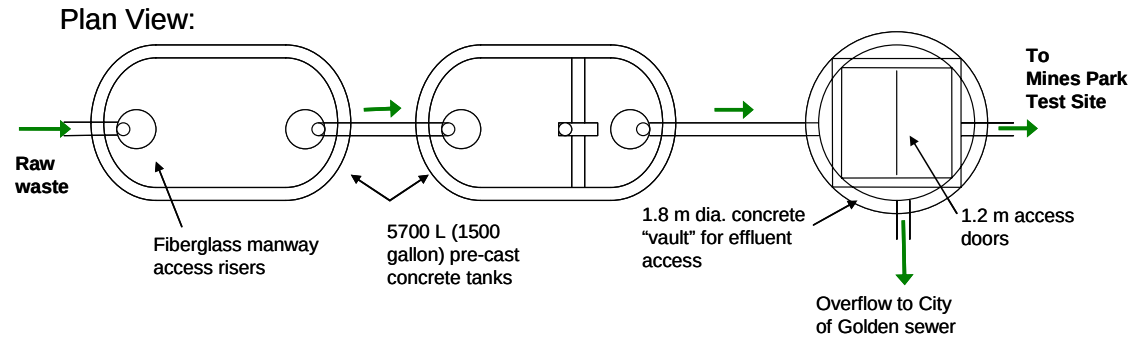
B COMPANION STUDY: WASTEWATER TREATMENT AS AFFECTED BY INFILTRATIVE SURFACE AREA AND HYDRAULIC LOADING RATE

B.1 Introduction

To facilitate OWS testing, research, and education through controlled field-scale experimentation, the Mines Park Test Site was established on the CSM campus with support from several entities. These entities include private industry, professional organizations, and local, state, and federal government agencies. Field testing was initiated at the Mines Park Test Site in May 2003 using *in situ*, three-dimensional test cells to evaluate the dynamic and interdependent behavior of biozone formation and hydraulic and purification processes during wastewater treatment by soil. Information gained during this testing aided the design, installation, and monitoring of the National Decentralized Water Resources Capacity Development Project (NDWRCDP) field study to evaluate the performance of engineered pretreatment units and their effects on biozone formation in soil described in this report. The results will also be used for validation and refinement of existing models from the watershed scale (Siegrist *et al.* 2004), to the site scale (HYDRUS 2-D), to an existing infiltration rate loss model (Siegrist 1987).

The establishment of the Mines Park Test Site southwest of the Mines Park student housing complex near the intersection of Hwy 6 and 19th Street in Golden, Colorado was completed in two phases:

- Phase 1 involved the installation of a wastewater interception and treatment facility to support OWS pilot-scale experiments and laboratory research. Phase 1 included installation of two 5,700-L buried pre-cast concrete tanks, an effluent filter, and a 1.8 m diameter concrete chamber for sample collection and pilot-scale treatment testing (Figure B-1).
- Phase 2, initiated during 2002, involved establishment of a field research area to enable controlled field-testing of OWS methods and technologies. A site evaluation of the research area was completed during spring 2002 (Lowe and Siegrist 2002). Field experiments to evaluate the performance of wastewater soil absorption systems as affected by infiltrative surface character and loading rate in an Ascalon sandy loam soil were initiated during fall 2002. Forty *in situ* test cells representing a pilot-scale soil absorption trench were installed. A set of test cells also receives tap water as a control.



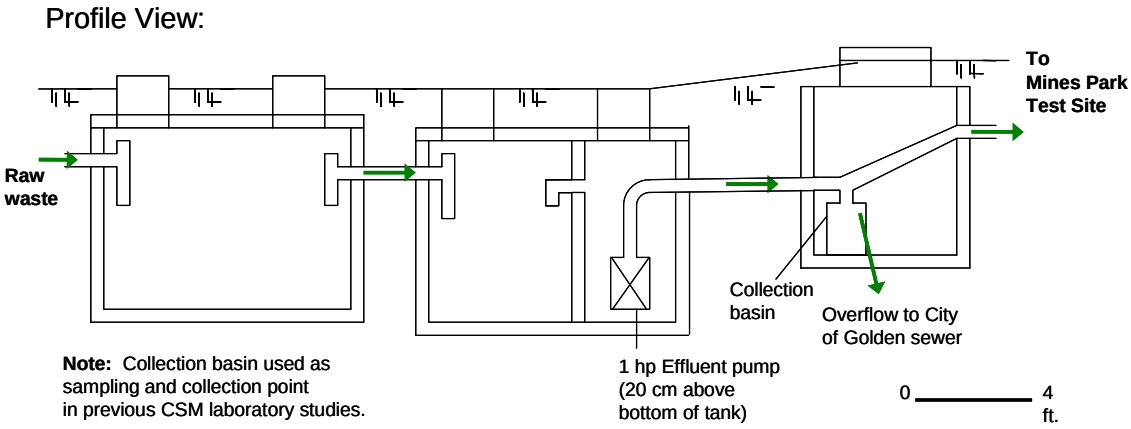


Figure B-1
Phase 1 Septic Tank and Effluent Vault at Mines Park

B.2 Materials and Methods

B.2.1 Site Selection and Evaluation

Initially, a site evaluation was conducted to assess the natural site and soil features critical to the design and performance of onsite wastewater treatment processes (Lowe and Siegrist 2002). This evaluation included:

1. Inspection of soil profiles within two backhoe test pits
2. Drilling and soil sample collection from nine soil borings from ground surface up to 6.7 m below ground surface (bgs)
3. Installation of seven shallow groundwater observation wells
4. Conducting percolation tests as prescribed by local OWS regulations

In addition, subsurface soil lithology and color were recorded for soil samples collected from the boreholes and analyzed for:

- Water content
- Total organic carbon
- Organic matter
- pH
- Total nitrogen
- Nitrate-nitrogen
- Ammonia-nitrogen
- Available potassium
- Percent sand/silt/clay
- Grain size distribution
- Cation exchange capacity

B.2.2 In Situ Test Cell Installation and Setup

Initial testing at the Mines Park Test Site was initiated in July 2002 to evaluate under field conditions, the performance of soil absorption systems as affected by infiltrative surface architecture (ISA) and hydraulic loading rate (HLR) in a sandy loam soil. Pilot-scale test cells were installed to mimic a typical soil absorption trench used to treat domestic septic tank effluent (STE). For this study, a replicated factorial design (2^2) was employed to evaluate three ISA (open, stone, and synthetic) and two HLR. Each condition, representative of feasible field conditions, was replicated five times (Table B-1) (3 ISA $\times$ 2 HLR $\times$ 5 replicates = 30 test cells) (Tackett *et al.* 2004). Half of the test cells receive STE at a design HLR of 4 cm/d delivered continuously during a 16-hr period (6 a.m. to 10 p.m.) each day through a single orifice in the center of the cell. The remaining test cells receive STE at a design HLR of 8 cm/d in the same fashion. By loading the test cells at daily design HLRs of 4 or 8 cm/d compared to the regulatory prescribed rate of 2 cm/d, six months of daily operation are anticipated to reflect periods of operation equal to approximately two and four years. That is assuming all of the applied STE is processed through the test cell. Six test cells were installed for ancillary testing. Finally, for control purposes, four test cells were installed and loaded with tap water at a design HLR of either 4 or 8 cm/d (Table B-1). Forty *in situ* test cells were installed (30 test cells + 6 ancillary cells + 4 controls) (Figure B-2).

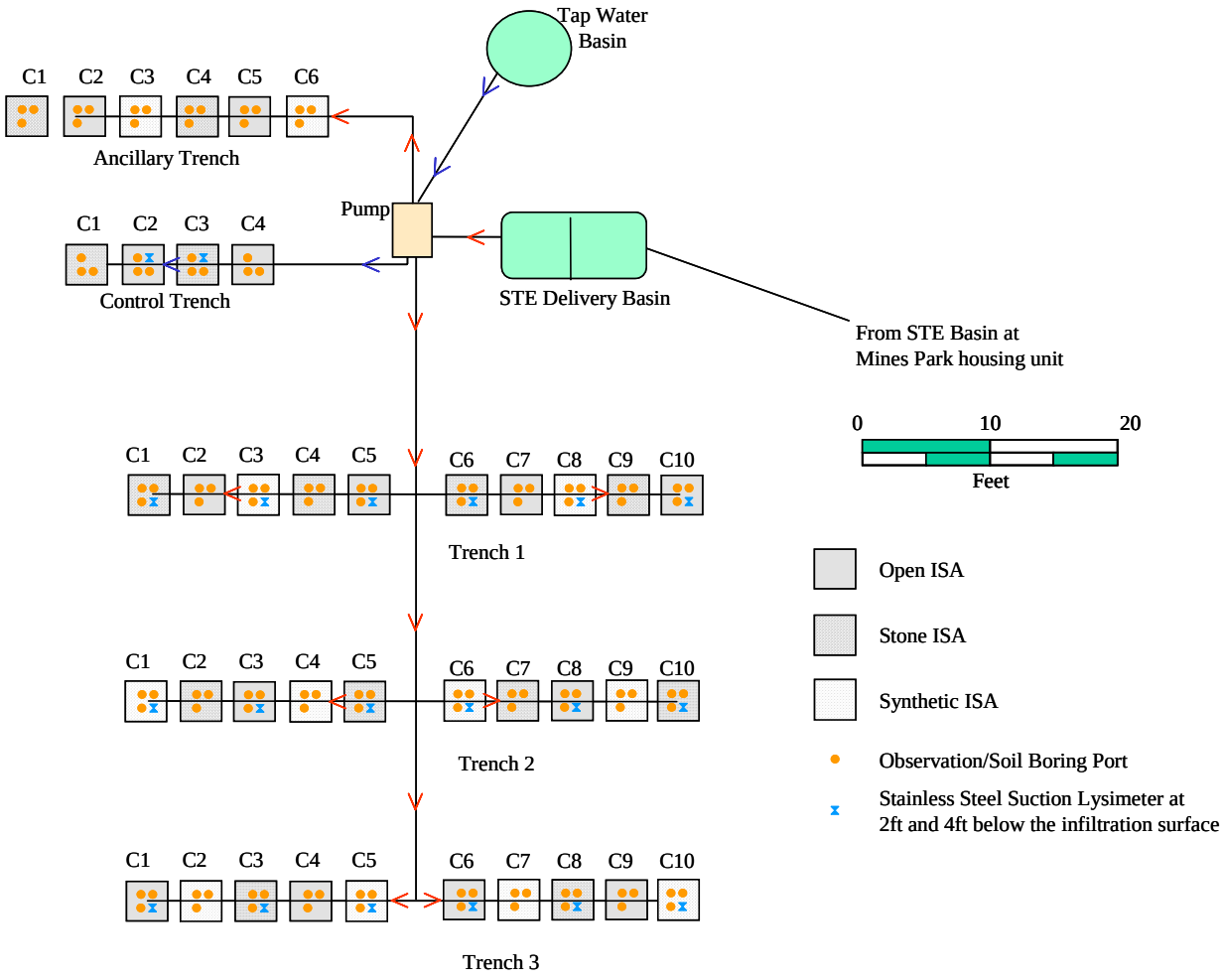


Figure B-2
Schematic Detail of Experimental Layout

Table B-1
Experimental Conditions for Test Cell Operation

Loading Regime	Test Cell ID	ISA	Exp. Design HLR	Loading Method Features
LR1	T1C1, T1C4, T2C2, T2C5, T3C3	Stone	4 cm/d (12.1 L/d per test cell)	STE with simulated gravity application; continuously dosed over 16 hr at 22 mL/min.
	T1C2, T1C5, T2C3, T3C1, T3C4	Open		
	T1C3, T2C1, T2C4, T3C2, T3C5	Synthetic		
LR2	T1C6, T1C9, T2C7, T2C10, T3C8	Stone	8 cm/d (24.2 L/d per test cell)	STE with simulated gravity application; continuously dosed over 16 hr at 44 mL/min.
	T1C7, T1C10, T2C8, T3C6, T3C9	Open		
	T1C8, T2C6, T2C9, T3C7, T3C10	Synthetic		
Control	TCC1	Stone	4 cm/d (12.1 L/d per test cell)	Tap water with simulated gravity application; continuously dosed over 16 hr at 22 mL/min.
	TCC2	Open		
Control	TCC3	Stone	8 cm/d (24.2 L/d per test cell)	Tap water with simulated gravity application; continuously dosed over 16 hr at 44 mL/min.
	TCC4	Open		

Average loading rates after 18 months of operation were 3.6 to 4.5 cm/d (design HLR of 4 cm/d) and 6.1 to 7.9 cm/d (design HLR of 8 cm/d)

Each test cell is approximately 67.3 cm long by 80 cm wide providing approximately 5,385 cm^2 of bottom area infiltrative surface (Figure B-3). Test cells were installed within a trench with the infiltrative surface (that is, bottom of the trench) located at approximately 76 cm below ground surface (bgs). To avoid potential hydraulic cross-connection between test cells, each cell was separated from the adjoining cell by approximately 30.5 cm. In addition, end plates were installed and sealed to the trench bottom and walls using a native soil slurry and bentonite. Stainless steel suction lysimeters were installed at 60 cm and 120 cm below the infiltrative surface within a 5-cm diameter borehole in 20 test cells (3 replicate conditions plus 2 control test cells) (Tackett 2004). The lysimeters were nested within the same borehole using a native soil slurry filter pack around the lysimeter and a bentonite seal between the two depths. During lysimeter installation, care was taken to avoid disruption of the infiltrative surface (for example, the drilling rig was not driven on the trench bottom). Prior to establishment of individual ISA, the infiltrative surface for each test cell was examined, photographed, and prepared in a similar fashion to remove any anomalous features and ensure replicate testing conditions between test cells.

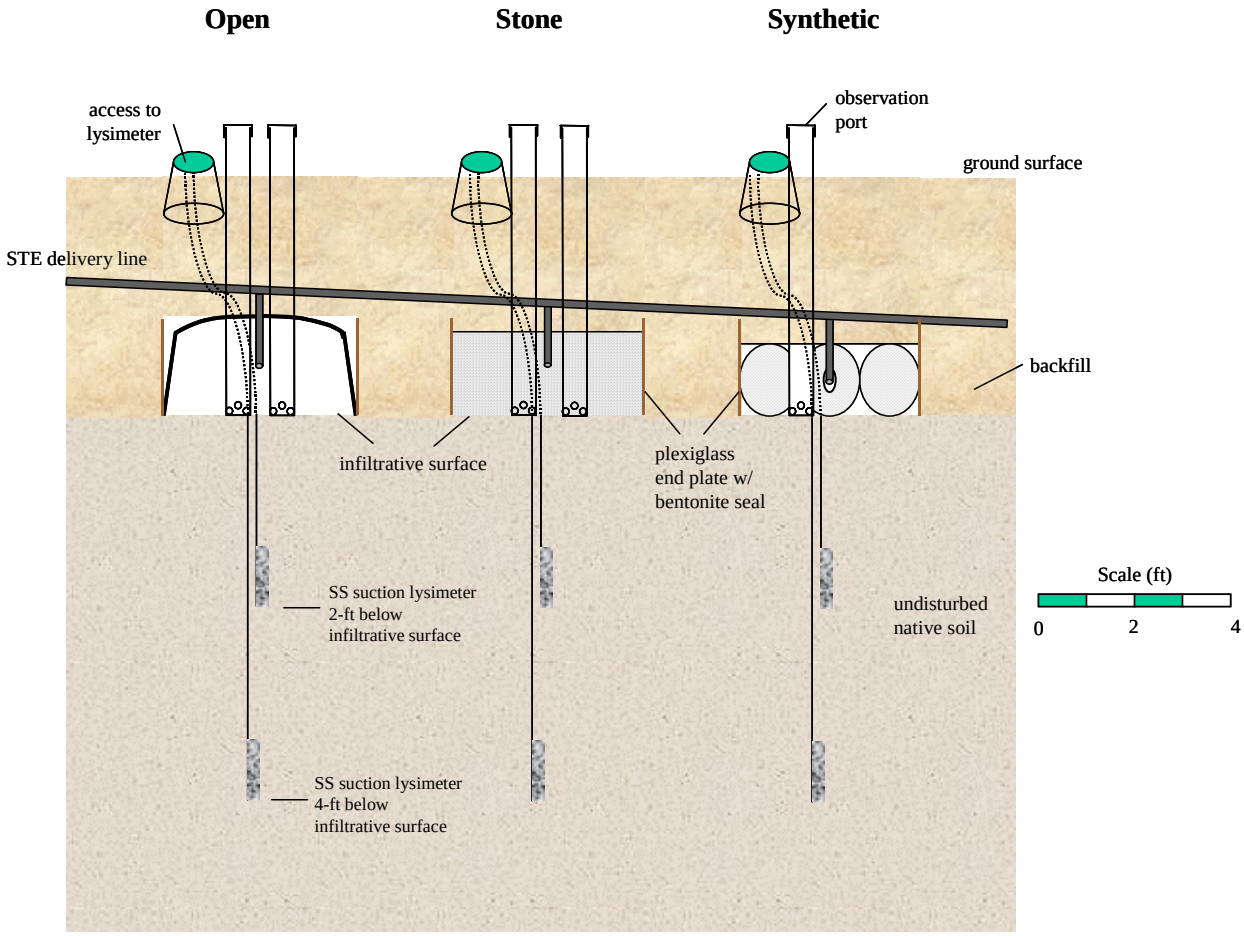


Figure B-3
Cross-Section View of Trench With Test Cells

The infiltrative surface of the ancillary test cells was not modified and is assumed representative of a typical OWS installation. Access ports were installed for inspection of the infiltrative surface and for collection of intact soil cores. Finally, the test cells were backfilled, compacted, and the site graded to minimize surface water ponding due to rainfall and snow.

B.2.3 Monitoring

Following test cell installation, baseline infiltration rates were measured using a constant head permeameter (Hargett *et al.* 1982; Siegrist 1987). At least three infiltration rate tests were conducted for each test cell with a constant 2.5 cm head at the soil infiltrative surface. Test cells were then loaded with clean water for seven weeks to establish equilibrium flow conditions prior to loading with STE. During clean water loading, a multi-surrogate tracer test was conducted (Van Cuyk *et al.* 2001b) with bromide (approximately 2,000 mg-Br/L) and MS-2 and PRD-1 bacteriophages (10^7 pfu/L) added to the clean water delivery basin and applied to the test cells during loading for 14 days. Samples were collected daily (24-hr composite samples) from each lysimeter within the test area. Bromide samples were analyzed using an ion selective probe and MS-2 and PRD-1 bacteriophage assays were made following the plaque-forming-unit (pfu)

technique (*Escherichia coli* and *Salmonella typhimurium* host, respectively) described by Adams (1959).

Loading of the test cells with STE began on May 5, 2003. Septic tank effluent from a nearby multifamily housing unit is being used and is the same STE used in previous research at CSM (Siegrist *et al.* 2002; Van Cuyk *et al.* 2001a). STE is pumped from the existing interception tank near the housing unit to a holding basin located at the Mines Park Test Site. From this holding basin, STE is delivered to individual test cells at the design HLRs of 4 and 8 cm/d (Figure B-1).

Routine field monitoring includes measurement of

- Applied effluent composition
- Applied effluent HLR
- Hydraulic behavior of the soil infiltrative surface (infiltration rate changes, ponding occurrence, and magnitude)
- Soil pore water quality

Grab samples of the STE are collected from the holding basin approximately weekly and analyzed for a suite of parameters including (APHA 1998, HACH 1998):

- pH
- Alkalinity
- Carbonaceous biochemical oxygen demand (cBOD)
- Chemical oxygen demand (COD)
- Total solids (TS)
- Total suspended solids (TSS)
- Nutrients (nitrogen and phosphorus)
- Fecal coliform bacteria

The volume of STE applied to each test cell is measured every one to two weeks by recording pumping rate and pumping duration. The infiltration rate (cm/d) of the infiltrative surface of each test cell was measured prior to effluent application, after one month of operation, and after 12 months of operation using a constant head technique (Hargett *et al*. 1982; Siegrist 1987). Measurements of the occurrence and magnitude of ponding of the infiltrative surface are made every one to two weeks using an observation port installed in each test cell. Soil pore water quality in the vadose zone was collected using stainless steel suction lysimeters after 1, 2, 3, 4, 5, 9, and 13 months of operation. Soil pore water samples were analyzed for: pH, alkalinity, dissolved organic carbon, and total nitrogen, ammonium nitrogen, nitrate-nitrogen, and total phosphorus (APHA 1998; HACH 1998).

During July 2004, a second multi-component surrogate tracer test was conducted. Similar to the clean water tracer test, bromide (approximately 1,500 mg-Br/L) and MS-2 and PRD-1 bacteriophages (approximately 10^7 pfu/L) were added to the STE holding basin and applied to the test cells during loading. Composite samples were collected every 48 hours from each lysimeter and analyzed using the methods described during the clean water tracer test. After approximately 21 days of tracer addition, duplicate intact cores were collected from selected test

cells and analyzed for the presence of the tracers at selected depths below the infiltrative surface. Analyses of the soil samples at multiple depths were also made for morphology, water content, total organic carbon, total nitrogen, ammonium nitrogen, nitrate nitrogen, total phosphorus, available phosphorus, and fecal coliform bacteria. Analysis and evaluation of the soil core characterization is ongoing. Preliminary soil core results applicable to the engineered pretreatment unit study are presented in Section 3.2.7 of this report.

Data collection and analysis is ongoing and will enable assessment of the time-dependent changes in soil infiltration rates as affected by ISA and effluent loading (Minitab 2002; Snedecor and Cochran 1980). Analysis of equilibrium infiltration rates achieved after system maturation will be assessed by analysis of variance (ANOVA) and other appropriate statistical tests (Siegrist *et al.* 2002). Analysis of variance and other statistical tests will also be completed to assess any differences or trends observed in soil properties with depth and between the test cells and the different experimental conditions.

B.3 Results and Discussion

This section provides results and discussion of the baseline characterization and routine monitoring.

B.3.1 Field Site Characteristics

Based on the site characteristics and soil conditions observed, the site southwest of the Mines Park housing complex was judged to be suitable for wastewater treatment and reclamation research while satisfying the general goal of public health and environmental protection. The results of the site evaluation are presented in Section 3.2.2 and Appendix A of this report. It is important to note that the specific goal of the site evaluation was to ensure that high groundwater, or low permeability of the subsurface native material would not diminish the soil treatment efficiency of proposed *in situ* test cells.

B.3.2 Baseline Characterization

Baseline infiltration rates were measured within each test cell using a constant head permeameter (2.5 cm head at the soil surface). A minimum of three tests was completed for each test cell with over 800 measurements made across the site. Based on these tests, the infiltration rate of the soil was consistent across the site with an average infiltration rate of 41.8 cm/d (standard deviation of 20.8 cm/d) (Figure B-4). Prior to wastewater loading to the test cells, a clean water multi-component surrogate and tracer test was conducted to evaluate baseline travel times. Results indicated consistent soil properties across the site with average bromide breakthrough at 50% ($C/C_o = 0.5$) at 60 cm below the infiltrative surface approximately eight days after tracer/surrogate addition and approximately 13 days after tracer/surrogate addition at 120 cm below the infiltration surface (Figure B-5). As expected, the bromide curve for each test cell showed a similar increasing trend during addition and a decreasing trend after tracer addition was terminated. Bacteriophage was sporadically detected (PRD-1 at less than 10 pfu/mL in under 30% of the samples and MS-2 at less than 50 pfu/mL in under 20% of the samples) in the

lysimeter samples indicating a high removal of the bacteriophage by the soil due to either inactivation or adsorption.

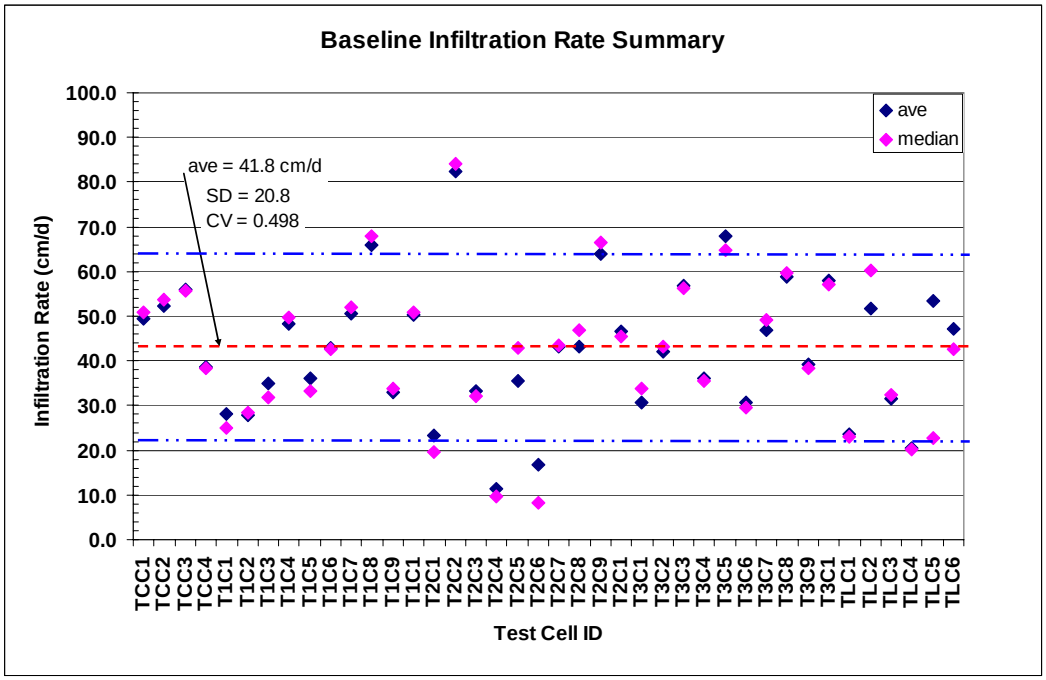


Figure B-4
Baseline Infiltration Rates Measured by Constant Head Permeameter

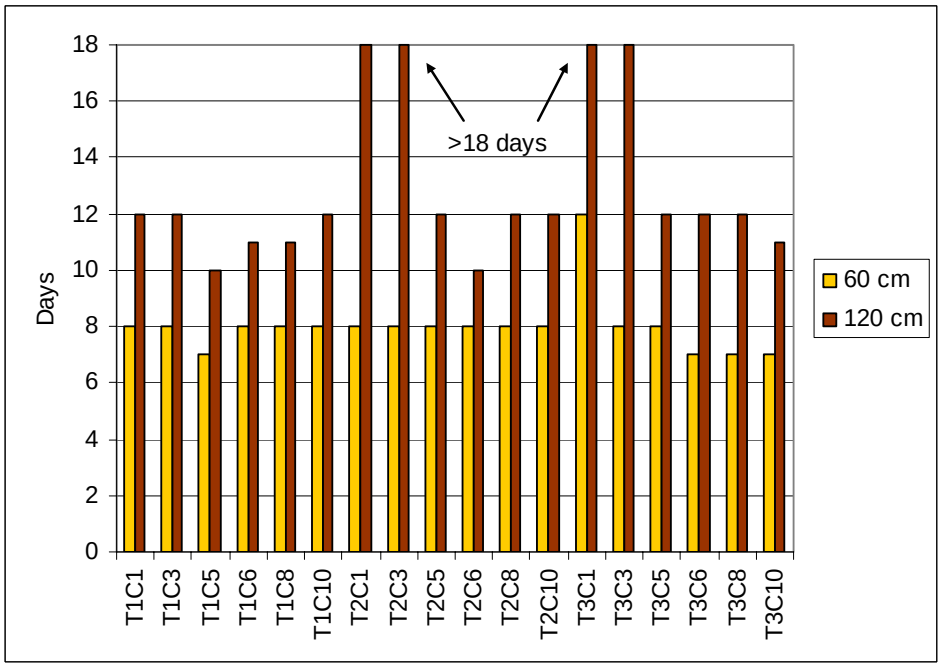


Figure B-5
50% Bromide Breakthrough During Clean Water Tracer Test as Measured in Lysimeters at 60 and 120 cm Below the Infiltrative Surface

B.3.3 Effluent Characterization

Loading the test cells with STE began on May 5, 2003. During the test cell operation, STE applied to the test cells was sampled and analyzed in conjunction with vadose zone solution sampling. Table B-2 displays the chemical composition of the STE prior to distribution to the test cells.

Table B-2
Effluent Composition Applied to Test Cells (Based on STE Grab Samples From the Delivery Basin, May 2003–October 2004)

Parameter	Units	Average	Std. Dev.	Coeff. Var.	Range
pH	–	–	–	–	6.7–7.6
Alkalinity	mg $CaCO_3$/L	274	42.7	0.16	150–410
TS	mg/L	318	263.0	0.83	122–1935
TSS	mg/L	37	39.1	1.06	5 – 175
DOC	mg-C/L	34.2	12.3	0.36	14.8–59.3
COD	mg/L	322	124.6	0.39	46–799
$cBOD_5$	mg/L	176	87.5	*0.50*	30–463
Total Nitrogen	mg-N/L	119	88.6	0.74	12–404
Nitrate-Nitrogen	mg-N/L	2.2	1.31	0.60	0.1–4.6
Ammonium Nitrogen	mg-N/L	61.7	19.4	0.37	40–150
Total Phosphorus	mg-PO_4/L	25.8	11.2	0.43	0.1–52.8
Fecal Coliform Bacteria	cfu/100 mL	–	–	–	1.2×10^3 to 3.7×10^5

B.3.4 Hydraulic Performance

Within approximately one month of operation with STE design application at 4 or 8 cm/d (2 or 4 times the design rate for this soil type—actual rates after 18 months of operation ranged from 3.6 to 4.5 cm/d and 6.1 to 7.9 cm/d, respectively), the soil infiltrability had declined. Continuous ponding was present, indicating the development of a biozone (Siegrist and Boyle 1987). This loss in infiltration rate was consistent with that expected, based on predictions made using the model of Siegrist (1987) which calculates infiltration rate loss as a function of cumulative mass loading of total BOD and TSS. Based on an STE composition with tBOD = 275 mg/L and TSS = 60 mg/L and a hydraulic loading rate of 4 cm/d, model predictions revealed that the loss in

infiltration rate would lead to ponding of the infiltrative surface after approximately two months of operation. At 8 cm/d, this was anticipated to occur after one month. To prevent overflowing of effluent from the test cell, "end state" of an individual test cell was defined at continuous ponding heights of 20 cm or more over three consecutive weeks. At end state, effluent delivery to the test cell was terminated, the infiltration rate was measured, and effluent delivery to the test cell was restarted at a reduced hydraulic loading rate of 2 cm/d.

Comparison of infiltration rates measured using a constant head permeameter before STE application to those after one month of operation revealed a 60 to 85% reduction in infiltration rate (Siegrist *et al.* 2004). After 18 months, infiltration rates for test cells continuously loaded at 8 cm/d have declined by 90% or more (typically in the range of 0.3 to 3.0 cm/d) with 13 of 15 test cells reaching end state; 4 of 5 open test cells, 4 of 5 stone test cells, and 5 of 5 synthetic test cells. Of the test cells that have reached end state, they appear to be moving toward long-term acceptance rates, and at a comparable infiltration rate loss, the open ISA has processed a greater cumulative volume of STE (Figure B-6). Similarly, Figure B-7 illustrates the average cumulative volume of STE processed at end state for each ISA. Only 1 of 15 test cells continuously loaded at 4 cm/d had reached end state after 18 months of operation.

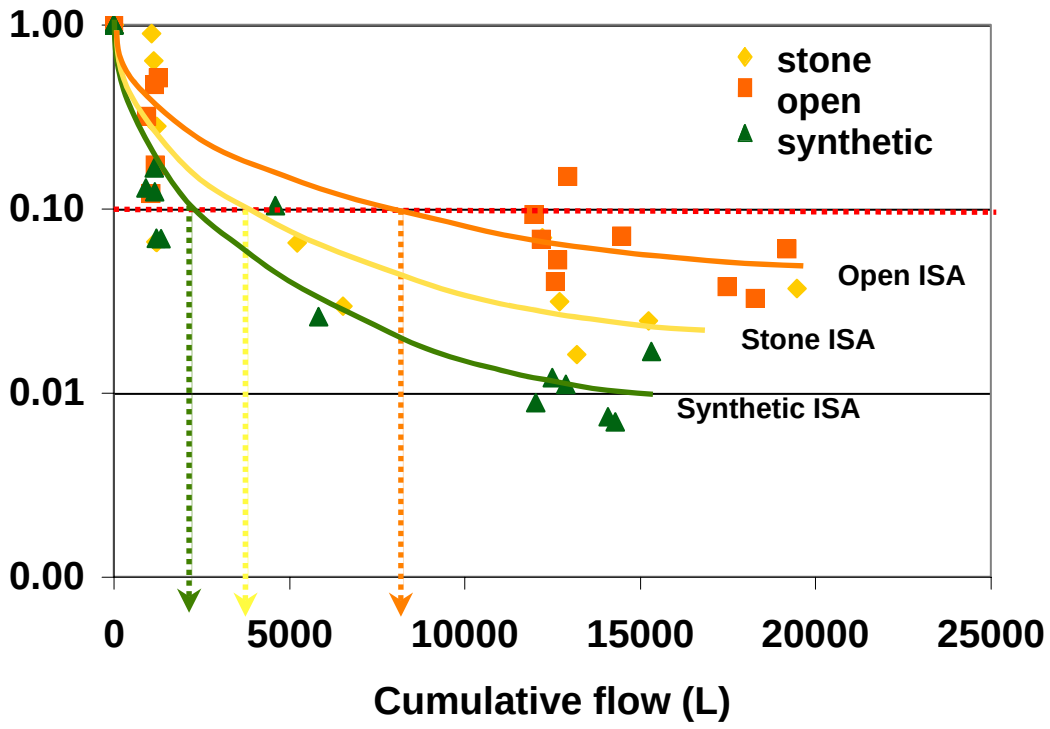


Figure B-6
Infiltration Rate Loss With Cumulative Volume of STE Processed at End State[1]

[1] y-axis is the Ratio IR_t / IR_o where IR_o is the baseline infiltration rate and IR_t is the end state infiltration rate
Only test cells loaded at 8 cm/d are shown

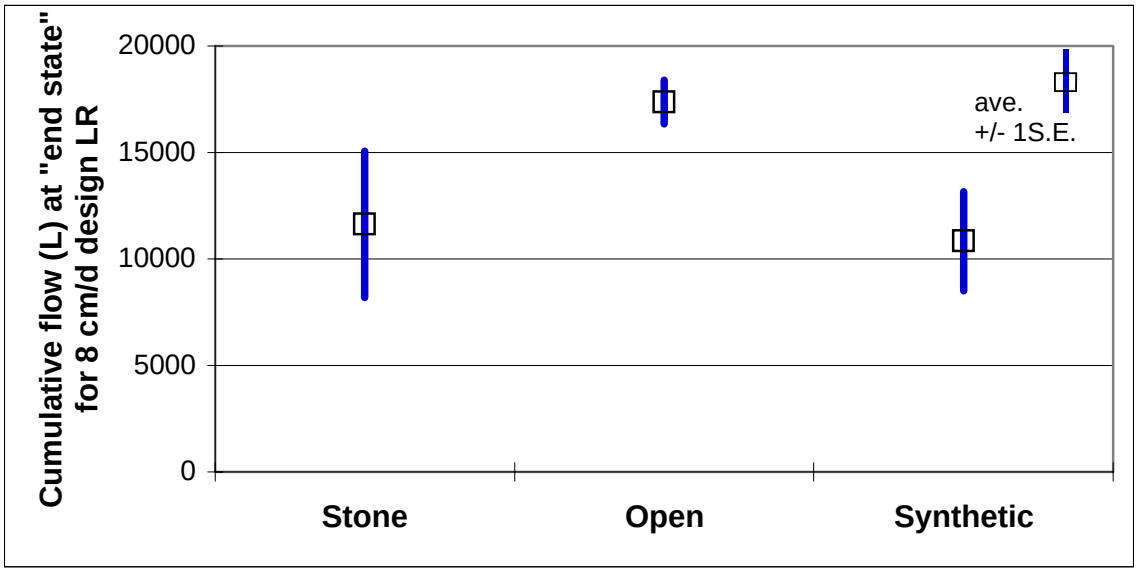


Figure B-7
Cumulative Flow of STE Processed at End State for 8 cm/d Design HLR

B.3.5 Purification Performance

Assessment of the treatment performance as measured by collection and analyses of soil pore water quality using stainless steel suction lysimeters at 60 cm and 120 cm below the infiltrative surface is ongoing. Based on monitoring completed to date, the following behavior has been observed. Results from lysimeter sampling conducted during the first nine months can be found in Tackett 2004 and Tackett *et al.* 2004.

Organic carbon removal efficiency was greater than 90% for both loading rates after nine months of operation (Figure B-8). However, with respect to mass removal, test cells receiving effluent at a higher HLR were able to remove more mass of organic carbon. A trend of increased DOC removal efficiency with time was evident for both HLRs. Depth of vadose zone was not a significant factor in organic carbon removal for the lower loading rate. However, additional removal was evident from 60 to 120 cm for the 8 cm/d loading rate. Depth of vadose zone soil becomes more important as loading is increased. However, the processes responsible for the majority of the DOC attenuation appear to be carried out in the shallow vadose zone.

Initially, little-to-no nitrogen was detected in soil solution samples at either depth across the study site, indicating minimal migration through the vadose zone during the first month of operation. Minimal amounts of nitrate were found in soil solution samples from weeks two to five, indicating nitrification processes were not yet established and retention of effluent ammonium was most likely due to interactions with the cation exchange capacity of the soil matrix.

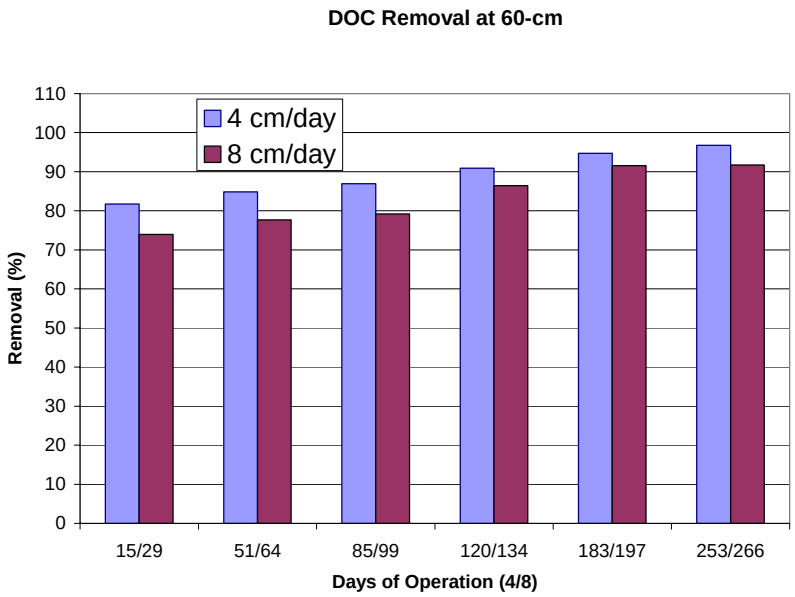


Figure B-8
Average DOC Removal (%) at 60 cm Below the Infiltrative Surface (Tackett 2004)[2]

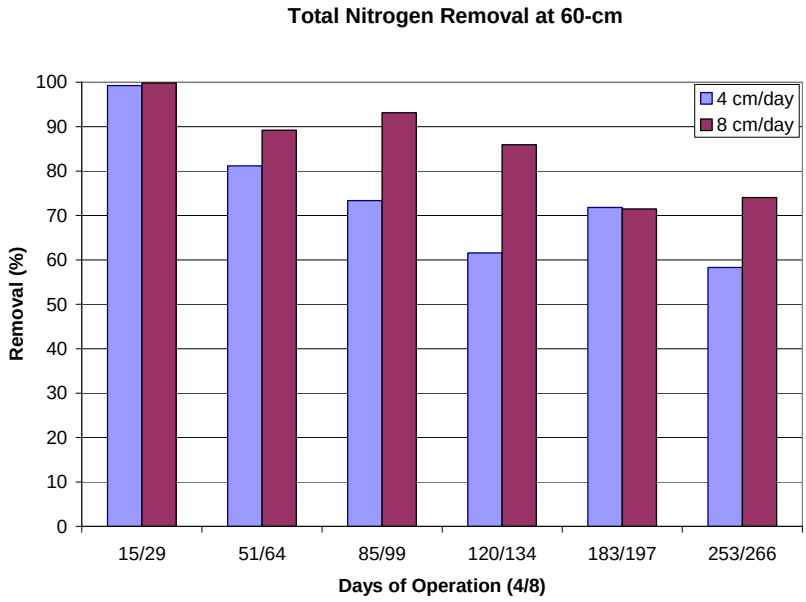


Figure B-9
Average Total Nitrogen Removal (%) at 60 cm for All ISA[3]

[2] Days of operation at 4 or 8 cm/d

[3] Mean STE total nitrogen = 119 mg-N/L, based on initial 9 months of operation (Tackett 2004) Days of operation at 4 or 8 cm/d

This initial near complete removal of total nitrogen, as evidenced by minimal nitrate, ammonium, and total nitrogen in soil solution samples, proceeded to decline with a concomitant increase in nitrate and total nitrogen as operational time continued and nitrification commenced to varying degrees in all test cells. Additionally, a decrease in effluent alkalinity was observed as nitrification began. As expected, no significant difference among infiltrative surface architectures was found for total nitrogen removal or in the nitrate concentrations observed at 60 cm. Nitrification was carried out to a greater extent in the 4 cm/day loading regime (based on reduction in total nitrogen concentrations in STE compared to total nitrogen concentrations in lysimeter samples). With respect to total nitrogen removal, test cells at the higher loading rate were able to remove a greater mass of total nitrogen (Figure B-9). No difference appears to exist between nitrogen removal at 60 cm and 120 cm for test cells receiving 4 cm/d.

Removal of phosphorus was observed to be near 100% throughout the duration of this field study. There appears to be no difference in removal efficiency between hydraulic loading rates (Table B-3). Removal efficiency calculations were based on the average total P concentration from all 60 cm sampling depths for all infiltrative surface types and the average STE value. Phosphorus removal is nearly complete during percolation from the infiltrative surface to the 60 cm sampling depth indicating the processes responsible for phosphorus attenuation occur in the shallow vadose zone. Phosphorus removal was complete with only 60 cm of vadose zone travel for all ISA and both loading rates. Because of the finite sorption capacity of the soil, the depth of vadose zone will play an increasingly more important role in P attenuation as operational time continues for years.

Table B-3
Percent Total Phosphorus Removal at 60 cm Below the Infiltrative Surface (Tackett 2004)

	% Removal Versus Operational Time					
HLR = 4 cm/d	99.7%	99.6%	99.8%	99.7%	99.9%	99.6%
Operational Days	15	51	85	120	183	253
HLR = 8 cm/d	99.2%	99.6%	99.3%	99.8%	99.9%	99.6%
Operational Days	29	64	99	134	197	266.0

B.4 Summary

The Mines Park Test Site was established on the CSM campus in 2002 to facilitate OWS testing, research, and education through controlled field-scale experimentation. Site evaluation work completed in spring 2002 deemed the area suitable for wastewater treatment and reclamation research while satisfying the general goal of public health and environmental protection. Installation of the *in situ* test cells was completed in fall 2002 with opearation and monitoring of the 40 *in situ* test cells to continue through at least fall 2005. Results from baseline infiltration rates and the clean water tracer test indicate that the soil conditions across the site are comparable.

Testing to evaluate wastewater treatment in soil as affected by infiltrative surface character and loading rate in a sandy loam soil is ongoing. Preliminary results indicate that within approximately one month of operation, the soil infiltrability had declined and continuous ponding was present. A previously determined infiltration rate loss model (Siegrist 1987) was useful to predict this observed infiltration rate loss in the field. Based on test cells that have reached end state, the ISA does effect the infiltration rate as evidenced by a higher open ISA infiltration rate compared to stone or synthetic ISA infiltration rates at comparable cumulative STE volume processed. After nine months of operation, high removal rates of nitrogen, phosphorus, and carbon were observed with no effect attributed to ISA. Changes to the soil properties, hydraulic behavior of test cells under the design conditions, and treatment performance will continue to be monitored through at least fall 2005. Results from this work will be used for model validation and refinement. Publications describing the testing at the Mines Park Test Site will be forthcoming in conference proceedings and journals.

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C SOIL CORE SAMPLE ANALYSIS METHODS

C.1 Analysis Methods

Analyses of the soil sub-samples taken from the soil cores were made for the following characteristics by the methods described (all soil results are expressed per gram or kilogram of dry soil):

- Soil color was recorded using the Munsell color chart
- Water content was measured gravimetrically and recorded as percent dry weight
- pH was measured on a 1-to-1 (soil-to-DI) extract using an electrode
- Bromide was measured on a 1-to-1 (soil-to-DI) extract using ion chromatography
- Total organic carbon via combustion

Exchangeable ammonium was measured using a 2.0 M KCl extraction. Ammonium is held in an exchangeable form in soils in the same manner as exchangeable metallic cations. Exchangeable ammonium was extracted by shaking soil in the presence of 2.0 M KCl. Because nitrate is water soluble, it was extracted by the same KCl solution. Briefly:

1. Weigh 5.0 grams of soil into a conical flask
2. Add 50 mL of 2.0 M KCl
3. Reduce to 1 gram and 10 mL to keep the 1 to 10 ratio
4. Shake for 30 minutes
5. Filter into pre-cleaned flask

Exchangeable ammonia and nitrate were analyzed using a spectrophotometer (NH_4-N, HACH Method 10030; NO_3-N, HACH Method 10020) (HACH 1998). Sample extract was frozen and will be rerun using ion chromatography (IC) for nitrate due to the bromide interference with the HACH method for nitrate.

Available P was measured by adding 2 g of dry soil to 20 mL of a 0.025 M HCL in 0.03 M NH_4F extracting solution (1-to-10 soil-to-extractant) followed by analysis using inductively coupled plasma (ICP) emission spectroscopy (Bray 1 method) (Soil and Plant Analysis Council 1992; USDA 2004; and NCSU 2004).

Total P was measured using a nitric acid / perchloric acid digestion on a 1 to 10 (soil to acid) extract using ICP emission spectroscopy (Carter 1993).

Phospholipid extraction (Total Viable Soil Biomass, PLE) was measured following the description in Rauch & Drewes (2005). All glassware used was cleaned in 0.1 N HCl for 24 hours, rinsed with milli-Q water and oven dried. For each sample, triplicates of 0.3 – 1 g soil as wet weight were filled into 20 mL glass vials with teflon screw caps with a clean spatula. Phospholipid extraction from soil was performed in a homophasic solution of milli-Q water, methanol (HPLC-grade) and chloroform (HPLC-grade) to keep ratios at 0.8-to-2-to-1 by volume. The volume of chloroform in mL was five to seven times higher than the fresh weight of the soil sample in grams. The vials were shaken for 24 hours. Purification of the phospholipid extract was achieved by adding 0.0306 M sulfuric acid and chloroform to receive a final ratio of chloroform-to-methanol-to-water of 1-to-1-to-0.9. The vials were shortly mixed after each addition and phases allowed to separate for at least eight hours.

The upper methanol-water phase was carefully removed without disturbing the remaining sample. An aliquot of the lower chloroform layer was then transformed into a glass ampule and completely dried under a purified air stream while warmed to 39 °C. Subsequently, 0.9 mL of saturated potassium persulfate reagent (5% potassium persulfate in 0.36 N sulfuric acid, stored light protected and refrigerated) was added to each ampule. The ampules were shaken and then air sealed over a methyl acetylene flame. Digestion was allowed for a minimum of eight hours at 96 °C to separate inorganic phosphate from fatty acid tailings.

After cooling, 0.2 mL ammonium molybdate reagent (2.5% $(NH_4)_6Mo_7O_{24} \times 4H_2O$ in 5.72 N sulfuric acid) was mixed into each ampule in presence of inorganic phosphate, ammonium molybdate forms, a phosphomolybdate complex. After 10 minutes, 0.9 mL of malachite green reagent was added (prepared by dissolving 0.111 percent polyvinyl alcohol in water at 80 °C and adding 0.011 percent malachite green hydrochloride after cooling). Color was allowed to develop for 30 minutes. Green color intensity was determined at 610 nm on a UV/VIS spectrometer against milli-Q water. Blanks were prepared in triplicates without addition of soil following the same procedure. Average blank absorbance readings were subtracted from sample results. Samples were compared against a standard curve ranging from 2.5–25 nmol PO_4 per ampule ($r^2 = 0.99$). Dry weight of the samples was determined after oven drying at 104 ºC.

Labile Polysaccharide Extraction (modification of Canadian Soil Science Society) was measured on soil prepared by grinding and sieving (1.7 mm) before extraction. One gram of the soil was weighed into an autoclaveable glass bottle and mixed with 100 mL of 0.5 M sulfuric acid. The bottle was sealed and autoclaved for 1 hour at 103 kPa. The extract was filtered through a 47 mm glass fiber filter. Soil residue was washed with an extra 100 mL of 0.5 M sulfuric acid that was added to the original filtrate. A phenol sulfuric acid test was used to estimate colorimetrically the total sugar content. This test consisted of mixing 0.5 mL of the sample, 0.5 mL of 5% phenol, and 2.5 mL concentrated sulfuric acid (rapidly mixed) in a glass test tube. The mixture was allowed to sit for 50 minutes, and then analyzed for absorbance using a spectrophotometer. The absorbance values were converted to mg/L sugar using a calibration curve that was prepared from glucose standards of 0, 25, 50, 75, and 100 mg/L (R^2=0.99).

Humic substance extraction (modification of Canadian Soil Science Society) was measured on soil prepared in the same way as the polysaccharide extraction. Fifteen grams of the soil was weighed into a 250 mL centrifuge bottle and mixed with 150 mL of 0.5 M HCl to remove plant debris and inorganic forms of carbon, nitrogen, phosphorus, and sulfur. The mixture was allowed to sit for one hour, and then centrifuged at 3,500 rpm for 10 minutes and the HCl was poured off. All subsequent centrifuge runs were at the same speed and length of time as described above. The soil was washed with an extra 150 mL of DI water, centrifuged, and the DI was poured off to remove salts. 150 mL of 0.5 M NaOH was added to the centrifuge bottle, which was placed on an end-over-end shaker for 18 hours. The extract was centrifuged, and the supernatant was poured into separate 250 mL centrifuge bottles. The soil remaining in the original centrifuge bottle was dried at 50 ^{o}C for 20.5 hours and was quantified as humic. The NaOH extract was acidified with 6 M HCl until a pH of 1.5 was attained and the humic acid began to precipitate. The acidified extract was centrifuged and the supernatant (fulvic acid fraction) was saved for analysis. The precipitate was washed once with DI water and filtered through a 47 mm glass fiber filter, dried at 50 ^{o}C for 16 hours. The weight was quantified as the humic acid fraction. The fulvic acid fraction was diluted and analyzed with the Sievers DOC instrument.

Fecal coliform analysis on soil core samples was performed aseptically by taking a known weight (approximately 5 grams) of moist soil and adding 40 mL of 1.5% beef extract solution to yield a final dilution of approximately 1-to-8 (soil-to-beef extract). APHA (1998) method 9221A suggests extraction for coliform bacteria in sediments and sludges using 10% phosphate buffered saline (PBS). However, a comparison of extraction methods conducted at the bench scale at the CSM microbiology laboratory using six different extractants (including PBS) proved beef extract to be the most efficient method for removing the coliform bacteria (Masson 1999). Following the addition of beef extract, samples were shaken for 30 minutes at about 60 rpm and then allowed to settle for at least 15 minutes prior to analysis. An aliquot of liquid (3–10 mL) was withdrawn from mid-depth of a sterile 50 mL conical (Masson 1999) and analyzed directly (for low levels) or diluted as needed (for high levels). Analyses for fecal coliform bacteria were made according to the membrane filtration method (APHA 1998 9222D).

Extraction for bacteria (heterotrophic plate count, fecal coliform and *E. coli*) was performed by taking 2–6 grams of soil placed into a pre-weighed 50 mL sterile conical. The exact weight of the sample was recorded to relate the microbiological data to dry weight of soil. 40 mL of an autoclaved 1.5% (w/v) beef extract solution was added to the 50 mL sterile conical. A slurry was made by agitating the soil and extract solution on a rotary wheel (approximately 150 rpm) for two minutes. The sample was then allowed to settle for one minute. Five mL of the supernatant was then extracted from the 50 mL conical at the 20 mL mark. This aliquot was placed in a 15 mL conical and used for microbiological analysis.

Heterotrophic plate count (HPC) was measured on 1 mL of the above aliquot placed into a 3 mL test tube. The aliquot was serially diluted into PBS (that is 100 uL sample into 900 uL PBS = 10^{-1}) until the target dilution range for plating was 10^{-20}. The resulting dilutions were vortexed and 100 uL plated on to HPC. The sample was spread on the plate and incubated at 37 °C for 48 hours (APHA 1998, 9215 C).

Escherichia Coli (*E. coli*) was measured on the same soil extract dilutions used for the detection and enumeration of *E. coli*. ChromAgar ECC was used as the media of choice (Alonso *et al.* 1999). Samples were filtered using the membrane technique and incubated at 45.5 °C for 24 hours.

Virus measured on 1 mL of the extracted sample aliquot was used for plaque assays (MS-2 and PRD-1) according to Van Cuyk *et al.* (2002). MS-2 and PRD-1 bacteriophage assays were made following the plaque-forming-unit technique (*Escherichia coli* and *Salmonella typhimurium* host, respectively) described by Adams (1959).

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