# NUTRIENT AND PATHOGEN CONTRIBUTIONS TO SURFACE AND SUBSURFACE WATERS FROM ON-SITE WASTEWATER SYSTEMS - A REVIEW

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## Abstract

A review of studies conducted in the past decades on nutrient and pathogen contributions to surface and subsurface waters is presented. Factors such as soil structure, water table levels, dissolved oxygen, organic matter content, and cation content in soils determine the movement of nutrient and pathogens in soils. Monitoring studies agree, in general, on the influence that these factors have on nutrient and pathogen movement in soils. Generally, data collected from monitoring wells show that nutrients and pathogens decrease in concentration for wells located farther away from the drainfield. Similar trends are found for wells placed deeper in the soil. Several studies that monitored nitrogen (N) removal from On-Site Wastewater Systems (OSWS) explained that N adsorption onto soils, dilution in the groundwater, and denitrification act as the main paths of N removal. Some studies found that little nitrification took place in nearly saturated soils. On the other hand, significant nitrate ($NO_3^-$) movement away from the drainfield was observed from OSWS placed in unsaturated soils with low organic matter content. However, $NO_3^-$ moving over long distances towards surface water was observed to decrease when the plume encountered sediments with high organic matter content. Studies that monitored phosphorus (P) removal from OSWS documented high P removal in close distance from the drainfield. Precipitation of phosphorus with aluminum ($Al^{3+}$), iron ($Fe^{3+}$) and calcium ($Ca^{2+}$) is the main path for P removal. However, dissolution of acid soil P precipitates is likely when saturated conditions are present. Nevertheless, P was not found in high concentrations in soils. Several studies concluded that filtration, adsorption, and die-off mechanisms are the main removal path of pathogens in soils. While large pathogens are removed mainly through filtration bacteria and viruses are removed through adsorption and die-off. High saturation levels, low cation content in soils, and rainfall may hinder bacterial and viral adsorption. Differences in behavior were documented between bacteria and viruses under these conditions. Studies that monitored pathogen removal near the drainfield documented that low temperatures enhanced survival of both bacteria and viruses. Furthermore, viral survival was lower than bacterial survival under summer temperature conditions. In addition, removal of both viruses and bacteria was more efficient under lower loading rates and unsaturated conditions. Several studies that monitored bacterial movement away from the drainfield confirmed that bacterial survival is enhanced by saturated conditions, rainfall events, high saturated conductivity, and low temperatures, However, for longer distances away from the drainfield relationships of viral survival with rainfall, low temperatures, and bacterial survival were not well established.


## Introduction

Water quality problems are common in the surface waters and groundwater of the North Carolina. North Carolinians often hear about algae thriving in estuaries and rivers, fish kills, and excess nutrients in our rivers. Water quality problems in North Carolina originate from numerous sources, both point and non-point. Point sources are those such as pipes discharging to streams and surface runoff. Non-point sources of pollution in the waters of the State are diffuse sources such as pesticides or nutrients that reach surface waters via runoff after their application in farm fields or sediments from construction sites that end up in rivers and lakes. Non-point source (NPS) pollution is a significant source of contamination.

On-site wastewater systems such as septic systems contribute to NPS pollution. On one hand, surface failing systems can contribute with significant concentrations of organic matter, nutrients, and pathogens

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entering surface waters directly or by stormwater runoff. On the other hand, these systems release nutrients and pathogens through their drainfields into the ground, to the groundwater and ultimately to surface waters through groundwater discharge. These nutrients and pathogens can potentially contaminate both subsurface and/or surface waters if not properly treated prior to reaching the groundwater.

Many studies have been performed on OSWS such as system performance and design, nutrients and pathogen transport near and away from the drainfield. However, we still ask ourselves the questions of how much these systems really contribute to pollution? To what extent are these systems contaminating the groundwater and surface waters? In an effort to answer these questions, this review looks at the relevant studies conducted throughout the country and elsewhere in the past decades. These studies assess fate and transport of nutrients and pathogens that are introduced in the soils after leaving an OSWS.

## Nitrogen Contributions from OSWS

Nitrogen contributions from OSWS to the soil and eventually groundwater or surface water depend on several factors including soil texture, water table level, organic matter content in the soil, dissolved oxygen, and soil temperature. Figure 1 shows schematically the typical N transformations that take place as the effluent from OSWS is applied on to the soil.

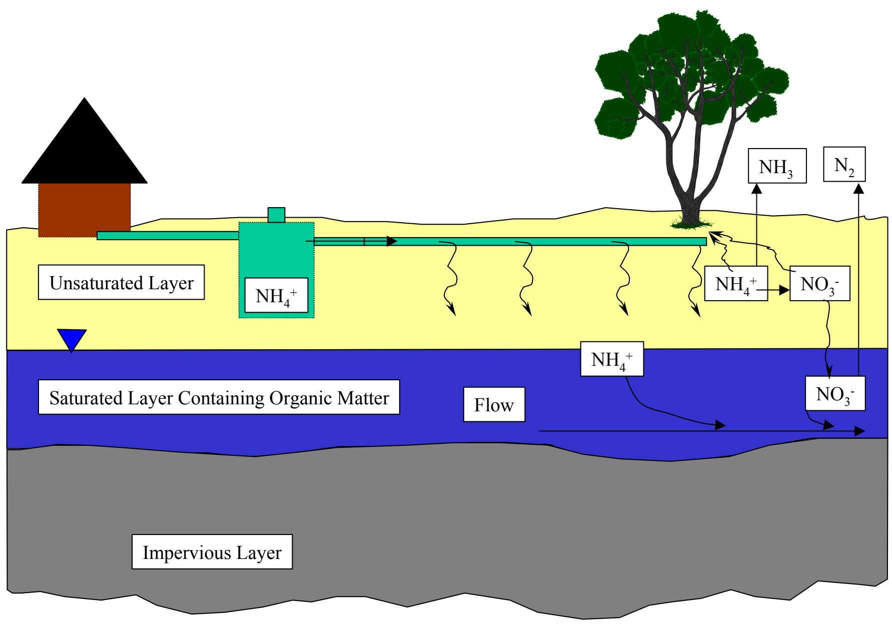


**Figure 1. N transformations in OSWS**

Figure 1 depicts the ammonium ($NH_4^+$) leaving the tank, flowing through the drain tiles and subsequently infiltrating in the soil. Nitrogen leaves the septic tank mainly in the form of dissolved $NH_4^+$ as documented by several research studies (Stewart and Reneau, 1981; Gerritse et al, 1995; Robertson et al, 1991; Walker et al, 1973; Viraraghavan and Warnock, 1975). These studies reported that $NH_4^+$ concentration in the tank effluent range from 30 mg/l to 100 mg/l ($NH_4^+$-N) and that 75-97 % of the N leaving the tank is $NH_4^+$.

The fate of $NH_4^+$ in the soil depends greatly on the water table level. For conventional septic tank systems, the highest water table level in the soil should be kept at a distance of 48 inches when suitable soils are present. Suitable soils for OSWS are described as well drained and range from sand to loamy sands to sandy loams to loams. These soils do not exhibit high water tables and ensure proper functioning of the septic tank. When $NH_4^+$ is applied on to these soils the oxygen present in the soil pores facilitates the transformation of $NH_4^+$ into $NO_3^-$. This process is called nitrification and occurs only when aerobic conditions are present. Several studies performed on sandy unsaturated soils reported almost complete nitrification (Weiskel and Howes, 1992 and Robertson et al., 1991).

Weiskel and Howes (1992) monitored a 53 ha residential area in Buttermilk Bay, Massachusetts with septic tank systems draining on to sandy soils with less than 0.1% clay content. The soils contained a negligible background N concentration. The study documented the depth and surface extension of the plume of contaminants as it flowed away from the drainfield into the bay. Weiskel and Howes (1992) found that $NO_3^-$ comprised 73% of the dissolved inorganic N in the plume before reaching the groundwater. Similarly Robertson et al. (1991) extensively monitored two individual households located on sandy soils in Cambridge, Ontario (Canada). This study also established a complex monitoring network of wells in order to document the depth, width and length of the plume as it flowed away from the drainfield. Robertson et al. (1991) reported that in one site 100% of the effluent $NH_4^+$ converted into $NO_3^-$ and 67% was reported in the other site.

Nitrification may not take place when nitrifying bacteria are not present. In such case $NH_4^+$ may adsorb onto soil particles via cation exchange. Bicki et al. (1984) summarized that $NH_4^+$ adsorption can be significant. The primary factors that determine the extent of $NH_4^+$ adsorption are the number of soil cation exchange sites exposed to the septic effluent, the affinity of the sites for $NH_4^=$, and the composition of the effluent. Adsorption of $NH_4^=$onto soils can reach an equilibrium once the cation exchange sites are filled with the cations from the effluent. Bicki et al. (1984) also explained that $NH_4^=$ can desorb and be subject to nitrification if the soil conditions allow it. Thus adsorption and desorption of $NH_4^=$ in soils can be a cyclic process driven towards finding an equilibrium.

The presence of high water table levels can cause $NH_4^+$ or $NO_3^-$ leachate into the groundwater. Continuously flooded conditions can limit the amount oxygen in the soil pores and therefore limit nitrification. In an anaerobic environment $NH_4^+$ is a stable cation that can diffuse towards groundwater and eventually move with the groundwater flow. When $NH_4^+$ leaches into the groundwater (as indicated in Figure 1) insignificant nitrification may take place at the groundwater table interface as the dissolved oxygen concentration rapidly decreases in the water column. Cogger and Carlile (1984) and Carlile et al. (1981) reported evidence of high $NH_4^+$ content in groundwater from septic systems.

Cogger and Carlile (1984) described the performance of conventional and alternative septic tank systems built on seasonally and continuously flooded soils. The study sites were located in Craven, Hyde, and New Hanover counties in the lower Coastal Plain of North Carolina. The soils found in the study sites were of coarse, silty, fine, and organic texture. Cogger and Carlile (1984) showed that continuously saturated systems exhibited higher $NH_4^+$ concentrations and lower $NO_3^-$ concentrations. In addition, they observed that seasonally saturated systems displayed higher $NO_3^-$ concentrations and lower $NH_4^+$ concentrations. The systems located on continuously saturated soils exhibited an $NH_4^+$ concentration of 8 mg/l and 5.7 mg/l [$NH_4^+$-N] on the average during the periods summer of 1979-summer of 1980 and fall 1980-winter 1981, respectively. Nitrate concentrations were rather low with averages of 0.3 mg/l and 2.1 mg/l [$NO_3^-$-N] for the same periods of time. In contrast, seasonally saturated systems averaged 3 mg/l and 1.9 mg/l [$NH_4^+$-N] for the above mentioned periods of time, respectively. The average $NO_3^-$ concentrations that the seasonally saturated systems reached were 2.1 mg/l and 4.9 mg/l [$NO_3^-$-N] during the same seasons. These results suggested that continuously saturated systems experience little nitrification; however, water table variability and alternate aerobic/anaerobic conditions in the soil prompt nitrification in seasonally saturated systems.

Carlile et al. (1981) presented an expanded a report on the same sites and systems as in Cogger and Carlile (1984). The septic tank systems included conventional, low-pressure pipe systems (LPP), mound and soil replacement. Some of the systems were exposed to continuously flooded conditions and some were seasonally flooded. Carlile et al. (1981) reported that $NH_4^+$ and $NO_3^-$ concentrations decreased with distance from the drainfield for both continuously flooded and seasonally saturated systems. On continuously saturated soils $NH_4^+$ concentrations ranged from average concentrations up to 26 mg/l [$NH_4^+$-N] for wells near the drainfield (5 m) to up to 3.0 mg/l [$NH_4^+$-N] for wells located at 100 ft from the drainfield. Nitrate concentrations were much lower averaging from up to 7 mg/l [$NO_3^-$-N] near the drainfield to 1 mg/l [$NO_3^-$-N] for wells located away from the drainfield. Three out of five of the LPP systems were continuously flooded. These systems displayed a similar behavior to the conventional systems.

If nitrification takes place in unsaturated soils that lack organic matter $NO_3^-$ usually leaches into the groundwater and eventually moves with the groundwater flow (see Figure 1). Numerous studies on OSWS have reported high $NO_3^-$ (greater than 10 mg/l [$NO_3^-$-N]) concentrations in groundwater and have indicated OSWS as the source. Bicki et al. (1984) summarized several of these studies and cited Preul (1966), Polkowski and Boyle (1970), Walker et al. (1973a,b), Childs et al. (1974), Wolterink et al. (1979), Rea and Upchurch (1980). These studies reported $NO_3^-$ concentrations varying from 10 mg/l [$NO_3^-$-N] to 70 mg/l [$NO_3^-$-N]. These concentrations were observed at distances greater than 10 ft and up to 100 ft. Recent studies such as Robertson et al. (1991) and Aravena et al. (1993) have also reported on significant $NO_3^-$ movement in a septic plume.

Robertson et al. (1991) documented rapid nitrification in a septic plume that diffused downwards toward the groundwater in a sandy soil in Cambridge, Ontario. The study followed the plume using a sodium tracer through a complex set of monitoring wells. The plume initially moved vertically downwards through the unsaturated soil layer, but as it reached the groundwater it switched to horizontal movement. Nitrate concentrations in the plume core varied within a range from 21 to 48 mg/l [$NO_3^-$-N]. Background concentrations varied from 17 to 34 mg/l [$NO_3^-$-N]. High background concentrations were due to agricultural practices. Nitrate concentrations did not change as the septic plume moved downgradient through a distance of 330 ft.

A later study performed at the Cambridge site by Aravena et al. (1993) confirmed Robertson et al. (1991) plume characterization and used stables isotopes of oxygen ($^{18}O$) and nitrogen ($^{15}N$) to identify $NO_3^-$ contributions to groundwater that originated in septic systems. As did Robertson et al. (1991), Aravena et al. (1993) identified the septic tank plume using a sodium tracer and described the plume as diffusing downwards right below the drainfield. The plume moved through a medium coarse sand aquifer until it reached a zone of compact pebbly sandy silt with lower hydraulic conductivity. Once the plume reached the more impermeable layer it switched from vertical movement to horizontal movement parallel to the groundwater flow. Aravena et al. (1993) concluded that the isotope $\delta^{15}N$ values in the plume reflected values reported for animal waste sources while the values of the non-plume $NO_3^-$ were within the values reported for organic N in soils. The study also showed that the isotope $\delta^{18}O_{H2O}$ values in the plume were smaller from those of the non-plume (-10.6±0.3‰ versus -9.4±0.4‰, respectively). An analysis of the $\delta^{18}O$ isotope in the well water showed a value of -10.8‰, which is within the range of the plume $\delta^{18}O_{H2O}$ isotope. Aravena et al. (1993) concluded that this isotope was a good tool to trace the plume. Aravena et al. (1993) was able to track the plume for 130 m from the tile bed. Nitrate remained dissolved in the groundwater and no significant denitrification took place.

Other N removal paths are ammonia volatilization, conversion of $NH_4^+$ into ammonia gas ($NH_3$), plant root uptake of $NH_4^+$ and $NO_3^-$, and microorganism incorporation of $NH_4^+$ and $NO_3^-$ (see Figure 1). These removal paths are considered non-significant and not permanent in the case of microorganism in the septic tank. However, some studies have found up to 45% N removal through plant uptake (Bicki et al., 1984). Plant uptake is not an important N removal path for conventional systems because the amount of N released by these OSWS greatly exceeds that which can normally be utilized by nearby plants. However, innovative and alternative OSWS such as recirculating sand hills and constructed wetlands can offer great pre-ground absorption N removal (Spooner et al., 1998).

Under anaerobic conditions denitrifying bacteria convert $NO_3^-$ into N gases. This reaction is the most significant N removal mechanism in soil environments. For denitrification to take place several conditions must be present in the soils: nitrification of $NH_4^+$, anaerobic conditions, presence of denitrifying bacteria, and adequate carbon source for the denitrifying bacteria must be present in the anaerobic soil. A study by Hinson et al. (1994) reported that these conditions might not always be present in the soils. Hinson et al. (1994) discussed on the performance of sand lined trench systems on wet clayey soils in North Carolina. This study monitored N at a site where drainage is used in conjunction with a septic system. The groundwater flow direction changed through out the year due to the complex drainage network in the area. The authors did not attempt to follow the N plume, nevertheless, high $NO_3^-$ concentrations were observed in some wells during the winter high water table. During low water table months, the authors believed that both $NH_4^+$and $NO_3^-$ were held in the unsaturated soil column. The study found that high water table months displayed high redox levels (Eh > 500 mV) and low water table months were associated with Eh values < 100 mV. Values for Eh between 100 to 500 mV (sub-oxic redox) are a sign of the anaerobiosis needed for denitrification to occur. Anoxic redox values with Eh <100 mV occurred through late summer to early fall. Subsequently, sub-oxic redox values preceded oxic redox values (Eh>500 mV) which started in early winter.

Other studies such as Robertson et al. (1991), Anderson (1998), and Gerritse et al. (1995) followed the septic plume and documented nitrate disappearance in soils. These studies did not report on redox levels, yet, organic matter content and low dissolved oxygen levels were pointed at denitrification as the cause of septic N disappearance in soils.

Robertson et al. (1991) monitored a site located on the edge of the Muskoka River in Ontario, Canada. The site was a conventional septic system located on a 10-m fine fluvial sand layer overlying granitic bedrock. The septic system discharged to a tile bed of 80 $m^2$ in area and 20 m away from the Muskoka River. The water table was located at about 3 m from the surface at the drainfield. The land use was mainly residential and no important N background levels were reported. This study followed the $NO_3^-$ plume as it moved away from the drainfield. The average $NO_3^-$ concentration reported in the drainfield was 0.1 mg/l [$NO_3^-$-N] while the average $NH_4^+$ concentration was 59 mg/l [$NH_4^+$-N]. The average concentrations of $NO_3^-$ and $NH_4^+$ found at the plume core were 39 mg/l [$NO_3^-$-N] and 0.5 mg/l [$NH_4^+$-N] respectively. Thus, nitrification was almost complete and $NO_3^-$ diffused in the groundwater. The $NO_3^-$ plume was extensively documented using a network of wells that captured $NO_3^-$ concentrations at several depths. As $NO_3^-$ approached the river bed sediment the average concentrations reported declined from 30 mg/l (in the center of the plume) to 10 mg/l (before reaching the riverbed). In addition, Robertson et al. (1991) reported that the fraction of organic carbon (*foc*) in the aquifer was 0.0003 while the *foc* in the riverbed sediments was 0.017, 60 times larger. Moreover, the dissolved oxygen concentrations were depressed in the plume but increased at the riverbed interface. The readings from seepage meters located at the riverbed interface indicate that $NO_3^-$ remained under 0.5 mg/l [$NO_3^-$-N]. These data strongly suggested that denitrification occurred in the aquifer riverbed interface. The most vigorous zone of denitrification found through the seepage meters was at 0.5 m below the riverbed.

Anderson (1998) monitored a septic system located on a raised lot that drained to the Indian River Lagoon (an estuary), in Florida. The soils were described as sandy soils consisting of dark brown fine sand with some organic material and varying to reddish brown fine sands with organic material. The sands contained a *foc* varying from 0.5 to 1.5 %. The dissolved oxygen varied from 3.2 to 5.1 mg/l. The water table was generally within the first foot of soil during the wet season. The groundwater gradient was established using a tracer study and a network of wells was installed to follow this gradient. The study reported $NO_3^-$ concentrations in the wells varying from 9 mg/l [$NO_3^-$-N] near the drainfield to <0.05

mg/l [$NO_3^-$-N] 50 ft away from the drainfield. Ammonium concentrations were not reported, however, total N concentrations did not differ significantly from those reported for $NO_3^-$. The author discussed that literature values for denitrification rates were positively correlated with reported values of *foc.* Using this argument, Anderson (1998) concluded that this site had a great potential for denitrification as the mechanism responsible for $NO_3^-$ removal from the soil near the lagoon.

Gerritse et al. (1995) measured N concentrations from a modified Australian septic tank system on the Darling Plateau in Western Australia located in a residential area. The septic system was located in a 0.4-ha residential lot that had only a residential land use. The site drained to a nearby creek located 262 ft away from the system. The septic tank drained into a 40-ft long leach drain made of an open-ended concrete slab or leach drain embedded in sandy soils. The soil was described as gently undulating terrain with well-drained, shallow to moderately deep (up to 3 ft) sands, overlying lateritic duricrust. The lateritic layer was saturated during three months at the end of the summer. A network of wells covering depths from 0.5-3 ft and a surface area of 130-ft wide × 230-ft long was used to monitor N species. Groundwater movement was monitored using a bromide tracer. The groundwater followed the topographic gradient. Tracer concentrations decreased with distance indicating a high septic plume dilution in groundwater. High $NO_3^-$ and $NH_4^+$ concentrations near the leach drain were reported. However, negligible amounts of both $NO_3^-$ and $NH_4^+$ were found in the wells located 32 ft away from the leach drain and those following downgradient towards the creek. The mass ratio of inorganic N to excess bromide declined with distance indicating that a factor additional to groundwater dilution was responsible for low N concentrations. Gerritse et al. (1995) concluded that denitrification was a likely cause of $NO_3^-$ disappearance. The authors argued that pH (6.1-7.7) and temperature (18$^o$C) conditions were ideal for denitrification. Furthermore, the changing water table above the lateritic layer and the temperature and pH conditions in the soils were conducive to fluctuating anaerobiosis in the soil. This study did not report any amount of organic matter or dissolved oxygen in the soil.

## Phosphorus Contributions from OSWS

Phosphorus in septic tank effluent is derived from organic molecules and detergents and dishwashing powders. Most of the P contributions from OSWS are in the form of orthophosphate ion ($PO_4^{3-}$). Wilhelm et al. (1992) reported that approximately 76% of the total P in the septic tank effluent was in the form of $PO_4^{3-}$. Polyphosphates and P in organic molecules are hydrolized to $PO_4^{3-}$. Once in the soil $PO_4^{3-}$ removal occurs by adsorption of $PO_4^{3-}$ onto hydrous oxides of iron, aluminum, manganese, and carbonate surfaces on soil particles (see **Figure 2**). Plant root and microbial uptake of $PO_4^{3-}$ is another non-permanent way of removing P. Adsorption processes for P are intricately related to precipitation reactions. These reactions are believed to be the main cause of $PO_4^{3-}$ removal in soils as complexation or precipitation occurs with very low concentration of $Al^{3+}$, $Ca^{2+}$, and $Fe^{3+}$. These precipitates are known as varisite, hydroxylapatite, and strengite respectively. These precipitates are formed in oxidizing conditions, that is, when oxygen is present in the soil pores.

Phosphorus concentrations in shallow groundwater are normally low with values ranging from 0.005 mg/l to 0.1 mg/l (Reneau et al., 1989). Septic tank effluent can contain total P concentrations ranging between 7 mg/l - 15 mg/l (Wilhelm et al., 1994). Regardless of these high contributions from OSWS, many studies have documented high degree of $PO_4^{3-}$ sorption and complexation within the first few meters downgradient from the drainfield (Reneau et al., 1989; Weiskel and Howes, 1992; Robertson et al., 1991; Wilhelm et al., 1994).

A study performed by Reneau et al. (1989) discussed the mechanisms that drive P removal in soils based on collected evidence from different OSWS studies. Reneau et al. (1989) argued that in acid soils P complexation occurs with $Fe^{3+}$ and $Al^{3+}$ whereas in basic calcareous soils complexation of P occurs with $Ca^{2+}$. This study referred to Reneau and Pettry (1976) who studied an OSWS that had been in use for 15 years. This system was built in an acid Plinthic Paleudult. This study found that compared to a control profile this soil showed large increases in P associated with aluminum and ferrous complexes. Complexation occurred within the first 0.15 m from the system. The authors reported that $Al^{3+}$-P and $Fe^{3+}$-P fractions increased with time after long-term exposure of the soil to the OSWS effluent.

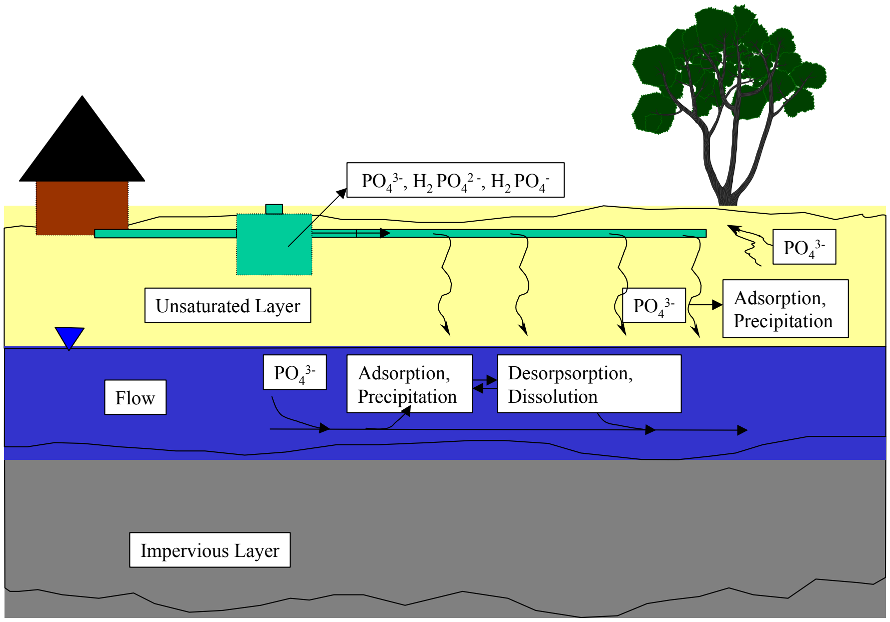


**Figure 2. P transformations in OSWS**

Weiskel and Howes (1992) reported a high degree of $PO_4^{3-}$ removal (approximately 97%) in the sandy soils of the Buttermilk Bay, Massachusetts. The soils were granite-derived sand and gravel materials and relatively rich in iron and aluminum oxides. Weiskel and Howes (1992) did not report any evidence of precipitate formation or levels of aluminum and iron. However, they monitored extensively the $PO_4^{3-}$ plume through the coastal watershed. This study showed that $PO_4^{3-}$ is rapidly reduced from an average concentration of 5.1 mg/l [$PO_4^{3-}$-P] in the septic effluent to 1.39 mg/l [$PO_4^{3-}$-P] immediately before the plume reached groundwater and concentrations decreased to zero within the first 2.5 meters. Monitoring at sites located close to the beach the study reported $PO_4^{3-}$ concentrations < 0.02 mg/l [$PO_4^{3-}$-P].

Robertson et al. (1991) reported rapid $PO_4^{3-}$ removal at two study sites in Cambridge and Muskoka, Ontario. At Cambridge, soils were described as a sand layer of glaciolacustrine and outwash deposits overlaying low permeable silt till. Phosphate concentration at the tile effluent was 8 mg/l [$PO_4^{3-}$-P], this amount was reduced to 5 mg/l [$PO_4^{3-}$-P] and very rapidly to less than 0.05 mg/l [$PO_4^{3-}$-P] in approximately 10 m downgradient. Attenuation in the unsaturated zone was little compared to that in the groundwater zone. The unsaturated zone was oversaturated with hydroxylapatite as documented by Wilhelm et al. (1994). Thus, $PO_4^{3-}$ had very few adsorption places available. The $PO_4^{3-}$ attenuation zone was a zone of increasing pH and high $Ca^{2+}$ concentrations of 88 mg/l. The authors concluded that these conditions favored hydroxylapatite precipitation. At the Muskoka sites soils were fine fluvial sands occurring to a depth of more than 10 m and overlying a granitic bedrock layer. The tile effluent was 13 mg/l [$PO_4^{3-}$-P] but before the effluent reached the groundwater the plume exhibited $PO_4^{3-}$ concentrations of < 0.1 mg/l [$PO_4^{3-}$-P]. The unsaturated layer was 3 m and a 99 % reduction was observed within the first 2 m. The calcium concentration in this soils was not very high, only 44 mg/l and the pH was 5.1. Robertson et al. (1991) argued that precipitation of strengite or varisite are the driving processes in $PO_4^{3-}$ removal. Background concentrations of iron and aluminum ions were not reported.

Reneau et al. (1989) suggested that the enhanced ability of P removal in such short distances is caused by soil regeneration of $Al^{3+}$, $Fe^{3+}$, and $Ca^{2+}$. This suggestion was based on evidence from several studies that found an enhanced sorption capacity of the soil (Ellis and Erickson, 1969; Sawhney and Hill, 1975, Uebler, 1984). In addition, Reneau et al. (1989) argued that field data that suggested an enhanced P removal was in discrepancy with laboratory data on sorption maximums as evidenced by Whelan and Barrow (1984). Whelan and Barrow (1984) observed that P sorption characteristics obtained in the laboratory agreed to the amount of P present in the field around an OSWS. However, the quantity of P accumulated in the soil was higher than the one predicted from the equilibrium isotherms for the septic tank plume. Laboratory sorption experiments predicted P horizontal movement of 5 m while only 135 cm was observed in the field.

Saturated conditions might dissolve $PO_4^{3-}$ precipitates formed in acid soils. Thus $PO_4^{3-}$ ion desorbs and becomes available for subsequent adsorption and precipitation. Studies such as Carlile et al. (1981) and Cogger and Carlile (1984) support this theory. These studies monitored total P in seasonally to continuously saturated soils in Craven, Hyde, and New Hanover counties in the lower Coastal Plain of North Carolina. The soils found in the study sites were of coarse, silty, fine, and organic texture. Seasonally saturated OSWS exhibited lower P concentrations than saturated systems. Saturated systems P concentrations exceeded 1.0 mg/l [P] for wells located 5 ft away from the drainfield. However there was no difference in P concentrations between seasonally saturated and continuously saturated systems for wells located at 25 ft and 50 ft away from the drainfield.

**Pathogenic contributions from OSWS**

Human beings who are infected or carriers of disease discharge pathogenic organisms in wastewater. Pathogenic organisms can be bacteria, parasites such as protozoa and helminths, and viruses. Pathogenic bacteria of human origin are the cause of diseases of the gastrointestinal tract such as typhoid fever, dysentery, diarrhea, and cholera. Table 1 shows the typical concentrations found in septic tank effluent and untreated wastewater and the corresponding infectious dose (Crites and Tchobanoglous, 1998).

**Table 1 Typical Concentrations of Pathogens in Septic Tank and Raw Wastewater Effluent**

| Organism | Concentration MPN/100 ml | Infectious dose MPN/100 ml |
|---|---|---|
| Bacteria | | |
| Coliform, fecal | $10^6$-$10^8$ | $10^6$-$10^{10}$ |
| *Shigella* | $10^0$-$10^2$ | 10-20 |
| *Salmonella* | $10^2$-$10^4$ | N/A |
| Protozoa | | |
| *Cryptosporidium parvuum oocyst* | 10-$10^3$ | 1-10 |
| *Giardia lamblia cysts* | $10^3$-$10^4$ | <20 |
| Helminths | | |
| Ova | 10-$10^3$ | N/A |
| *Ascaris lumbricoides* | N/A | 1-10 |
| Viruses | | |
| Enteric virus | $10^3$-$10^4$ | 1-10 |
| Coliphage | $10^3$-$10^4$ | N/A |

Protozoa such as Cryptosporidium parvum and Giarda lamblia can cause severe diarrhea, stomach cramps, nausea, and vomiting. These symptoms can be of great concern in people with compromised immune systems. Helminthic parasites (adults or eggs) can infect humans; some of these parasites are resistant to environmental stresses and can survive usual wastewater disinfection procedures. Enteric viruses can be present in the fecal matter of infected humans. The most important ones are enteroviruses (e.g. polio), Norwalk viruses, rotaviruses, calciviruses, and hepatitis A.

Fecal coliforms are usually used as indicator organisms of pathogenic water contamination because they are easy to test for and more numerous. However, their presence may not directly indicate that other pathogens are present. A study conducted by Sobsey and Scandura (1981) in various septic system sites in the coast of North Carolina found that fecal coliform movement through the soil differed significantly from viral movement. Viruses were found at distances up to 35 m from the drainfield while no findings were obtained for fecal coliforms. In addition, fecal coliform counts did not coincide with virus counts for closer distances. However the presence of coliform bacteria always coincided with the presence of virus in the soil.

Several processes can act in the removal of pathogens from the septic effluent once it is applied onto the soil. Pathogens can be retained in the soil by entrapment or filtering, soil adsorption, and natural die-off. Pathogen filtering is enhanced as soil pore size decreases. Helminths and other protozoa (12 μm - 400 μm) are the largest and are comparable in size to sand particles (20 μm - 2000 μm). Protozoa and some bacteria (10 μm- 100 μm) are similar in size to silt particles (6 μm- 60 μm). Generally bacteria (0.2 μm- 5 μm) are the size of fine silt and coarse clay particles (2 μm- 6 μm). Finally, viruses (0.02 μm- 0.25 μm) are the size of very fine clay particles (0.01 μm- 0.1 μm). Peterson and Ward (1989) modeled possible filtering processes for given water velocity and soil pore size. They concluded that filtering occurred mainly for large sized pathogens (protozoa and helminths) as they are trapped in the soil pore spaces. Bacterial filtering also occurs but in lesser quantities. As bacteria fails to be sieved by the soil so do viruses.

Adsorption of microorganisms in the soil is maximized when conditions such as uniform effluent distribution, development of a surface clogging mat, well drained soils, and moisture deficits are present in a septic tank system (Reneau et al., 1989). These conditions are typical of unsaturated flow. Studies such as Reneau et al. (1989) and Pekdeger and Matthess (1983) discussed that the cation bridging between the negatively charged microorganisms and soil colloids as they repulse each other drove adsorption processes. Studies performed in soil columns such as Drewry and Eliasten (1968) and Carlson et al. (1968) confirmed that high ionic strength in soils enhanced virus adsorption. Thus, ionic strength of the water is an important feature that can maximize pathogen adsorption on to soil particles.

On the other hand, pathogens can desorb when lower ionic strength waters, such as rainfall, come into contact with soils with high ionic strength (Reneau et al., 1989). Scandura and Sobsey (1981) performed laboratory experiments on viral movement through soil columns. They observed that simulated rain (distilled water) applied on to soil columns resulted in more viral movement along the column. Sandy soils showed high viral elution concentrations (~10000 MPN/100 ml - 100000 MPN/100 ml) similar to the concentrations of applied septic tank effluent (8000 MPN/100 ml - 10000 MPN/100 ml). However, Sobsey et al. (1980) reported that soils that had clay content of 30% did not present viral movement or elution. Thus, lower ionic strength waters can drive pathogenic movement through soils subject to soils structural characteristics.

Aerobic conditions are unfavorable to septic bacteria and viruses and promote the survival of aerobic soil bacteria. Eventually large sized pathogens die off as they are subject to predation by aerobic soil bacteria. When anaerobic conditions occur, survival shifts in favor of the septic anaerobes. Survival competition was suggested by studies such as Sobsey et al. (1980), which found that for eight different types of soils virus survival was longer in sterile distilled water soil suspensions than in the presence of aerobic bacteria. Other studies such as Badgarsaryan (1964), Romero (1970), and Hurst et al. (1980) confirm this finding.

Soil structure can influence pathogen movement through soil pores. Clays can be more effective than sands in reducing pathogens. Crane and Moore (1983) reported that under unsaturated flow conditions bacterial population could be reduced to up to 95% within the first 1-5 cm of soil. This study also noticed that clays were more effective than sand in bacterial removal. Sobsey et al. (1980) found that clays achieved 99.99% reduction of viruses while sandy and organic soils achieved only 95 to 99.8% reductions.

Soil moisture content and temperature can influence pathogen survival in the soil. Reneau et al. (1989) reported on those studies such as Gerba et al. (1975) and Hurst et al. (1980). These studies found that under controlled laboratory conditions moist soil and low temperatures favored bacterial and viral survival. In addition, Sobsey and Scandura (1981) found that winter temperatures and water table conditions favored the survival of viruses in sand and sandy loams. Viruses were detected up to 59 days after inoculation during winter compared to 41 days during summer.

Several water quality surveys have assessed the occurrence of groundwater and surface water contamination with pathogens. These surveys pointed out that OSWS are the likely cause of contamination. However, many of these studies did not actually evaluate the extent of the contamination that originated from OSWS. Bicki et al. (1984) summarized some of these water quality surveys such as Woodward et al. (1961) and David and Stephenson (1970), among others. Woodward et al. (1961) reported on contamination in groundwater from OSWS in 39 communities around the Minneapolis-St. Paul, Minnesota area. The soils were described as till, sand, gravel or fractured or jointed solution riddled limestone. Groundwater wells showed elevated $NO_3^-$ concentration (11% of surveyed wells had concentrations greater than 10 mg/l $NO_3^-$-N) and pathogenic contamination. Davis and Stephenson (1970) indicated that 51% of 194 private wells surveyed in Bartow County, Georgia were contaminated with bacteria and OSWS were the likely cause.

Bowers (1980) performed a study in Henry County, Indiana. This study reported that streams and ditches were contaminated with fecal bacteria. Bacterial counts were up to $3.9*10^6$ MPN/100 ml and as low as 10 with an average count of 564000±300000 MPN/100 ml. A large percentage (78%) of the soils in this county had low permeability rates, present ponding conditions and underlain a glacial till located 36 inches deep. This study summarized reports on several infectious diseases such as diarrhea, hepatitis, infectious hepatitis, viral meningitis, and encephalitis. The author argued that previous to 1978 the amount of denied permits for OSWS was null and that permits were issued after the systems were already in built in place. However, contributions of OSWS to surface or subsurface waters were not identified.

Other studies, available in the literature, did assess the contributions from OSWS to pollution of surface and groundwater. Some of these studies (U.S. EPA, 1975; Brandes, 1972; Wilson, 1982) were reported by Bicki et al. (1984). The U.S. EPA (1975) reported on studies performed in Florida and North Carolina. These studies showed high concentration of fecal coliforms in surface waters at Punta Gorda and Big Pine Key, Florida; and at Atlantic Beach and Surf City, North Carolina. These cities comprised dense housing developments with OSWS built in close proximity to surface waters. Effluent from OSWS reached surface waters as evidenced by dye tracer studies. These studies followed the path of Rhodamine WT dye from house drains to septic tanks and detected the dye at the nearby canal. At Punta Gorda the detection time was 25 hours while at Big Pine Key it took 110 to 150 hours for the dye to reach the canal. In contrast, at Atlantic Beach and Surf City the dye was detected after 4 hours and 60 hours respectively. The background coliform concentrations at Punta Gorda and Big Pine Key were 203 MPU/100 ml and 10 MPN/100 ml respectively. However, canal concentrations ranged from 436 to 871 MPN/100 ml in a residentially undeveloped canal and 176 to 1809 MPN/100 ml in a developed canal. At Big Pine Key the coliform concentrations for an undeveloped canal were not different from the background but at a developed canal the concentrations ranged from 14 to 32 MPN/100 ml. At Atlantic Beach the background concentrations and mean surface water concentrations were 3400, 400, and 360 MPN/100 ml for the end, middle and mouth of a canal with high residential density. At Surf City a septic plume was traced with dye and concentrations were found to be greater than $2.4*10^6$ MPN/100 ml at the mouth of a canal.

Brandes (1972) reported on OSWS located 27 to 53 ft from Lake Chemong, Ontario (Canada). Soils were sandy loam and silt loam fill materials with stones and boulders. The water depth was 5 to 7 ft. The effluent from the OSWS contained an average concentration of $8*10^6$ and $4.7*10^6$ for total and fecal coliforms respectively. The concentrations reported in the groundwater at 5 ft from the drainfield were $8*10^6$ and $2.4*10^6$ MPN/100 ml for total and fecal coliforms respectively. At distances of 22 and 34 ft from the drainfields the concentrations for fecal coliforms dropped to 1500 and 100 MPN/100 ml respectively.

Wilson (1982) monitored the effluent flow from eight systems located on moderately well to somewhat poorly drained soils. Artificial drainage was practiced in these soils using a perimeter drainage system. Tile drains located at 20 ft from the drainfield were placed 6 ft below the soil surface. This system effectively lowered the water table by 2.5 ft below the drainfield. Discharge from the tile outlet had a wide range of total coliform concentrations from 470 to 2380 MPN/100 ml with an average of 1468 MPN/100 ml. The average fecal coliform concentration was 202 MPN/100 ml with a range of 47 to 484 MPN/100 ml.

Kerfoot and Skimmer (1981) used a septic leachate detector on the shoreline of Crystal Lake, Benzie Co., Michigan. The soils were drained sands to loamy sands underlying wash plains and till plains. This study detected three types of plumes that originate in OSWS: erupting, dormant and surface water plumes. The plumes were detected using an instrument that was sensitive to ultraviolet (UV) fluorescent organics derived from surfactants, softeners, and natural degradation products that persist in low oxygen conditions. Kerfoot and Skimmer (1981) defined emergent plumes were as subsurface discharges of effluent that had a higher content of dissolved solids from that present in the background. Dormant plumes indicated the subsurface seepage of effluent with no apparent difference in dissolved solids with the background concentration. Stream source plumes were subsurface discharges of effluent that could be traced back to the shoreline. This study found high densities of erupting plumes in both the northeast and southeastern shores of the Lake. The west shore had no plumes, which coincided with the fact that many housing units in this shore were located far from the shoreline and the groundwater flow direction goes from the lake into the shoreline. Reverse groundwater flow in the west shore was explained as groundwater movement from Crystal Lake to Lake Michigan, which was 6 meters lower in altitude. The east bank was devoid of plumes, which matched the presence of a sewered town. Measurements of the groundwater flow showed that dormant plumes coincided with reversed flow plumes (from the lake into the shoreline). Coliform content was reported in a range from 100 to 9300 MPN/100 ml for total coliforms and 10 to 120 for fecal coliforms.

Sobsey and Scandura (1981) conducted a study on OSWS located in Craven Co. and New Hanover Co., North Carolina. The sites in Craven Co. had sandy soils that varied to clay and loams. These sites had very low organic matter content, varying from < 0.1 to 5.4 %. The soils were slightly acidic with pH varying from 4.1 to 6.5. The ionic content of the soils was low varying from 1.2 to 8.5 Me/100cc. Water depth from the soil surface varied from 0.7 to 4.8 ft on the average. Fecal coliform content of the septic plume varied from <2 to 3218 MPN/100 ml for wells placed 5 ft away from the drainfield. For wells placed 50 ft away from the drainfield the fecal coliform concentrations varied from <2 to 1600 MPN/100 ml. Fecal coliform concentrations reported during the winter and spring experiments were markedly lower than those reported in the summer. Sobsey and Scandura (1981) observed that groundwater movement was generally in one distinct direction.

Sobsey and Scandura (1981) also monitored the groundwater wells for specific viruses (BE-1 and E-1) chosen as tracers. The study found that detection of fecal coliforms was not a good indicative of the presence of viruses especially for long distances away from the drainfield. A very weak correlation between virus concentrations and fecal coliform concentrations in the well water (r=0.645). However, the authors find this correlation inconclusive due to significant differences in sample population and timing of events. Similarly, a very weak correlation (r=0.667) was found for the levels of virus concentrations in the wells and the distances from the wells to the drainfield. The authors argued that most wells were within the first 10 ft from the drainfield. In addition, most of the groundwater samples containing viruses were from wells located at 5 ft from the drainfield. Sobsey and Scandura (1981) also found that the relationship between rainfall occurrence and virus movement in these systems was unclear. There was no significant correlation between the appearance and non-appearance of viruses in the wells after a rainfall event. Neither there was a significant correlation between the intensity of a rainfall event and the observed virus concentrations. Similar observations were made for coliform organisms. Sobsey and Scandura (1981) and Scandura and Sobsey (1997) reported that a direct correlation (r=0.918) was found between the frequency of virus isolations in the groundwater wells and the groundwater pH values. The authors explained this result by septic plume mixing with acidic groundwater which allowed viruses to survive longer especially under saturated conditions. In addition, the authors concluded that high pH values in the groundwater of typically acid soils can indicate extensive contamination by septic effluent.

Hagedorn et al. (1978) used antibiotic bacteria to monitor the degree of movement and subsequent groundwater contamination by OSWS drainfield under saturated conditions. Antibiotic strains were used in order to differentiate fecal coliforms originated in OSWS from other fecal bacteria and to determine flow movement rates. Bacteria were inoculated at concentrations ranging from $3*10^8$ to $5*10^8$ MPN/ml into deep pits constructed to simulate drainfields. The soils were described as silt loam varying to clay loam. The pH levels were from 5.4 to 6.0 and the organic content varied from 0.6 to 5.2%. The cation content varied from 11.2 to 35.1 Meq/100g The bacteria were monitored for 32 days using wells installed in concentric rings at 50 cm and 100 cm from the inoculation point. Diffusion of antibiotic bacteria occurred in all directions for the first 50-cm from the inoculation point. The northeastern and eastern directions were determined as the preferential directions. A more complex network of wells was installed to follow the preferential directions. Samples were taken at several distances (3, 5, 15, and 30 m) from the drainfield at 50 cm depth from ground surface. During the first day after inoculation bacteria had traveled from 3 to 5 m in both sites. The authors observed that rainfall events helped the bacteria population move downgradient as a front or pulse through the soil. The pulse traveled through the groundwater wells and its peak concentration attenuated along its path due to groundwater dilution and soil filtration of bacteria. The authors concluded that these bacteria survived in appreciable numbers throughout the 32 days-sampling period.

Rahe et al. (1978) investigated bacterial movement in the event a drainfield was submerged in a perched water table. This study inoculated three distinct bacterial strains, at a concentration of $1.4*10^9$ MPN/ml into three horizontal drains that were installed into A, B, and C horizons at two sites located in a western Benton Co. Oregon Hillslope. The soils in the first site were described as silt loam varying to massive clay. Soils for the second site were silty clay loam varying to overlying fractured saprolite, which was located at 2 ft deep. At the first site, the soil values of pH varied from 5.6 to 6.1, the organic matter content varied from 0.9% to 4.8%, and the cation content was within the range of 12.4 meq/100g to 20.85 meq/100g. At the second site, the soil values of pH varied from 4.7 to 5.2, organic matter content was within the range of 3% to 4.3%, and the cation content varied from 17.5 meq/100g to 52.63 meq/100g. Monitoring wells were placed at various depths, varying from 0.4 ft to 6.5 ft and at distances from the drainfield varying downgradient from 8.2 ft to 50 ft. Artificial water tables were maintained and monitored by applying water with sprinklers. Background concentrations were negligible. The second site showed rapid movement of bacteria with little to no dilution and/or diffusion was observed. Movement was observed in great quantities at depths between 0.4 ft to 6.5 ft and at distances from the drainfield from 16.5 ft to 50 ft. The study reported concentrations varying from 0 to $10^4$ MPN/ml after 2 hours, 4 hours, and 12 hours of inoculation. Distances downgradient presented concentrations up to $10^4$ especially for times after 12 hours of inoculation.

Results on the first site showed that the movement rate was much more slowly. The authors attributed this result to lower hydraulic gradient and hydraulic conductivity. Lower numbers of bacteria were recovered due to greater soil filtration. Bacterial counts varied from 0 to $10^3$ after 12 hours. These numbers were obtained from wells located up to 16.5 ft downgradient and within 1 ft from the ground surface. Bacterial counts were observed up to $10^2$ MPN/ml after 48 and 72 hours bacterial counts for downgradient wells. These numbers were only observed within the first 2.6-ft of soil. After 56 hours only the wells located at 16.5 ft from the drainfield showed up to $10^2$ MPN/ml bacterial counts for the first 2.6 ft of soil. The authors concluded that macropores played a very important role in bacterial movement in flooded soils. In addition, the authors recommended that a true saturated hydraulic conductivity is an important figure to be assessed when faced with soils with a significant percentage of macropores. The study warned that OSWS installed on soils with fractured saprolite could potentially contribute to significant pathogenic contamination of surface waters.

Ijzerman et al. (1993) inoculated two antibiotic resistant *Escherichia coli* strains and two coliphages in a LPD system installed in William Co., Virginia. The soils at this site were loam varying to silty clay loam overlying shale rock. The soils were very shallow and shale rock was located at only 2 ft from the soil surface. The study analyzed three independent LPD systems operating under different actual loading rates of 4.1, 7.7, and 16.71 l/m$^2$d. The fate and transport of the biological tracers below each system was monitored through a network of sampling wells located at various depths (0-3.3 ft) below the trenches, which were installed at 1 ft depth from the soil surface. The bacterial population inoculated was $7.8*10^6$ colony forming units (cfu) per milliliter during the summer and $2.0*10^7$ cfu/ml during the winter. The concentration of coliphage used in the summer inoculation was $1*10^4$ plaque forming units (pfu) per milliliter and $1*10^7$ pfu/ml during the winter. The reported background concentrations were negligible of both bacteria and coliphage microorganisms. Ijzerman et al. (1993) reported that for both experimental sites, during the summer period the bacterial tracer was observed with more frequencies at levels >100 cfu/50 ml than the coliphage tracer. However, during the winter both tracers were observed at levels of >100 cfu/50 ml and 100-1000 pfu/50 ml for bacteria and coliphages respectively. The authors found that during the summer the system with the lower loading rate had the greatest removal of 99.9% compared to 99% for the greater loading rate. During the winter both loading rates met a high level of tracer retention but the lowest loading rate was the most effective. Winter season proved to favor both coliphage and bacterial survival for lower depths over 72 hours after inoculation. The authors concluded that low temperatures increased the survival of both pathogenic tracers.

Cogger et al. (1988) evaluated wastewater treatment by septic systems functioning on fine sand typical of the conditions found in Atlantic Beach, North Carolina. The study assessed effects of loading rate and water table depth on bacterial counts. Two drainfields, nominated upper and lower according to topographic conditions, were used under loading rates of 1, 4, and 16 cm/d. Monitoring wells were installed around the trenches at depths of 4.9 to 6.2 ft for shallow wells, which were coupled with deeper wells. The study monitored nutrient and pathogenic movement through the sandy soils directly beneath the drainfields. Both fields had a variable separation distance between trench bottom and water table. The upper field presented a separation of 2-3 ft 72% of the time. The water table was 1-1.5 ft away form the lower field 50% of the time and the trenches were nearly saturated 10% of the time. The upper field displayed bacterial concentrations varying from $1*10^{1.3}$ MPN/l to $1*10^{2.4}$ MPN/l. These numbers were much lower than those observed in the lower field ($1*10^{1.5}$ MPN/l to $1*10^5$ MPN/l). The authors explained these differences as a result of water table proximity. The lower field had nearly saturated and anaerobic conditions which favored pathogen survival. Concentrations in the upper field also varied with loading rate. A concentration of $1*10^{2.4}$ MPN/l was observed for the highest rate while a concentration of $1*10^{1.3}$ MPN/l was observed for the lowest rate during the first year. Similar results were observed with the lower field. Yet, during the second period with drier conditions and deeper water tables, these marked tendencies were not observed with either drainfield. The authors reported $\log_{10}$ reductions for different pathogenic dosage. Viruses such as BE-1 and coliphages in the upper field showed removals of 99.96%

and 99.87% respectively. In contrast, fecal coliforms and fecal streptococci in the upper field showed a lower removal of 99.7% and 90.0% respectively. Similar trends were observed for the lower field. The authors did not monitor pathogenic or nutrient movement away from the drainfield. The sea and a marsh were located 33 ft away.

## Summary and Discussion

Despite groundwater dilution and adsorption N contributions to groundwater from OSWS can be significant through long distances away from the system. Denitrification occurs only when anaerobic conditions and enough carbon content are present in soils. These conditions can be present at the groundwater/river interface and can lead to denitrification. Although denitrification may happen at a particular site conditions that lead to denitrification may not always be present in time. Redox levels in soils can change with time as a function of variable water table levels.

Phosphorus contributions to surface and subsurface waters from OSWS are minimal. Phosphorus adsorption and precipitation in soils are a major pathway for P removal. Phosphorus concentrations may increase when saturated conditions are present. However, even under these conditions high P removal occurs within a few feet from the drainfield.

Pathogenic contributions to groundwater can be significant. Rainfall, low temperatures, and saturated conditions can enhance bacterial movement to great distances from the drainfield. Viral movement can differ greatly from bacterial movement. Low temperature and saturated conditions enhance virus survival and movement near the drainfield. However, for distances further away from the drainfield viral movement and survival is not well understood.

Assessments at basin and subbasin levels of OSWS nutrient and pathogenic contributions to surface and subsurface waters are needed. These studies must be targeted for basins or subbasins with nutrient sensitive waters and critical soils. Monitoring studies on these areas must be comprehensive and must be able to differentiate background concentrations of nutrient and pathogens from those originating in OSWS. Important issues to be studied are the effects of OSWS density on surface and subsurface waters, rainfall effects on nutrient and pathogen movement is soils, and proximity of OSWS to surface waters amongst others.

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