ABIOTIC NITROGEN REMOVAL MECHANISMS IN RAPID INFILTRATION WASTEWATER TREATMENT SYSTEMS

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- Abstract -

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The rapid infiltration (RI) land treatment process is a reliable, cost effective method for secondary and/or tertiary treatment of municipal wastewaters. When properly designed and operated, RI systems can achieve a significant level of nitrogen removal via coupled biological processes, namely nitrification-denitrification. Generally, it is believed that lower overall nitrogen levels can be achieved when influent wastewater is fully nitrified. However, at a specific RI facility located in Colton, CA higher nitrogen removals were observed when non-nitrified influent wastewaters were introduced. As a result, it was first hypothesized that an abiotic mechanism, ammonium adsorption, to the soils was occurring. This hypothesis, led to the conduct of an initial effort to evaluate the sorptive phenomenon that was occurring at this site. As a result of that initial effort, it was determined that ammonium adsorption was not occurring and that no nitrogen removal was observed under abiotic (sterile) conditions. Nitrogen removal was observed only under biotic conditions.

Subsequent to that initial effort, a second study was conducted in an effort to confirm and better understand the biological nitrogen removal mechanisms that are occurring at the Colton RI facility. In addition, experiments were conducted to evaluate whether nitrogen removal could be enhanced at the facility via organic carbon amendment to the influent wastewater. For design purposes, a 2:1 mass ratio of organic carbon to nitrogen is recommended for nitrogen removal in RI systems. The normal organic carbon to nitrogen ratio at the Colton RI facility is 1:3, highly organic carbon deficient. Experimental systems were amended with additional organic carbon in the form of methanol. Additional organic carbon in the Colton RI facility influent water may improve the denitrification rate within some portions of the soil column.

INTRODUCTION AND PROBLEM STATEMENT

The rapid infiltration (RI) land treatment process is a reliable, cost effective method for secondary and/or tertiary treatment of municipal wastewaters. When properly designed and operated, RI systems can achieve a significant level of nitrogen removal via coupled biological processes, namely nitrification-denitrification. Generally, it is believed that lower overall nitrogen levels can be achieved when influent wastewater is fully nitrified. However, at a specific RI plant located in Colton, CA, the Rapid Infiltration and Extraction (RIX) Facility, higher nitrogen removals were observed when non-nitrified influent wastewaters were introduced. As a result, it was hypothesized that an abiotic mechanism, ammonium adsorption, to the soils was occurring. This hypothesis, led to the conduct of a previous study to determine the sorptive phenomenon that was occurring at this site [3]. However, it was determined that ammonium adsorption was not occurring. Nitrogen removal was not observed under abiotic (sterile) conditions. Nitrogen removal only was observed under biotic conditions.

The removal of nitrogen at the RIX facility is important because the effluent discharged into the Santa Ana River will be required to be below the National Primary Drinking Water Standard of 10 mg N/L in the near future [12]. At the RIX facility, the average effluent nitrogen concentration has varied from 16.5 to 5.0 mg/L over the past five years. Thus, while at times, the effluent nitrogen concentration has been below 10 mg/L, it has not been so consistently.

In order to improve nitrogen removal in the RIX system, it is important to understand the mechanisms by which nitrogen is removed. As part of a project funded by the UC Water Resources Center and the City of San Bernardino, the Department of Chemical and Environmental Engineering at the University of California, Riverside undertook a study to confirm and better understand nitrogen removal at the RIX facility.

Generally, it is believed that lower overall nitrogen levels can be achieved when influent wastewater is fully nitrified. However, at the RIX facility, higher nitrogen removals were observed when non-nitrified influent wastewaters were introduced. At first, it was hypothesized that an abiotic mechanism, ammonium adsorption, to the soils was occurring. This hypothesis, led to the conduct of a previous study to determine the sorptive phenomenon that was occurring at this site [3].

As a result of this prior study it was determined that ammonium adsorption was not occurring. No nitrogen removal was observed under abiotic (sterile) conditions. Nitrogen removal only was observed under biotic conditions. This present study was conducted in an effort to confirm and better understand the biological nitrogen removal mechanisms that are occurring at the RIX facility.

In addition, experiments were conducted to evaluate whether nitrogen removal could be enhanced at the facility via organic carbon amendment to the influent wastewater. For design purposes, a 2:1 mass ratio of organic carbon to nitrogen is recommended for nitrogen removal in RI systems. The normal organic carbon to nitrogen ratio at the RIX facility is 1:3, highly organic carbon deficient. Experimental systems were amended with additional organic carbon in the form of methanol.

OBJECTIVES

The overall objectives of this latter study were to confirm and evaluate the biological mechanisms for nitrogen removal at the RIX facility. Specific objectives were to:

1. Create and operate pilot-scale experimental RI systems to simulate nitrogen performance of the RIX facility.
2. Confirm nitrogen removal mechanisms and rates.
3. Identify one or more potential ways to improve nitrogen removal performance at RIX and to test at least one of those methods.

The scope of this investigation included construction, operation, and maintenance of a column simulated rapid infiltration system as well as sample collection, sample analysis, data analysis, and data interpretation. Nitrogen concentrations were studied in the forms of ammonium and nitrate as they proceeded through the treatment process. This entailed the collection of water samples at various depths of soil.

BACKGROUND

The information in this chapter is divided into several sections: 1) a general overview of land-based municipal wastewater treatment systems; 2) design and operation of rapid infiltration (RI) land treatment systems; 3) description of nitrogen removal mechanisms in RI systems; 4) a description and overview of the RIX facility; 5) RIX operating data and the hypothesis for the original study conducted by Kevin Bell; and 6) a summary of results from Kevin Bell and reassessment of the original hypothesis.

Land Treatment of Wastewater

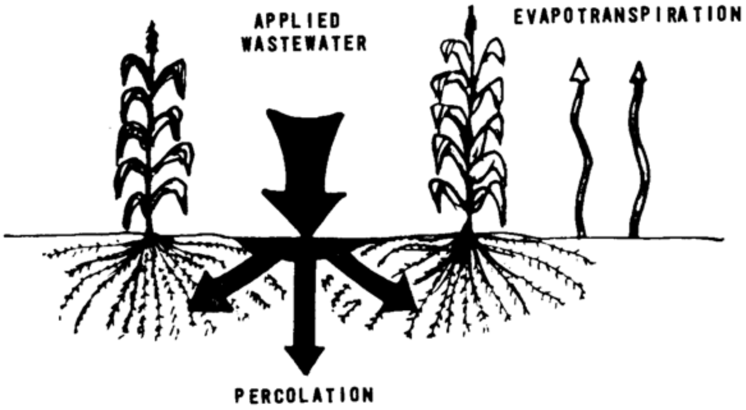
There are three distinct types of land-based systems for wastewater treatment, each defined by their own characteristic loading rates, types of soil, and operation. Schematics of typical slow rate, overland flow and rapid infiltration systems are shown in Figure 1. Typical features of the various land-treatment systems are summarized in Tables 1, 2, and 3 [13]. Slow rate and overland flow systems are only briefly described below. Rapid infiltration (RI), the land treatment process used in this study is described in detail in the next section.

When selecting a land based treatment system it is important to look at the soil conditions. The slow rate process (SR) is typically used when the soil is suitable for a percolation rate or hydraulic loading rate (volume of wastewater applied per area per unit time), of 1 to 10 cm/wk (0.25 to 4 in/wk) or moderately permeable.

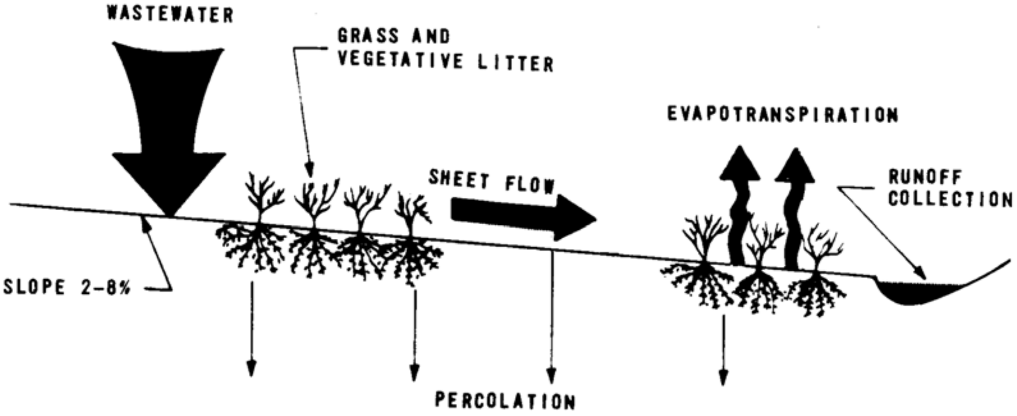
Wastewater applied to SR systems follow two hydraulic pathways. First, the wastewater can percolate through the soil until it reaches ground water. Second, it can be evaporated directly into the atmosphere from the soil surface and/or it can be taken up by plants growing in the SR system and transpired into the atmosphere. Evapotranspiration is the term associated with the combined effects of direct evaporation and transpiration [13]. Generally, SR systems are designed to primarily meet crop evapotranspiration demands and thereby minimize percolation to the groundwater.

Of the three types of land-based treatment systems, SR systems generally are able to achieve the highest level of overall nitrogen removal [12]. Although most of the water does not percolate into the groundwater, when designed for nitrogen removal, the SR hydraulic loading rate is limited by the amount of nitrate that is expected to enter the groundwater, which the EPA has set at less than 10 mg/L as nitrogen. Nitrogen is removed from the wastewater mostly by crop uptake making crop selection an important design factor [13]. The nitrogen in the wastewater is used as fertilizer by the SR crop. Ultimate removal of the nitrogen is achieved by harvesting the SR crop.

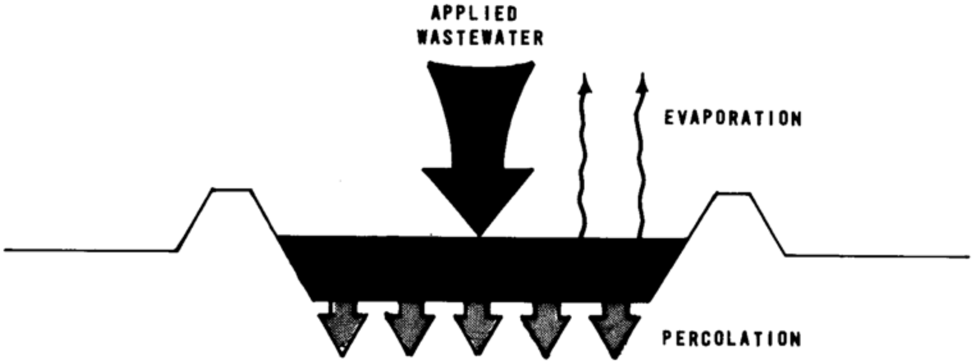
Overland flow (OF) systems are used in areas where soil has a low permeability, such as clayey soils. In OF systems wastewater is applied at the top of a grass or vegetated covered slope and allowed to flow down to a collection ditch. This process is designed for a higher loading rate of 6 to 40 cm/wk (2.5 to 16 in/wk) than that of the SR process. Like the SR process, very little of the applied wastewater percolates into the groundwater in an OF system; however, most of the wastewater runs off the end of the OF system and is not lost via evapotranspiration as the SR



(a) SLOW RATE



(b) OVERLAND FLOW



(c) RAPID INFILTRATION

Figure 1. SCHEMATICS OF LAND-BASED SYSTEMS FOR WASTEWATER TREATMENT [12]

Table 1. TYPICAL SOIL PERMEABILITY CLASSIFICATIONS FOR LAND TREATMENT PROCESSES

	Slow Rate	Rapid Infiltration	Overland Flow
Soil permeability range, cm/hr	>0.15	>5.0	<0.5
Permeability class range	Moderately slow to moderately rapid	Rapid	Slow
Textural class range	Clay loams to sandy loams	Sand and sandy loams	Clays and clay loams

*U.S. EPA 1981

Table 2. TYPICAL DESIGN FEATURES FOR LAND TREATMENT PROCESSES

	Slow Rate	Rapid Infiltration	Overland Flow
Application techniques	Sprinkler or surface	Surface	Sprinkler or surface
Annual loading rate, m	0.5-6	6-125	3-20
Field area required, ha	23-280	3-23	6.5-44
Weekly loading rate, cm	1-10	10-240	6-40
Minimum preapplication treatment	Primary sedimentation	Primary sedimentation	Grit removal and comminution
Disposition of applied wastewater	Evapotranspiration and percolation	Mainly percolation	Surface runoff and evapotranspiration with some percolation
Need for vegetation	Required	Optional	Required

*U.S. EPA 1981

Table 3. EXPECTED QUALITY OF TREATED WATER FROM LAND TREATMENT PROCESSES

	Slow Rate		Rapid Infiltration		Overland Flow	
	Average, mg/L	Upper limit, mg/L	Average, mg/L	Upper limit, mg/L	Average, mg/L	Upper limit, mg/L
BOD	<2	<5	5	<10	10	<15
Suspended solids	<1	<5	2	<5	10	<20
Ammonia nitrogen as N	<0.5	<2	0.5	<2	<4	<8
Total nitrogen as N	3	<8	10	<20	5	<10

*U.S. EPA 1981

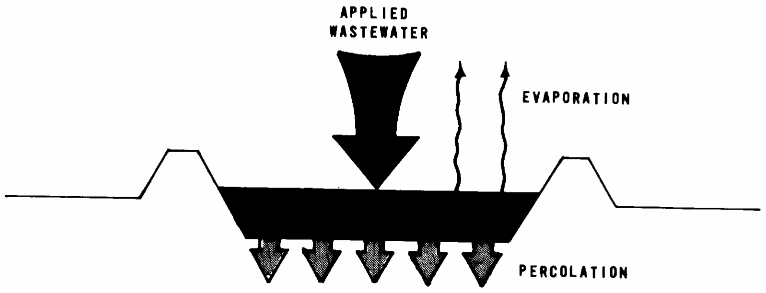
process does. This last characteristic also means that effluent wastewater from an OF system must be ultimately discharged into a surface receiving water such as a river [9].

Rapid Infiltration Process Description

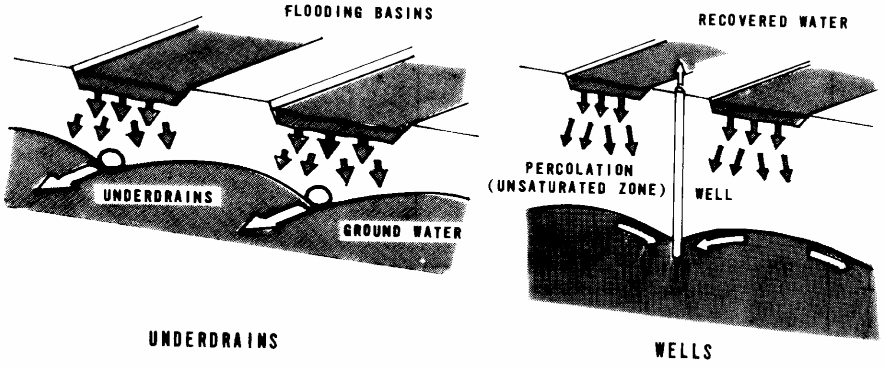
As summarized in Table 2, of the three types of land treatment systems hydraulic loading rates are highest in RI systems ranging from 10 to 240 cm/wk (4 to 95 in/wk). Highly permeable soils are required for effective treatment in the rapid infiltration system. To facilitate the infiltration process, wastewater is applied into an earthen infiltration basin where the water level is allowed to rise and thereby increase the rate of percolation through the soil matrix and into the underlying ground water. The water is either allowed to intermingle with the native aquifer or is extracted back up to the surface for additional treatment, typically disinfection, and discharged into a surface receiving water.

The major parts of RI system design include the wastewater distribution system, basin layout, storage and flow equalization, operating schedule, loading rates, cold weather modifications, and drainage. A process schematic showing the RI system hydraulic pathways is shown in Figure 2 [13].

Vegetation cover can be used in RI basins, but generally vegetation is not used so that the basin soil can be more readily scarified at regular intervals [13]. Pollutants are removed through a combination of physical, chemical and biological mechanisms.



(a) HYDRAULIC PATHWAY



UNDERDRAINS

WELLS

(b) RECOVERY PATHWAYS

(c) NATURAL DRAINAGE INTO SURFACE WATERS

Figure 2. RAPID INFILTRATION HYDRAULIC PATHWAYS [12]

RI Application Cycles and Wastewater Distribution

RI basins are operated in a cyclic manner. Alternating wetting and drying times make up an operating cycle. The wetting cycle is the time the wastewater is applied to the RI basin. Drying periods are needed to reaerate the soil and allow trapped organic matter and other contaminants to degrade aerobically between application periods. Continuous wastewater application and/or extended application periods can result in anaerobic subsurface conditions, decreased contaminant removal rates, and potentially malodorous conditions due to sulfide generation. The alternating wastewater application and drying periods constitute the RI operating cycle. Depending on the treatment objectives and influent wastewater characteristics, the length and ratio between application and drying periods varies. A summary of typical RI operating cycles is presented in Table 4 [12].

Table 4. TYPICAL RI OPERATING CYCLES

Loading cycle objective	Applied wastewater	Season	Application period, days	Drying period, days
Maximize infiltration rates	Primary	Summer	1-2	5-7
		Winter	1-2	7-12
	Secondary	Summer	1-3	4-5
		Winter	1-3	5-10
Maximize nitrogen removal	Primary	Summer	1-2	10-14
		Winter	1-2	12-16
	Secondary	Summer	7-9	10-15
		Winter	9-12	12-16
Maximize nitrification	Primary	Summer	1-2	5-7
		Winter	1-2	7-12
	Secondary	Summer	1-3	4-5
		Winter	1-3	5-10

*U.S. EPA 1981

Sprinklers can distribute wastewater in RI systems during the application period; however wastewater is more commonly applied via surface spreading in a designed infiltration basin. Infiltration basins are flat to create an even distribution of the applied wastewater. Overflow weirs are used to regulate the maximum depth of wastewater in the basin during the application cycle. Weirs serve as a means of collecting excess wastewater or diverting the overflow to another infiltration basin [13].

Basin Layout

Topography, distribution system hydraulics and the design loading rates determine the basin layout and dimensions. To allow continuous operation RI systems have multiple basins so that influent wastewater can be applied into one or more basins while the remaining basins are being allowed to drain and dry, or are under maintenance. Total area required for infiltration is also a determining factor in the number of basins needed. Infiltration basins typically range in size from 0.5 to 5 acres but basins as large as 20 acres have been designed for high flow volume systems. A higher number of smaller basin sizes allow plants to operate with a minimum of disruption when basin maintenance is required. The loading rate on the remaining operating basins is not drastically increased when one or more basins are taken offline.

Topography and soil conditions are also factors in determining the size of each basin. Equal size basins are hard to find at most RI facilities; instead basin sizes are determined by what will fit within a given area. Usually basins are rectangular in shape to utilize the space most effectively [13].

Storage and Flow Equalization

When peak flows occur, either daily or seasonally, storage is sometimes required to maintain proper operation of the RI system. Some systems are set up to use some basins as infiltration basins and others as storage. This allows for continuous flow and cycling of the infiltration basins. Storage of wastewater is rare, however, and usually is needed only if the permeability soil area is limited. Most systems are set up to always have a basin or two empty in times of normal wastewater flow, to allow for these basins to be filled when peak flows occur. In this case, the plant maintains a relatively even surface-loading rate because the empty basins take up the peak flows and the water is not stored [13].

Drainage

Rapid infiltration systems must have proper drainage to continue maximum loading rates and to allow proper aeration in order to achieve maximum treatment. Aerobic treatment processes such as nitrification will be limited if the soil is not aerated. Infiltration rates may be limited by high groundwater conditions. Thus, for RI systems to be effective there must be sufficient travel length for the wastewater to achieve the required level of treatment before entering the underlying groundwater.

Alternatively, the treated water mixed with the ground water can then be pumped up for further treatment such as disinfection. In some cases, a subsurface drainage system may be required to keep the treated water separate from the groundwater [13].

Operating Parameters

The primary operating parameters in the RI process are the application rate, application period, drying period, and hydraulic loading rate. The annual hydraulic loading rate is a function of infiltration rate and application factor. Field measurement tests used to determine the infiltration rate determine the application factor. There are 3 tests commonly used, each with a unique application factor. The basin infiltration test has an application factor of 10 to 15%, the cylinder infiltrometer and air entry permeameter measurements test is 2 to 4%, and the vertical hydraulic conductivity measurements test is 4 to 10%. The annual hydraulic loading rate (HLR_a) in ft/yr is calculated by [15]:

$$HLR_a = BIT \cdot C \cdot AF \tag{1}$$

where, BIT = basin infiltration test value, in/hr; C = conversion factor, generally 730 (ft·hr)/(in·yr); and AF = application factor as a decimal

Application rate (r_a) is determined by the annual hydraulic loading rate and loading cycle [13].

$$r_a = \frac{HLR_a}{\tau \ \cdot d_a / t_t} \tag{2}$$

where τ = application period, days; d_a = application days per year (usually 365); t_t = total time for operating cycle (wetting plus drying time), days.

The discharge rate to the basins, r_d (gal/min) can then be determined by [13]:

$$r_d = CA_b r_a \tag{3}$$

where C = conversion factor; A_b = basin area

The application period and frequency is determined by the infiltration rate, nitrogen removal, and nitrification rate as well as temperature. For maximum infiltration rates an application period of 1 to 3 days is recommended, followed by a drying period of 4 to 5 days for summer and 5 to 10 days for winter. To obtain the highest removal of nitrogen a longer application period, 7 to 9 days in summer and 9 to 12 days in winter, and longer times for drying, 10 to 15 days in summer and 12 to 16 days in winter, is recommended. Longer drying times are needed for winter because temperatures are lower and generally infiltration rates are not as high [6].

Process Performance

The rapid infiltration process is capable of removing major wastewater constituents. This process is often favored over conventional wastewater treatment plants, if land permits, because wastewater sludge is not produced. Treatment and disposal of sludge are not needed, saving on operational costs. Mechanisms for removal of the major wastewater constituents are briefly described below.

Organic Matter and Suspended Solids

Organic matter, measured as BOD, TOC, and/or COD, and suspended solids are removed initially through the physical process of filtration and later biologically by soil bacteria. Filtration of particulate BOD and suspended solids occurs at or near the soil surface. Adsorption occurs within the soil to help remove soluble BOD. Bacteria help aid in the removal of organic matter by degradation and consumption. The level of pretreatment of the wastewater does not usually affect the removal of BOD and SS. However, if high loading rates are combined with high concentrations of organic matter, the bacteria cannot consume all that is coming in. The organic matter will remain in the soil at the surface of the basin and the soil can become clogged [13].

Phosphorus

Phosphorus is removed through the combination of adsorption and precipitation in the soil. Phosphorus is adsorbed primarily by clay minerals in the soil. The soil pH and the amount of clay in the soil affect phosphorus sorption. If the soils are coarse, acidic, or highly concentrated with organic matter, the sorption of phosphorus will be less. The soil in a given area can be tested to determine the amount of phosphorus the soil can remove. The soil profile is capable of holding a finite amount of phosphorus. When the soil reaches saturation soluble phosphorus will remain in the treated water. The life of an RI facility can be estimated by the amount of phosphorus the soil can adsorb [13].

Trace Elements

The removal of trace elements in rapid infiltration involves the same processes as those for phosphorus. In addition, ion exchange and complexation contribute to the removal of trace elements from wastewater. Trace elements are removed mostly in the upper soil layers or surface soils of a rapid infiltration basin. Fine textured, organic soils have a greater adsorption capacity than sandy soils. Adsorption of most trace elements occurs on the surface of clay minerals, metal oxides, and organic matter. Sandy soils do not provide as many sites for ion exchange as do fine textured soils [13].

Trace Organics

Trace organics are removed through volatilization, sorption and degradation. Biological degradation is the primary means of removing trace organics from the wastewater, however degradation of trace organics can occur chemically as well [13].

Microorganisms

Bacteria, viruses, parasites, and worms are removed through filtration, adsorption, desiccation, radiation, predation, and contact with other harsh conditions. Large microorganisms, such as the parasites and worms are removed through filtration at the soil surface. Filtration, as well as, adsorption is responsible for removing the bacteria at the soil surface and top layer of the basin soil. Viruses are removed primarily by adsorption. They are generally too small to be filtered at the soil surface and are removed throughout the soil column as the water travels downward. Fecal coliform removal is achieved with adequate travel distance through the soil. State agencies may require additional treatment, such as UV disinfection, if the water is used for edible crops or if the public is in contact with the treated water [13].

Nitrogen Removal in the Rapid Infiltration Process

Nitrogen removal in RI systems is achieved predominantly by biological nitrification and denitrification. Nitrification is the oxidation of ammonia nitrogen to nitrite and nitrate. Denitrification is the anaerobic conversion of nitrate to nitrogen gas, which is released into the atmosphere. Native bacteria in the soil column accomplish both of these processes. Temperature, pH, organic carbon concentration, and oxygen concentration are all important parameters affecting the success of nitrifying and denitrifying bacteria. Ammonium exchange and plant uptake remove minimal amounts of nitrogen.

Nitrification

Nitrification is the oxidation of ammonium to nitrite and nitrate. There are two steps in the nitrification process, each identified by the specific chemo-autotrophic bacteria involved in the oxidation process. Several bacteria including *Nitrosomonas, Nitrosospira, Nitrosococcus, Nitrosolobus, and Nitrosovibrio* can perform the first step, the conversion of ammonium to nitrite. *Nitrobacter* are the only known bacteria that can perform the second step of nitrification, the conversion of nitrite to nitrate.

The nitrification process is shown by the following two-step reaction.

$$NH_4^+ + 3/2\ O_2 \rightarrow NO_2^- + H_2O + 2\ H^+ \quad (4)$$

$$NO_2^- + 1/2\ O_2 \rightarrow NO_3^- \quad (5)$$

The nitrification process in soil is limited by dissolved oxygen concentration, temperature, moisture, pH, and ammonium concentration. There is usually more organic carbon supply than ammonia supply and more energy comes from the organic carbon. This creates a situation where the heterotrophs grow faster or get more energy than the autotrophic "nitrifying" bacteria. In fact, the heterotrophic bacteria that feed off the organic carbon will not only grow faster than the nitrogen bacteria, but will also use up the oxygen faster. The optimum temperature range for nitrification bacteria to grow is 25-35 ^oC. Increasing moisture increases the nitrification rate, however high moisture content limits oxygen transfer to the soil, which then limits the nitrification rate. The pH range for nitrification in soil is from 6 to 8. High levels of ammonia in the soil can be toxic to *Nitrobacter* bacteria, resulting in a buildup of nitrite [2].

Denitrification

Denitrification is the reduction of nitrate and nitrite to nitrogen gas. Chemoheterotrophs, which use organic carbon as their energy source, are the most common denitrifying bacteria. Autotrophic bacteria, which grow on H_2 and CO_2, also contribute to the denitrification process. *Pseudomonas, Microccocus, Archromobacter, Thiobacillus,* and *Bacillus* are all types of bacteria that reduce nitrate according to the following reaction.

$$NO_3^- \rightarrow NO_2^- \rightarrow NO \rightarrow N_2O \rightarrow N_2 \quad (6)$$

Nitrate and oxygen concentrations are controlling factors in the above reaction. The denitrification process occurs in an anoxic environment since oxygen is more favored as an electron acceptor than nitrate. To ensure that nitrate is the electron acceptor the reaction is performed without oxygen present [2].

Another controlling factor in the denitrification process is organic carbon concentration. The amount of organic carbon required for the reaction is expressed in terms of the C/N ratio. Typical values range from 1.5 to 5 for C/N. Generally, a ratio 2 to 3 will enable complete denitrification and this value is typically used for design purposes. The range in values depends on the type of organic matter present or added to the treatment system [13]. A study by Biswas and Warnock concluded that denitrification occurs best with a 4:1 ratio of organic carbon to nitrogen [4]

Temperature, pH, and moisture content also contribute to the denitrification rate. The process can occur in a temperature range of -5^oC to 35^oC. However, at low temperatures bacteria will slow down in their growth rate and removal of nitrate. The optimum temperature is 20^oC. A pH range of 6 to 8 is suited for the denitrification process to occur. The highest rate of denitrification occurs between a pH of 7 to 7.5. High moisture content can inhibit the amount of oxygen transferred to the soil [2].

Nitrogen Removal

Coupled biological nitrification and denitrification are the primary means of nitrogen removal in rapid infiltration systems. As such, alternating between aerobic and anoxic conditions in the soil is important to achieve maximum nitrogen removal. This is achieved by having appropriate wetting and drying cycles. If wastewater does not have an adequate supply of organic carbon, longer wetting cycles are needed to promote denitrification [4].

Description and Overview of the RIX Facility

The cities of Colton and San Bernardino opened the Rapid Infiltration and Extraction (RIX) facility in March 1996. The RIX facility serves as a tertiary treatment and disinfection facility for wastewater before it is released into the Santa Ana River, which flows into Orange County. The RIX facility treats 120,000 m^3/d (30 MGD) of secondary wastewater on a 20 ha site made up of 10 infiltration ponds. The ponds are rotated on a wetting/drying cycle to allow continuous operation and maintenance. After the water has percolated through the soil it is pumped up and sent through ultraviolet disinfection, then discharged into the Santa Ana River.

RIX Operating Data and Hypothesis for Initial Study

The RIX facility was initially reviewed and studied to determine if algal uptake of nitrogen and/or ammonium sorption was a major nitrogen removal mechanism. An initial survey was conducted and found that there was not a significant amount of biomass present with respect to the nitrogen removal data (see Table 5). Assuming that only 5 mg/L of nitrogen was incorporated into biomass, there would be approximately 40 tons of biomass produced everyday with a flow of 120,000 m^3/d.

Further review of the RIX site revealed that there was a significant amount of mica and mica-like materials (vermiculite and illite). There has been little research on RI systems to demonstrate that ammonium sorption is a major nitrogen removal mechanism. However, one related study did reveal that ammonium sorption onto mica has been shown to obey first-order kinetics. Based on the data trends and soil characteristics, an initial study was conducted on the hypothesis that ammonium sorption, and subsequent fixation, is the major nitrogen removal mechanism [3].

Table 5
RIX NITROGEN OPERATING DATA

	Influent	Effluent	Removal (%)
NH_4-N	4.5-15 (9.72±3.3)	1.8-0.2 (1.02±0.46)	88 - 95
NO_3-N	2.9-11 (6.7±2.4)	2.4-4.5 (3±0.815)	17.2 - 59
NO_2-N	0.3-3.2 (1.4±0.89)	0.1-0.3 (0.64±0.33)	14.3 - 96.8
Total inorganic nitrogen	13-23 (17.7±2.73)	3.3-6.1 (4.64±1.3)	78 - 73.5

*Data averaged from June, July, and August 2001

Result of Initial Study and Reassessment of Hypothesis

To test the original hypothesis that ammonium adsorption was a primary nitrogen removal mechanism at RIX, a series of adsorption tests were performed with soils from various locations and depths at the RIX facility. Ammonia and nitrate nitrogen concentrations in both synthetic and actual wastewaters were measured before and after 24-hr of soil/wastewater mixing to determine whether adsorption was occurring.

Repeatedly, while there was a drop in the ammonia nitrogen concentration, there was a concomitant increase in nitrate nitrogen concentration. The total nitrogen concentration in solution changed very little. Thus, the main conclusion from the original study was that abiotic nitrogen removal via adsorption is not a significant nitrogen removal mechanism at the RIX facility. Rather, it appeared that biological mechanisms, namely nitrification, were responsible for the nitrogen transformations seen in these adsorption tests. Denitrification probably did not occur because of the aerobic nature of the adsorption (well-mixed) tests.

To further test this supposition, adsorption tests were conducted under sterile and non-sterile conditions, in which soil was and wastewater was sterilized either by means of an autoclave or addition of a mercury (Hg) salt, $HgCl_2$, to the wastewater such that the calculated Hg^{2+} concentration was 1 mg/L.

Typical results from these experiments are shown in Figure 3. For both sets, the non-sterilized soils/wastewaters experienced a decrease in ammonium concentration coupled with an increase in nitrate concentration and little or no change in the overall nitrogen concentration. This was observed in multiple experiments regardless of source of the soil's location or depth at the RIX facility. In contrast, transformations were not observed when sterilization was employed. This supported the revised hypothesis that ammonium removal and total nitrogen removal at RIX was biological in nature rather than abiotic via ammonium adsorption.

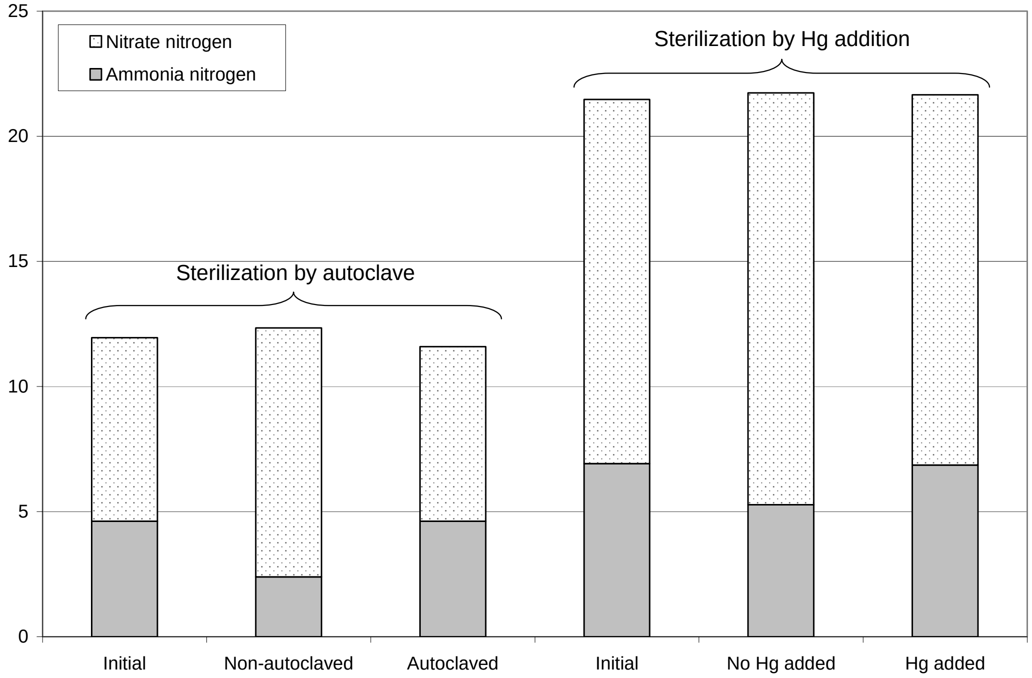


Figure 3. RESULTS (TYPICAL) FROM INITIAL STUDY [3]

FACILITY AND METHODS

This field study was conducted to confirm that biological processes, rather than abiotic sorption, are the primary nitrogen removal mechanisms at the City of San Bernardino Rapid Infiltration and Extraction (RIX) Facility. In addition, limited studies to investigate whether organic carbon addition may improve overall nitrogen removal at RIX were conducted. Detailed descriptions of this facility, pilot plant setup, sampling procedures, and water quality analyses are provided.

PILOT STUDY FACILITY

The RIX facility, designed to remove nutrients, nitrogen and phosphorus, has been operating for about eight years to treat secondary-treated wastewater from the cities of San Bernardino and Colton. The facility is located in the City of Colton and consists of 10 ponds (see Figure 4). An area located near one of the turnouts was allotted for the UCR Department of Chemical and Environmental Engineering to set up experimental columns that were fed RIX influent wastewater. The designated location allowed for convenient access to the facility’s influent water and electrical hookups.

UCR Experiment Setup

The experimental rapid infiltration system consisted of 15 soil columns, three of each depth, at depths of 2.5, 5, 10, 15, and 20 feet. The columns were constructed using eight-inch diameter PVC pipe cut into five-foot sections. Each column was started by placing end caps in fairly level soil, then adding the first five-foot section. Dry soil taken from an active pond was used to fill the columns. Soil was taken from two varying depths, 0-5 feet and 5-10 feet, out of one of the largest ponds. The 2.5 and 5-foot columns were filled with the top layer of soil, depth of 0-5 feet, and the others were filled with a bottom layer of soil, depth of 5-10 feet, up to their five-foot mark, then the top five feet were filled with the top layer of soil. The soil was packed in the columns with a six- inch plate compacter. This method was used to approximate the soil distribution found in the percolation ponds. Each column was completed with an empty five-foot section of pipe added on top of the soil column to allow the water to sit on the soil with a 2-foot head as it does in the ponds at RIX. Water was applied to the columns at the top by a pump located in the turnout (see Figure 5).

Initial Setup

The experimental setup as outlined above resulted from a series of modifications that were implemented to ensure representative sampling with depth. The study first began with three 20-ft columns. They were constructed and packed with soil in the manner described above. After construction, sampling ports were drilled along the face of the columns at depths of 2.5, 5, 10, 15, and 20 feet. The idea behind this was that as the water trickled down the length of the column, the

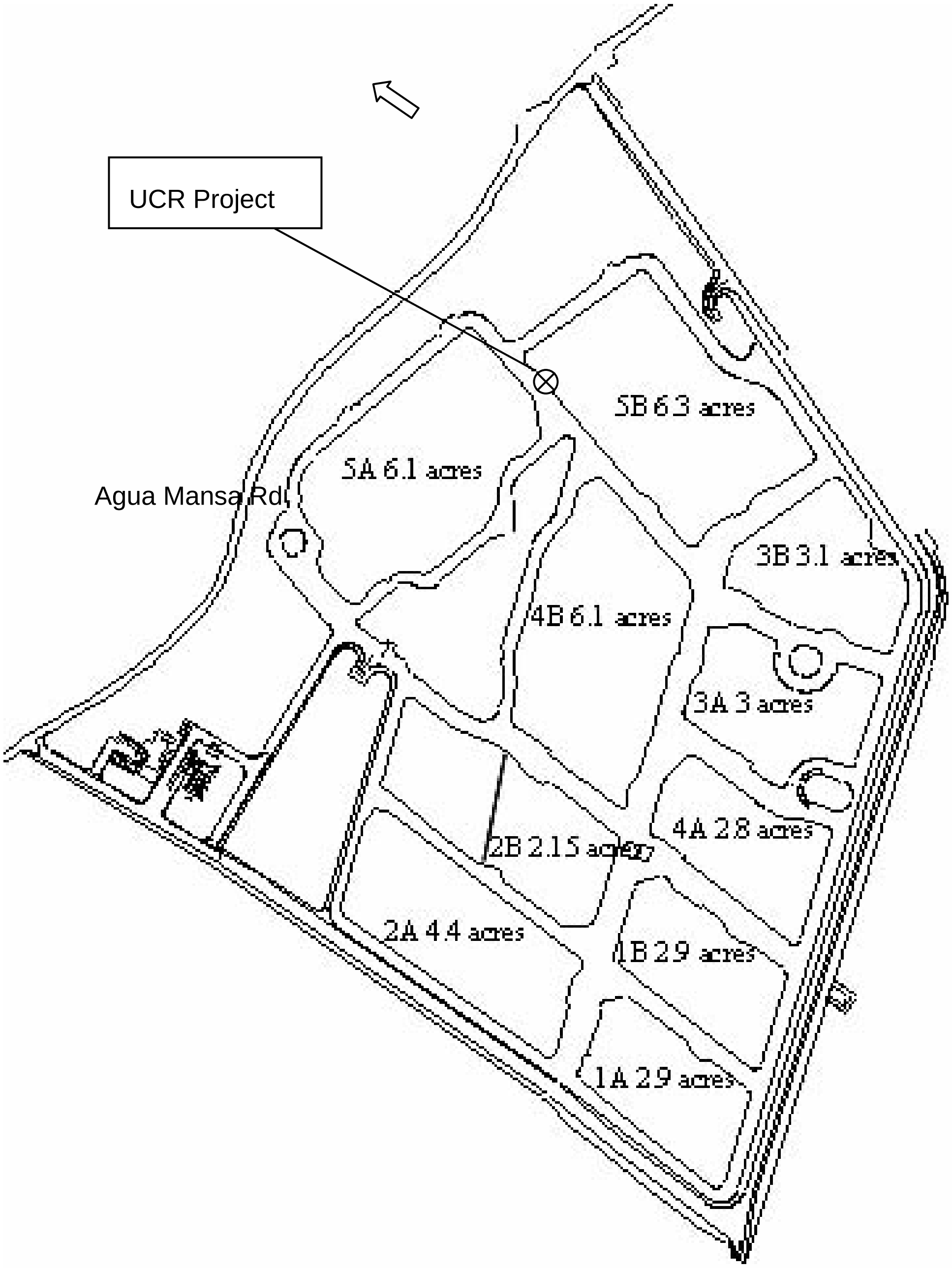


Figure 4. CITY OF SAN BERNARDINO RIX FACILITY LAYOUT

sample port could be opened and the water would be collected. The first set of sampling ports did not work in this manner. More sample ports were drilled around the perimeter of the columns at the same depths as before. This was more successful than before, but there were still some depths at which water could not be collected. It appeared as if the water was taking a distinct path through the center of the soil in the column rather than flowing uniformly through the soil.

After the columns had been drilled excessively, the experimental setup was modified.

Modification 1

To fix the problems with the sampling ports, 12 more columns were constructed, three of each height, at heights of 2.5, 5, 10, and 15 feet. The influent water was applied at the top and flowed down the length of the column. Water was collected at the bottom of each column. This method worked quite well for collecting water at various soil depths.

After running for a few weeks, the infiltration rate slowed and the columns started to back up and overflow the top of the columns. Water did not flow out the bottoms as expected.

Figure 5. WATER PIPING SYSTEM

Modification 2

To better simulate the ponds physically, an empty 5-foot section of pipe was added to each column (see Figure 6). This added section of pipe allowed for a higher static water head, 2 to 5 feet, at the top of the column instead of 3 to 4 inches. With this added head the water flowed better through the columns. The ponds are operated at a head of 2 to 5 feet. The columns were adjusted to 2 feet of head to better simulate the ponds at least amount of head.

Figure 6. MODIFICATION 2 COLUMNS IN PLACE AT RIX

FIELD STUDY PROCEDURES

The columns were constructed to simulate the ponds in a semi-controlled environment. The columns allowed for the ponds to be studied without having to ever take them offline or impact the water quality of the RIX facility while the study was conducted. To reflect RIX operation, the columns were operated on the same wetting and drying schedule, 5 days wetting and 5 days drying, as the full-scale facility. Along with the cycling, the columns were operated at the same percolation rate of 2 feet per day. This was achieved by placing flow-control emitters on the end of the sampling tubes and regular adjustments following flow measurement. The emitters are designed to allow a set flow of water through them.

System Maintenance

Daily and weekly maintenance was required to keep the columns running smoothly. Sample ports were cleaned daily to collect water samples. Algae would grow around the sample ports sometimes restricting the flow of water. The soil in the columns would also clog some of the sample ports causing the percolation rate to be affected. Emitters placed on the ends of the sample ports were replaced when the membrane broke or became clogged. New sample ports were drilled and the old ones closed off if they became irreversibly clogged.

After many months of continuous operation, the columns became clogged at the top. Clogging occurred because of a buildup of biomass and algae sitting and growing on the surface of the soil. The percolation rate slowly declined and eventually diminished. This problem was addressed by scraping the surface of the soil to loosen the biomass. When the columns were filled with water the biomass floated and was easily removed. This cleaning procedure took place at the beginning of every application period to prevent future buildups.

Sampling

Water samples were collected from the bottom of each column in Nalgene 125-mL plastic bottles (see Figure 7). Initially samples were collected every day during the wetting period. When the data were shown to be consistent for many cycles, the sampling days were cut back to every other day. When samples were collected, the pH was measured immediately with a portable Oakton Waterproof pHTestr BNC. The flow rates or percolation rate of each column was measured once per cycle while samples were being collected. When all samples were collected, they were transported immediately to the UCR laboratory for analyses.

Figure 6. SAMPLE COLLECTION

Laboratory Analyses

Collected samples were analyzed each day they were taken for NO_3^--N, NH_4^+-N, TOC, TIC, pH, and alkalinity. The nitrogen species, TOC, and TIC were measured in the UCR laboratory. The pH was measured in the field. Alkalinity was determined from TIC results.

NO_3^--N and NH_4^+-N analysis was performed using the Dionex DX-120 ion chromatograph (IC) and Accumet nitrate and ammonium ion selective electrodes. The Dionex IC required samples to be filtered to 0.1 micron to prevent organic buildup in the column.

The TOC and TIC analysis was performed using the Shimadzu TOC 5050 total organic carbon analyzer. Samples were filtered to 0.45 micron before they were tested.

EXPERIMENTAL STUDIES

This experiment consisted of two studies. The first study was performed under the RIX influent wastewater as received with no modification. Following the first study, a second study was performed to investigate whether overall nitrogen removal could be improved through the addition of supplemental organic carbon in the form of a methanol solution to the RIX influent wastewater.

Nitrogen Removal Study

The purpose of this study was to evaluate whether nitrification and denitrification occur within the subsurface at RIX and to describe the kinetics of the nitrogen removal process. For this study, ammonium, nitrate, TOC concentration, pH, and alkalinity were observed as a function of column length. Circumstantial confirmation of nitrification-denitrification was determined by comparing changes in these parameters with expected trends that are known to occur when nitrification and denitrification take place.

Enhanced Nitrogen Removal Via Organic Carbon Amendment Study

After the first study was complete, the TOC concentration of the influent RIX wastewater was adjusted to determine the effects on nitrogen removal. Methanol was added to the influent wastewater of one of the three columns for each height to double the concentration of influent organic carbon. The other two columns were operated without any modification for control. Comparison of the nitrogen removal trends of the two column sets was used to assess the impact of increased organic carbon loading.

RESULTS AND DISCUSSION

The results of a 12-month column study on nitrogen removal and organic carbon spiking are described in this section. As a prelude to the presentation of the actual results generated during this study, a description of expected system behavior as a result of nitrification-denitrification nitrogen removal is presented. Results of the actual system performance to those expected provide circumstantial evidence supporting the hypothesis that nitrification-denitrification is the primary nitrogen removal process at the RIX facility.

Water Quality Changes Resulting from Nitrification-Denitrification

Nitrification

Biologically, nitrification is favored when the availability of readily biodegradable organics is low. Bacteria generally obtain a greater amount of energy when oxidizing organic carbon compared to reduced nitrogen forms such as ammonium ion. In a competitive environment heterotrophic organic carbon utilizing bacteria out compete nitrifying bacteria for oxygen resources and nitrification does not occur. In order for nitrification to proceed at an acceptable rate, the organic carbon to ammonium nitrogen concentration mass ratio must be relatively low, typically below 1.

In a RI system, if the influent wastewater contains a high organic carbon concentration, relative to the ammonia nitrogen, nitrification will not occur until the organic carbon has been consumed. Thus, nitrification may not occur near the soil surface, but may occur farther below the surface.

The overall reaction for biologically-mediated nitrification is:

$$NH_4^+ + 2O_2 \xrightarrow{bacteria} NO_3^- + H_2O + 2H^+ \qquad (8)$$

For every mole of ammonium converted to nitrate, two moles of hydrogen ion are produced. Conversion of 14 mg/L of ammonia nitrogen could potentially result in the generation of 2 x 10^{-3} mole of H^+ and a significant pH drop. Fortunately, most wastewaters are buffered by alkalinity, primarily bicarbonate ions. Excess H^+ ions are neutralized by the following reaction:

$$HCO_3^- + H^+ \rightarrow H_2CO_3 \rightarrow H_2O + CO_2 \uparrow \qquad (9)$$

Bicarbonate combines with H^+ to form carbonic acid, which in turn is released as carbon dioxide. Conversion of 14 mg/L of ammonia nitrogen could result in the consumption of up to 2 meq/L of alkalinity, or 100 mg/L as $CaCO_3$.

If biological nitrification is indeed occurring at the RIX facility, ammonia nitrogen reductions should be accompanied by a slight decrease in pH and a loss of alkalinity.

Denitrification

Unlike nitrification, which is an aerobic process, denitrification is an anoxic biological process. In the absence of oxygen, denitrifying bacteria utilize nitrate as the electron acceptor and organic carbon as the electron donor to derive energy. The overall stoichiometry of the reaction depends on the organic carbon constituent present. Using methanol as the example organic carbon source, the overall stoichiometric energy reaction is:

$$6NO_3^- + 5CH_3OH \xrightarrow{bacteria} 5CO_2 + 3N_2 \uparrow + 7H_2O + 6OH^- \qquad (10)$$

Based on this reaction, the stoichiometric amount of organic carbon needed per mg of nitrate nitrogen converted to nitrogen gas would be a minimum of 0.71 mg. In addition to organic carbon being consumed, one mole OH^- is generated per mole of nitrate nitrogen reduced. The generation of OH^- could potentially result in a significant increase of pH. Fortunately, as in the nitrification process, the presence of alkalinity (and acidity) by the carbonate species will moderate any pH changes.

$$\begin{aligned} &HCO_3^- + OH^- \rightarrow H_2O + CO_3^{-2} \\ &CO_2(aq) + OH^- \rightarrow HCO_3^- \end{aligned} \qquad (11)$$

Referring again to the denitrification energy reaction with methanol, additional carbon is needed for new cell (denitrifiers) growth and, in cases where dissolved oxygen is present, organic carbon is used by other heterotrophic bacteria to consume the dissolved oxygen. Accounting for all of these demands for organic carbon, an empirical relationship has been suggested to estimate the methanol requirements for denitrification [11].

$$C_m = 2.47N_0 + 0.87D_0 \qquad (12)$$

where
C_m = required methanol conc., mg/L
N_0 = initial nitrate nitrogen concentration, mg/L
D_0 = initial dissolved oxygen concentration, mg/L

For a typical secondary wastewater containing 10 mg/L of nitrate nitrogen and 5 mg/L of dissolved oxygen, the minimum methanol requirement would be 29.1 mg/L, which is equivalent to 10.9 mg/L TOC or a TOC:NO_3^--N mass ratio of 1.1. Typically, a higher TOC:NO_3^--N mass ratio is used to ensure anoxic conditions are maintained for denitrification.

If biological denitrification is indeed occurring at the RIX facility, nitrate nitrogen reductions should be accompanied by a decrease in organic carbon, a slight increase in pH, and a potential loss/gain of alkalinity.

Co-current Nitrification-Denitrification

An interesting aspect of RI systems and subsurface flow is the fact that the soil column can have both aerobic and anoxic areas within close proximity. Both nitrification and denitrification can occur simultaneously (in different environments), not sequentially, as the wastewater passes through the soil column. Even though denitrification may occur, the nitrate concentration may increase due to nitrification.

To understand the impact of these mechanisms concurrently, a theoretical modeling exercise is informative. Using conditions similar to that at RIX, in which the influent ammonia nitrogen concentration is about 17 mg/L and the influent nitrate nitrogen concentration is 14 mg/L and assuming single-constituent Monod-type kinetics for both ammonium and nitrate removal, ammonium, nitrate, and total nitrogen removal as a function of soil depth can be modeled.

$$\begin{aligned} r_{NH_4-N} &= \frac{k_{NH_4-N} C_{NH_4-N}}{K_{S,NH_4-N} + C_{NH_4-N}} \\ r_{NO_3-N} &= \frac{k_{NO_3-N} C_{NO_3-N}}{K_{S,NO_3-N} + C_{NO_3-N}} \end{aligned} \qquad (13)$$

where
r_i = rate of removal for constituent i, mass/(volume $\cdot$ time)
k_i = rate constant for constituent i, mass/(volume $\cdot$ time)
$K_{S,i}$ = half velocity constant for constituent i, mass/volume
C_i = concentration of constituent i

and the differential equations are:

$$\begin{aligned} \frac{dC_{NH_4-N}}{dt} &= -r_{NH_4-N} \\ \frac{dC_{NO_3-N}}{dt} &= r_{NH_4-N} - r_{NO_3-N} \end{aligned} \qquad (14)$$

Typical kinetic coefficients for nitrification and denitrification can be found in reference [11]. Using these values a number of different scenarios are possible, depending on the relative nitrification and denitrification rates. Modeled results using the above kinetic models are shown in Figure 7. For this simulation, wastewater is assumed to move through the soil column at a constant velocity and that reaction time is a function of the depth of wastewater travel only.

As shown in Figure 7, assuming favorable conditions for nitrification exist, the ammonia nitrogen concentration will decrease as the wastewater percolates down the soil column. The nitrification process generates additional nitrate. Thus, depending on the relative rates of nitrification and denitrification, the nitrate concentration may or may not decrease. In Figure 7a, an increase in nitrate

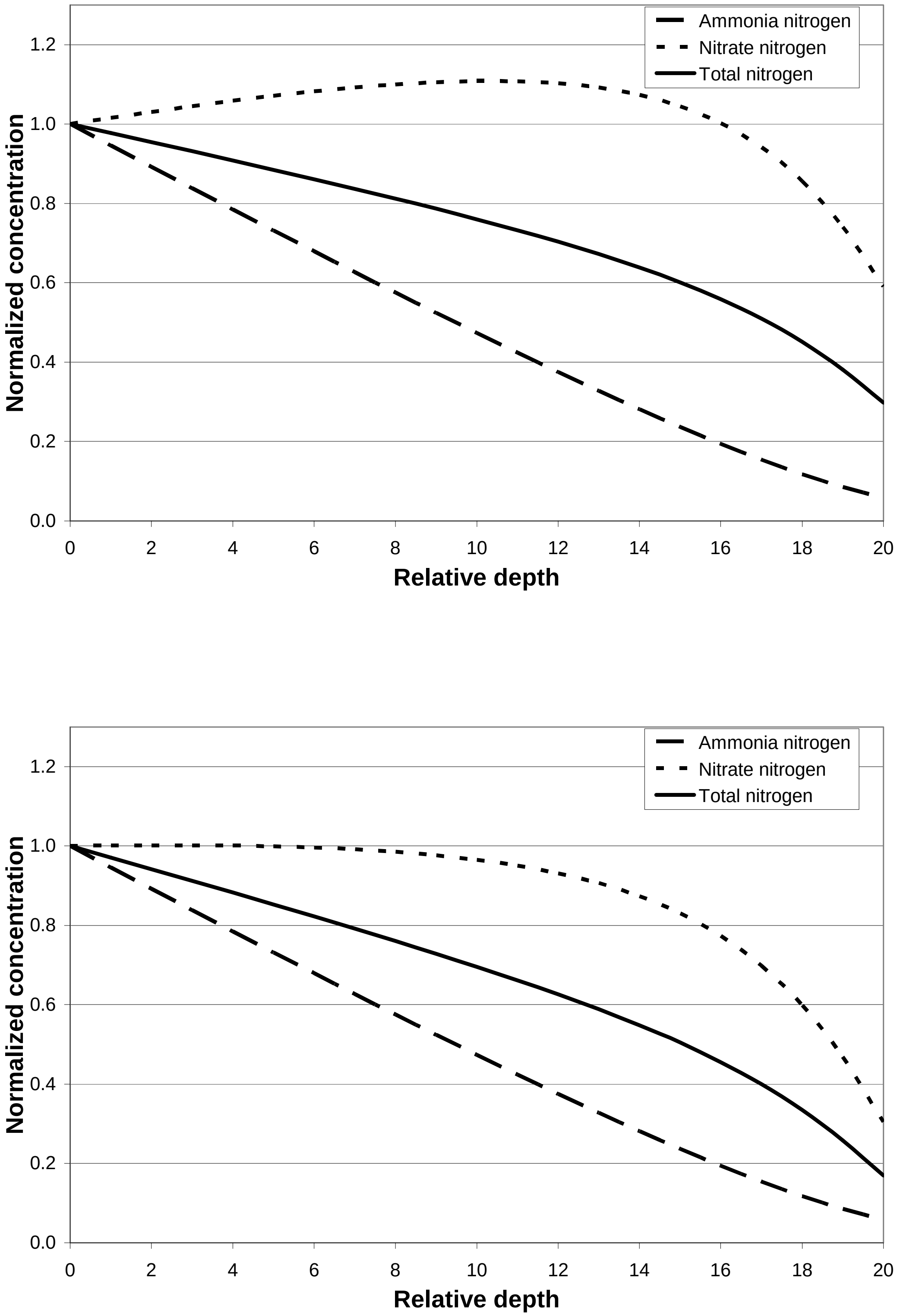


Figure 7. TYPICAL RESULTS FOR TWO DIFFERENT RATES OF NITRIFICATION-DENITRIFICATION

concentration is observed for some distance down the soil column, and then decreases rapidly once the source of nitrate from nitrification is nearly depleted. In Figure 7b, there is no decrease of nitrate initially down the column. The nitrate concentration is relatively level until the available ammonia nitrogen is nearly depleted. Thereafter, the nitrate concentration decreases.These scenarios are merely illustrative of many different possibilities. Environmental conditions vary within the soil column and the rates of nitrification and denitrification may increase or decrease depending on factors such as pH, organic carbon availability, and dissolved oxygen concentrations.

COLUMN STUDIES AT RIX

Experimental studies were conducted to determine the relationship between nitrogen removal and percolation depth in the Rapid Infiltration and Extraction (RIX) facility in Colton, CA. The study was also designed to confirm nitrification-denitrification as the primary mechanism for nitrogen removal at the RIX facility. The main investigation involved columns of increasing depth filled with soil from the RIX facility that were operated to mimic field operation. In addition, a short-term organic carbon spiking experiment was conducted to see whether low concentrations of TOC in the RIX influent wastewater are limiting the overall nitrogen removal.

In a typical laboratory study using a continuous-flow system with a well-characterized and synthetically-derived surrogate wastewater, system performance can be assessed from relatively few measurements after the system has reached a quasi-steady state. However, assessing performance of the RIX column studies presents inherent challenges due to the semi-controlled nature of the experiments and the cyclic operation of RI systems. The changing nature (hourly, daily) of the influent characteristics, field temperature, heterogeneity of the columns, random nature of flow patterns within the soil, wall effects, relatively long residence times within the column, and intermittent nature of the treatment cycle all contribute to variability in the performance measurements.

The average flow velocity through the RI basins toward the groundwater is approximately 10 ft/d (5 cm/hr). Average residence times for 2.5, 5, 10, 15, and 20-ft columns are approximately 6, 12, 24, 36, and 48 hours, respectively. A single set of grab samples taken at the same time from the various length columns (or even from the same column) may or may not be representative of treatment as a function of depth. Representative performance for RI system is best represented by analysis of composite samples collected over the duration of the application cycles and the averaging of performance from numerous cycles over periods of years.

For this study, however, composite sampling over 10 days for 15 separate columns was not possible with the available resources nor was sampling over several years possible. Therefore, as an alternative, grab samples of influent and effluent flows at various times during each application cycle were collected and analyzed. The results from these analyses were averaged together to represent

Table 6. SUMMARY OF RESULTS FROM COLUMN STUDIES FOR 8 CYCLES

Parameter		Column height, ft				
		0 (influent)	2.5	5	10	15
pH, pH units	Average	7.39	7.29	7.23	7.11	7.01
	Minimum	6.99	6.67	6.46	6.62	6.50
	Maximum	8.30	7.66	8.13	7.56	7.54
	Std deviation	0.22	0.23	0.25	0.21	0.20
TOC, mg/L	Average	15.0	8.9	9.4	7.6	7.5
	Minimum	8.2	0.4	1.2	0.1	1.2
	Maximum	27.5	23.4	24.3	19.7	18.7
	Std deviation	5.1	4.7	5.2	4.0	3.6
NH_4^+-N, mg/L as N	Average	23.6	18.3	17.0	12.2	8.0
	Minimum	15.1	7.7	4.6	1.0	2.0
	Maximum	35.8	34.4	31.3	28.3	17.7
	Std deviation	5.5	6.2	6.4	5.4	3.6
NO_3^--N, mg/L as N	Average	14.0	9.5	9.3	9.1	8.8
	Minimum	6.4	0.4	0.5	0.3	0.4
	Maximum	31.3	30.0	38.1	34.0	28.8
	Std deviation	6.2	5.9	7.1	6.7	5.3
NH_4^+-N + NO_3^--N	Average	38.3	27.6	25.3	21.3	16.8
	Minimum	26.4	12.1	10.1	9.6	9.3
	Maximum	56.0	45.3	38.5	37.9	23.8
	Std deviation	7.5	7.5	7.0	6.5	4.2
Alkalinity, mg/L as $CaCO_3$	Average	252	262	279	279	272
	Minimum	131	99	134	152	147
	Maximum	419	466	452	572	626
	Std deviation	56	70	66	80	88

performance for a single application cycle. Further, to address random variability due to soil and column construction, results from the three columns of similar length were averaged together. As shown in Table 4-1, a considerable range of variability in the constituent concentrations was observed at each depth. Due to the lack of available pump head, flow to the top of the 20-ft columns was intermittent as well as the resultant data, which was deemed to be unreliable. The results from the 20-ft columns are not included. The concentrations of ammonium and nitrate averaged over twelve continuous cycles as a function of depth are plotted in Figure 8. A similar plot for TOC is shown in Figure 9. Alkalinity and pH plots are presented in Figure 10.

As seen in Figure 8, average ammonium and nitrate concentrations were found to decrease as a function of depth of soil. The concentration of ammonium decreased fairly consistently as a function of column length, typical of zero-order removal kinetics. Assuming Monod kinetics (Eqn 1), this type of behavior would occur when the half-velocity constant, K_S, is relatively small, below 1 mg/L. The observed zero-order rate constant for nitritification is approximately 1 mg L^{-1} ft^{-1}.

Nitrate concentrations decreased primarily in the first 2.5 feet of soil and slowly, or not at all, thereafter. However, as noted previously, absence of nitrate concentration decrease is not an indication that denitrification is non-occurring. Ammonium is converted to nitrate during nitrification. Thus, nitrate concentration would increase if denitrification was not occurring. Rather, because the nitrate concentration is relatively constant, it appears as though the rate of denitrification is similar to that of the nitrification, namely 1 mg L^{-1} ft^{-1}.

As reported in the literature by Crites and Reed, nitrification and denitrification mostly occurs within the first 18 inches of soil depth in RI systems [6]. The results from this study indicate that both nitrification and denitrification occur throughout the soil column at depths greater than 1.5 feet. In this respect, the RIX facility may or may not be unique, but these trends are contrary to what is reported in the literature.

In reviewing the removal of TOC as a function of depth (see Figure 10), the rate of TOC removal is greatest in the first 2.5 ft of the soil column, decreasing about 6 mg/L, from 15 to 9 mg/L. After the first 2.5 feet, TOC removal is relatively slow, decreasing only about 1.5 mg/L in the next 12.5 feet of soil.

TOC removal is very similar to the nitrate removal, which is fastest in the first 2.5 feet and slower thereafter. What is interesting to observe is the amount of nitrogen removed per mass of TOC removed. Based on Figure 4-2, the amount of nitrogen removed in the first 2.5 feet of soil column, presumably via denitrification within anoxic regions of the soil column, averages a little more than 10 mg/L as N. Over that same 2.5 feet of soil column, the decrease of TOC is about 6 mg/L. Thus, the removed TOC to removed nitrate nitrogen mass ratio is less than 0.6. This ratio is below the estimated stoichiometric requirement of 0.7 to 2 outlined earlier in this chapter.

After 2.5 feet, the imbalance becomes worse. From 2.5 to 15 feet, the amount of nitrogen removed is a little more than 10 mg/L and the TOC concentration decreases on average about 1.5 mg/L. The TOC:N removal ratio is less than 0.2. Based on the reported TOC requirements for denitrification, it is not evident as to how the energy and carbon needs of the denitrifying bacteria are being met within the soil column.

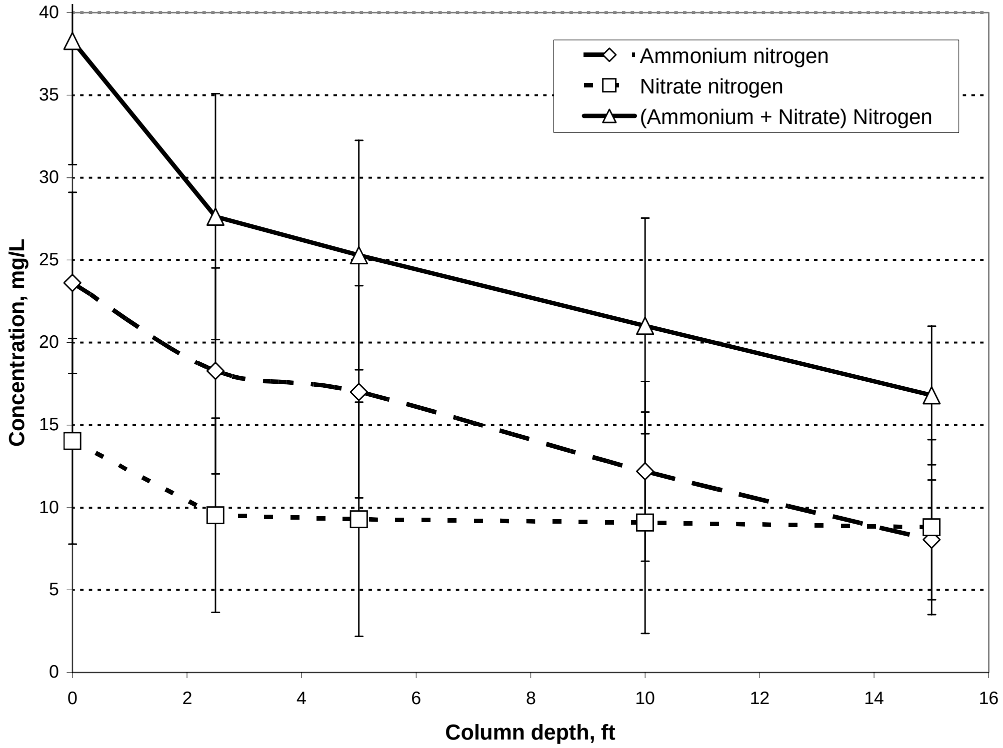


Figure 8. AVERAGED AMMONIUM AND NITRATE CONCENTRATIONS

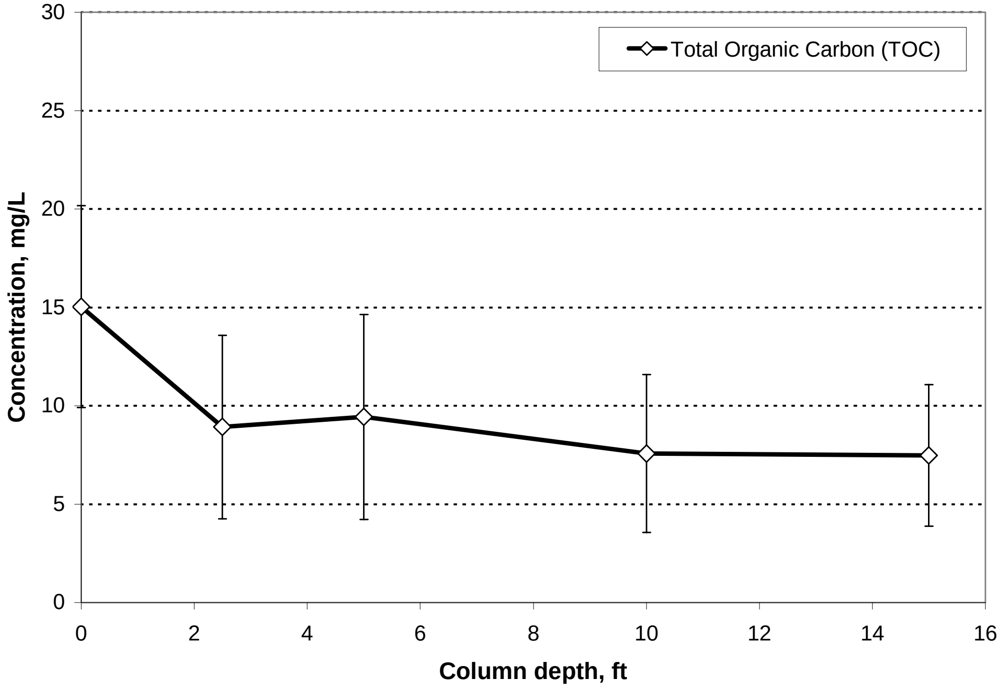


Figure 9. AVERAGED TOTAL ORGANIC CARBON CONCENTATION

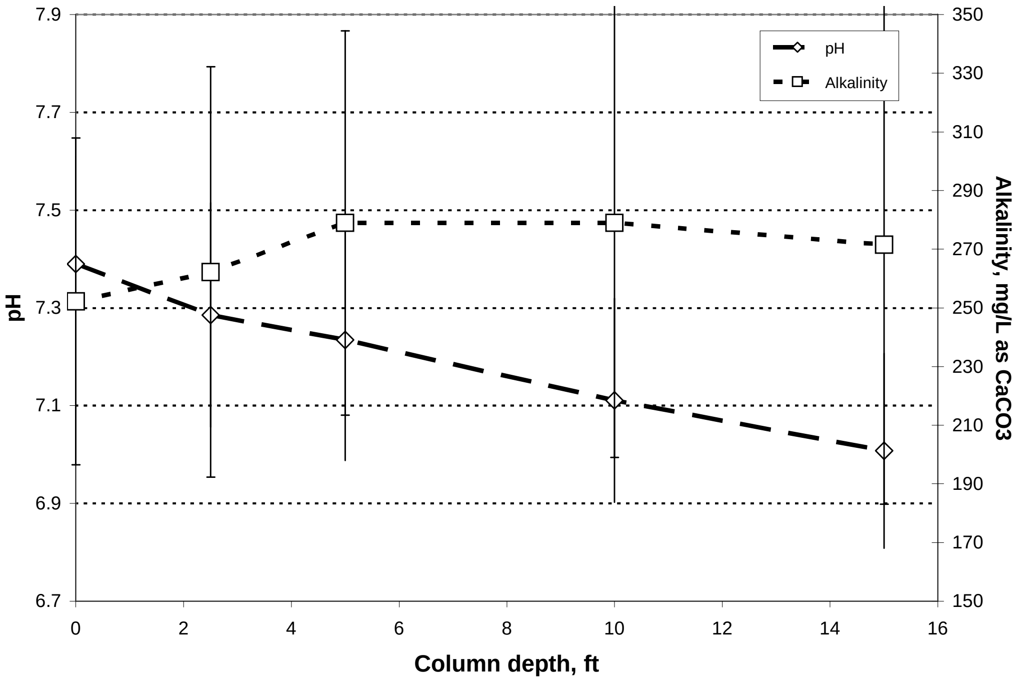


Figure 10. AVERAGED PH AND ALKALINITY

Although this mechanism is not known, it does appear that when TOC is available that the denitrification (and nitrogen removal) is enhanced. As noted, in the first 2.5 feet of the soil column, the rate of nitrogen removal and TOC removal mirrored one another.

Organic Carbon Spiking Studies

A controlling factor in most denitrification processes is the availability of organic carbon. Denitrifying bacteria use organic carbon as their energy source; nitrate serves as the electron acceptor. Based on the observations during the column in which the TOC:N ratio was noted to be less than that specified in the literature, the addition of organic carbon was studied to see whether nitrate removal and overall nitrogen removal was enhanced.

Methanol was added to the influent water of one column of each height. The two remaining columns of each height were operated under normal conditions (no methanol added) to use as comparison. A system was set up to double the concentration of organic carbon in the influent wastewater. A concentrated methanol solution was prepared ranging from 5000 mg/L to 10000 mg/L so that when added to the normal influent flow it made up a relatively small proportion of the total flow. This solution was pumped slowly using a peristaltic pump into the influent wastewater at the top of each column.

The concentrations of ammonium and nitrate averaged over three cycles as a function of depth for columns with and without methanol added are plotted in Figure 11. A similar plot for TOC is shown in Figure 12. Alkalinity and pH plots are presented in Figure 13.

In comparing the results in Figure 11 between the columns with and without added methanol, there appears to be little difference in the nitrate concentration decrease throughout the length of soil. However, with respect to ammonia nitrogen reduction, the rate of decrease accelerates between 2.5 and 10 feet with the addition of methanol. During the first 2.5 feet there is little or no difference in the rate.

This trend seems contrary to the expectation that methanol is being added to improve denitrification. The nitrate concentration trends are relatively unchanged with the additional of methanol; however the nitrification rate increases. The addition of methanol was not expected to impact the nitrification rate.

While the denitrification rate does not appear to increase on the basis of the nitrate concentration results, it in fact is increasing. The ammonium is transformed by the nitrification process into nitrate, which is in turn removed via denitrification. When the denitrification rate matches the nitrification rate, the nitrate concentration will remain relatively constant. Based on the similar nitrate concentrations observed between the columns with and without methanol, the nitrification and denitrification rates appear to be similar throughout the column. Thus, the addition of methanol seems to increase both the nitrification and denitrification rate, at least in a portion of the column.

In looking further at Figure 11 the ammonia nitrogen and combined nitrogen reduction rate decreases significantly after 10 ft in the methanol added columns, whereas the removal rate in the non-methanol columns remains constant. After 15 feet, the overall nitrogen concentration is lower in the methanol added column. But if the same trends were to occur to 20 feet, the concentrations would be nearly the same. Thus, it is not clear that the addition of methanol will have a long-term benefit on denitrification and nitrogen removal.

The additional TOC associated with the methanol appears to be depleted after 5 feet (see Figure 12). Thereafter, the TOC concentrations with and without methanol addition are nearly identical. On the basis of these results, it would seem that the benefits of the methanol addition would occur within the first 5 feet of soil depth. However, looking at the nitrogen results in Figure 11, the impact seems to occur after the 5-ft level, not before.

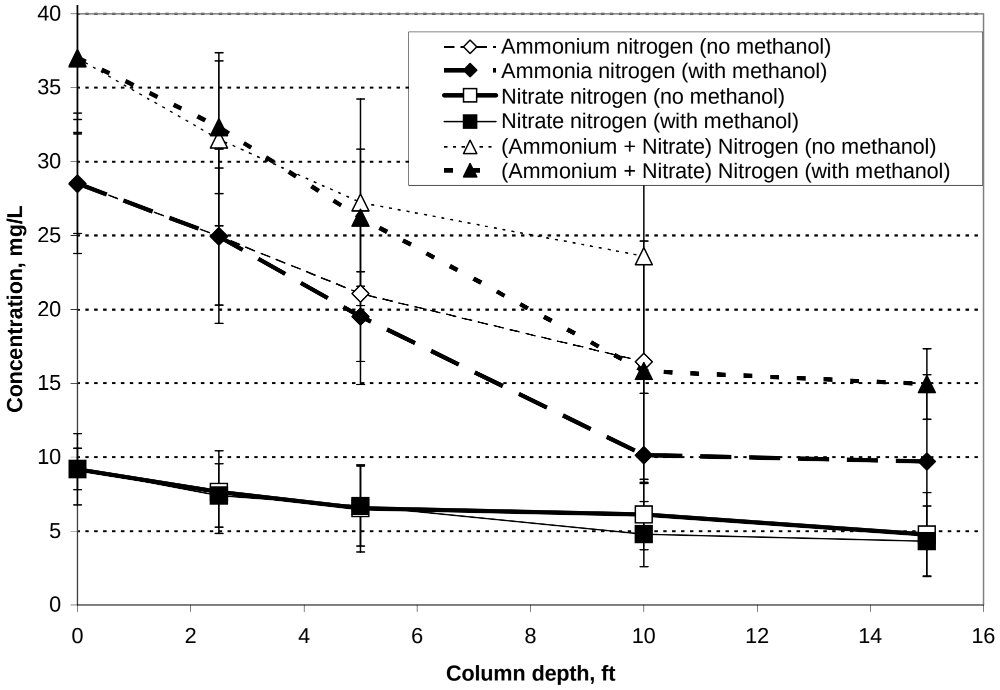


Figure 11. AVERAGED AMMONIUM AND NITRATE CONCENTRATIONS WITH AND WITHOUT METHANOL

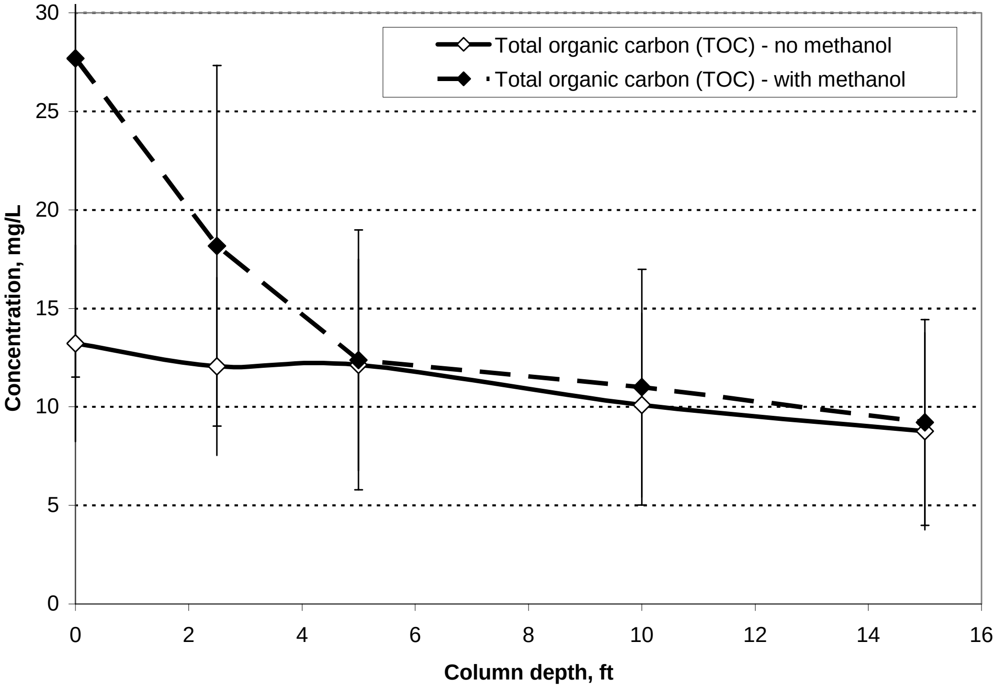


Figure 12. AVERAGED TOTAL ORGANIC CARBON CONCENTRATION WITH AND WITHOUT METHANOL

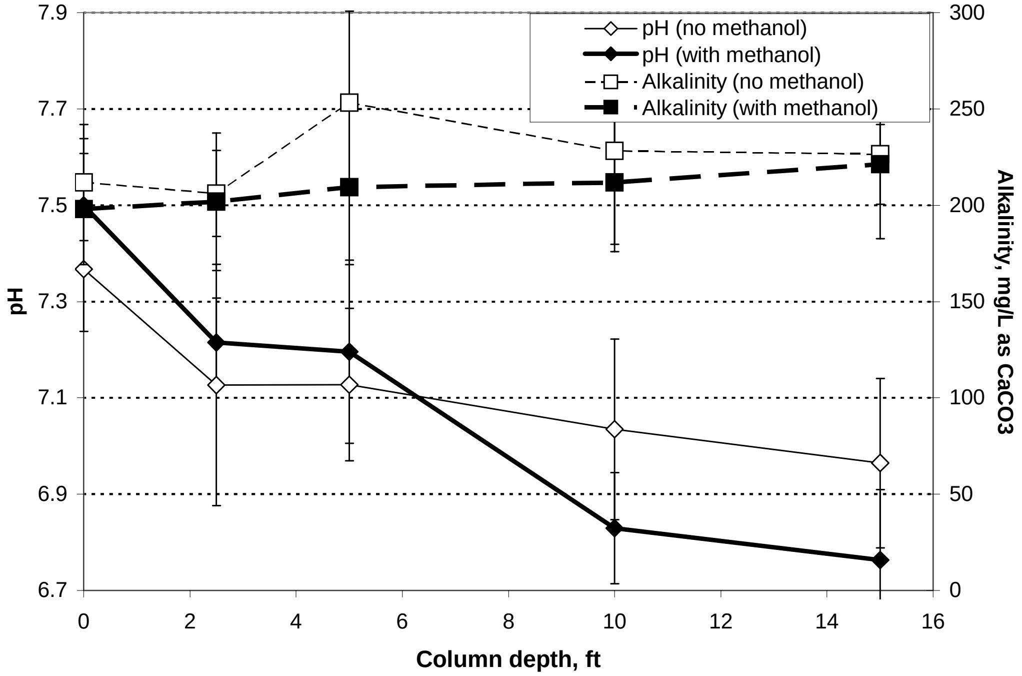


Figure 13. AVERAGED PH AND ALKALINITY WITH AND WITHOUT METHANOL

It is not clear why this phenomenon occurs, but it is speculated that one factor may have to do with the dissolved oxygen conditions. The higher TOC concentrations and removal associated with the methanol addition may decrease the dissolved oxygen levels in the wastewater flowing through the soil favoring increased denitrification. Near the top of the soil column, the incoming dissolved oxygen in the wastewater influences the denitrification rate more. However, as the wastewater flows down the soil column, the dissolved oxygen is depleted at a faster rate with higher TOC concentrations.

The results from this study provide some evidence that additional organic carbon in the RIX influent water may improve the denitrification rate when compared to influent water without added organic carbon within some portions of the soil column. However, the overall impact on nitrogen removal will most likely depend on the specific RI facility.

CONCLUSIONS

Based on the column study conducted at the RIX site in Colton, California over a period of 6 months, it is apparent that nitrogen removal in the rapid infiltration process is primarily occurring through biological activity, namely nitrification and denitrification, and not ammonium adsorption as originally speculated. The previous study at UCR concluded that there was no observable adsorption of ammonium by the RIX soil under abiotic conditions and led to the development of a study of biological removal of nitrogen via nitrification-denitrification at the RIX site.

Based on the results of this study, nitrification and denitrification take place throughout the 20-foot depth of soil in the column. Nitrification, the biological conversion of ammonium to nitrate, follows a zero-order mechanism throughout the first 15 feet of soil at the RIX facility with a rate constant equal to approximately -1 mg L^{-1} ft^{-1}. The denitrification rate is greatest in the upper 2.5 ft of soil column, where the TOC degradation rate is also highest. Thereafter, the denitrification rate is similar in magnitude to the nitrification rate beyond that depth. The rate of TOC removal beyond 2.5 ft is relatively low.

The denitrification process at RIX appears to be operating with TOC:NO_3-N ratios well below those recommended and/or theoretically derived and reported in the literature. Calculated ratios at RIX from this study range from 0.2 to 0.6, whereas the recommended/theoretical literature values are in the range of 0.7 to 2. The reason for this phenomenon is unknown at this time and may be an appropriate focus for a future study.

The results from this study provide evidence that additional organic carbon in the RIX influent water may improve the denitrification rate within some portions of the soil column. However, it was not apparent that the overall nitrogen removal will improve over extended soil depths.

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