Coastal pollutant Loading from On-Site Treatment & Disposal Systems

D.E. Meeroff, F.J. Morin, and Frederick Bloetscher

D. E. Meeroff, Ph.D., and Frederick Bloetscher, Ph.D., P.E., are assistant professors in the Department of Civil Engineering at Florida Atlantic University. At the time this article was written, F.J. Morin, MSCE, was a graduate student in the department.

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Household wastewater contains a variety of constituents, including 1) oxygen-demanding organics, which can deplete dissolved oxygen in receiving waters; 2) pathogenic microorganisms, which can spread disease; and 3) nutrients, which can stimulate the growth of aquatic plants. Domestic wastewater may also contain potentially toxic, mutagenic, or carcinogenic compounds. For these reasons, proper disposal of this waste stream is necessary to protect the public health and the environment.

As opposed to a centralized treatment facility, which collects, treats, and discharges wastewater on a regional basis, onsite treatment and disposal systems (OSTDS) treat and discharge sewage within the site where the wastewater originates. The goal of an OSTDS is to adequately treat a relatively small waste flow to prevent the surrounding surface water, groundwater, and soil from being contaminated.

OSTDS are generally most applicable to residential use. The most common type of OSTDS is the septic tank system (Kaplan 1991). Septic tanks are buried, watertight receptacles designed and constructed to receive wastewater, separate and store solids, provide limited digestion of organic matter, and allow partially clarified effluent to discharge to a gravel drainfield through a series of perforated underground pipes (USEPA 1999) for soil absorption.

When operated under normal conditions, a typical septic tank system can be expected to remove about 50 to 70 percent of suspended material. Andreadakis (1987) noted that septic tanks (when properly operated and simulated in the laboratory) can remove over 90 percent of BOD and TSS, with ammonia nitrogen removals of up to 70 percent through nitrification, but when OSTDS are not properly sited, installed or maintained, the amount of treatment is reduced substantially, and the partially treated effluent may pose a higher potential risk to public health and the environment.

As of 2000, onsite treatment and disposal systems serve approximately 25 percent of the population of the United States. The Bureau of the Census indicated that 10 percent of those systems have stopped working (USEPA 2000), a figure that may exceed 70 percent in some communities.

A substantial portion of the over 5,200 water bodies in the United States do not meet ambient water-quality standards for their designated uses as a result of pathogens, and nearly 4,800 are impaired as a result of nutrients, causally linked to failed OSTDS (USEPA 2000). DeBorde and coworkers (1998) found that 42 percent of all waterborne disease outbreaks were associated with untreated groundwater impacted by septic tanks. It was also reported that viruses exhibited long survival times and low sorption and therefore could travel well over 100 feet in the subsurface, while remaining infectious and in quantities that could readily cause infection.

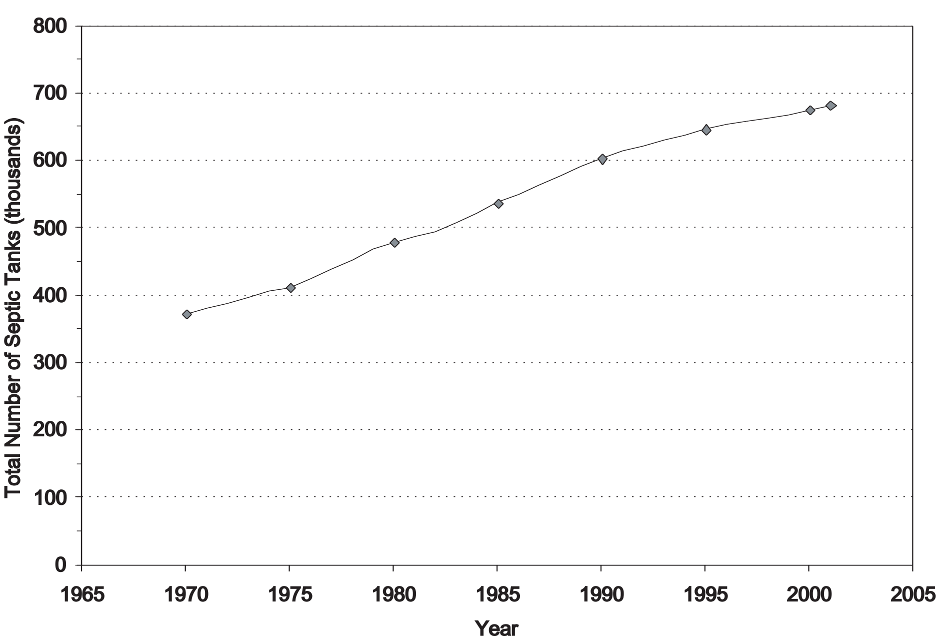

Figure 1 – Number of on site treatment and disposal systems in the Southeast Florida coastal counties (including Monroe, Miami-Dade, Broward, Palm Beach, Martin, St. Lucie, Indian River, Brevard, and Volusia) from 1970-2001 (http://www.doh.state.fl.us/environment/OSTDS/statistics/ostdsstatistics.htm).

In the year 2000, an estimated 2.3 million OSTDS were in operation in Florida, serving approximately 4.5 million people (FDEP 2001). These systems discharge over 426 million gallons per day into the subsurface soil environment. Nearly 40 percent of those systems are found along Florida's southeastern Atlantic coastline, and according to the Florida Department of Health (FDoH), the number of septic tanks installed by registered contractors has doubled over the last 30 years (see Figure 1).

Throughout the state of Florida where the water table is high (< four feet below ground surface), OSTDS have proven to be problematic from a groundwater protection/water resource perspective (Bloetscher and van Cott 1999). Impacts are traced to a lack of regulations before 1980 and to high densities of septic tanks on relatively small lots—a problem that is particularly acute in summer when the groundwater table is highest. Lack of proper maintenance is also believed to be a contributing factor.

An added complication is that many of the high-density developments, historically inhabited only in the winter months when the water table was low and performance was optimal, are now inhabited by full-time residents.

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Operational problems with septic systems generally go unnoticed until rainfall raises the groundwater table. In South Florida between the summer rainy season and late spring, the groundwater table elevation typically fluctuates from two to four feet below the ground surface; therefore, during the summer months, when the groundwater table is raised, septic tanks can not operate properly because the saturated zone is often above the drainage pipes, interfering with the designed hydraulics of the system and complicating pollutant migration modeling.

Furthermore, high loading rates during periods of saturated flow increase the linear distance that biological and chemical contaminants are transported. Subsequently, the potential for groundwater and surface water contamination is increased, particularly in porous soils and areas with high water table elevations that are characteristic of South Florida.

OSTDS failure can result in problems, including direct exposure to inadequately treated sewage, ground and surface water pollution, and contamination of shellfish beds. The most common reason for onsite systems to fail, and potentially contribute to the impairment of natural waters, is their placement in soils with insufficient assimilation capacity.

Another common reason for system failure is installation in locations where only a small percentage of the available soil is actually used because of restricted lot sizes, placement next to canals and waterways, or soil types. The lack of periodic removal of accumulated solids can also cause problems by reducing the effective volume utilized for treatment within the OSTDS. The result is an excess of effluent solids in the discharge, which leads to solids-laden effluent entering the soils and long-term drainfield plugging. Moreover, drainfield trench lengths are limited in urban areas, increasing the risk of groundwater contamination.

Scandura and Sobsey (1997) reported that septic tank/soil absorption systems that treat domestic wastewater have contaminated groundwater with enteric viruses and other pathogens, leading to outbreaks of waterborne disease. They measured the survival of an enterovirus (BE1) and fecal coliforms in sandy soils and found that viral detection in the groundwater was greater in winter than in summer; they also found that the trend was correlated to the proximity to the exfiltration lines.

Critical information is lacking to develop strategies to reduce pollutants from failed septic systems to meet Total Maximum Daily Loads (TMDLs). A 2001 task force report to the governor and the Florida Legislature recommended that that the FDoH estimate the amount of pollution that would be reduced if 45 percent of the homes with failing septic systems or septic tanks within a 100-year floodplain, documented to be contributing to receiving water body impairment, were hooked up to regional sewer systems. The FDoH, however, does not have adequate information to address this issue, and without such information, it is very difficult to accurately estimate the amount of intervention required to reduce pollution loading—even though it was concluded that septic tanks were an important source of pollutants.

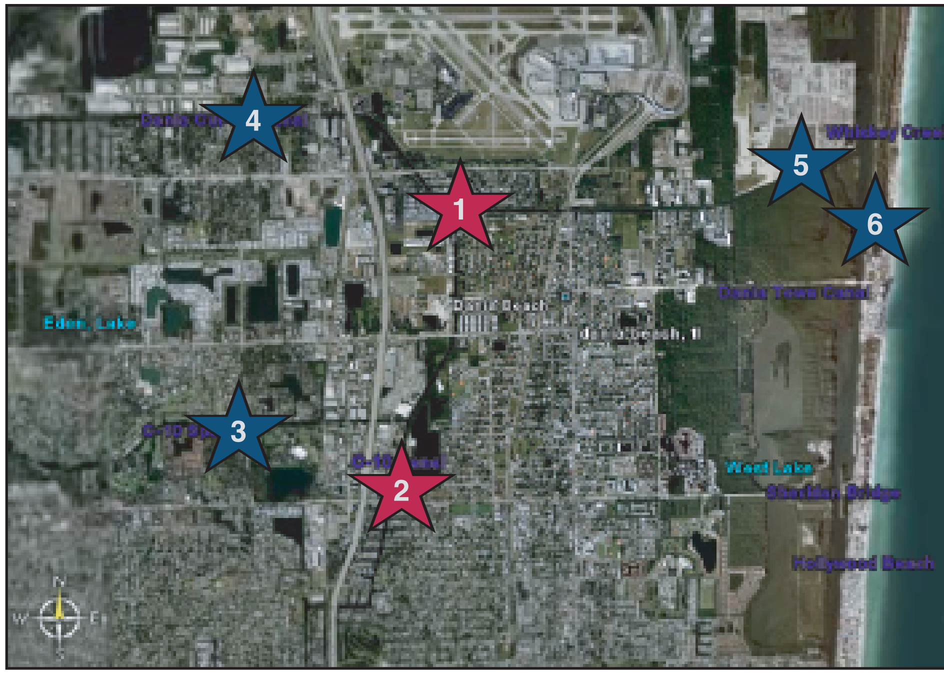

Figure 2 – Approximate locations of sampling sites in Dania Beach and Hollywood (BCDEP, 2000).

In many instances when a coastal water-quality issue arises, septic tanks are immediately included in the probable list of causes, but removing septic tanks and installing sanitary sewers is not a trivial matter. If the contribution from the septic tank turns out to be minimal, the cost burden of the conversion will be questioned, so it is clearly necessary to quantify the contribution of environmental degradation attributable to OSTDS.

The need to maintain acceptable water-quality levels is a priority that can not be ignored, so the possibility of effluent from OSTDS reaching and contaminating coastal canals and groundwater supplies is a risk that must be better understood.

The primary objective of this study was to generate data that will permit regulatory agencies and researchers to identify the potential pollutant loading contributions from OSTDS on coastal canals, specifically with regard to nutrients and pathogens indicators. The difference between seasonal high water table sampling events and seasonal low water table contributions will also be assessed, since the literature indicates that groundwater table elevation and precipitation affects the quantity and movement of nutrients and pathogens.

The hypothesis is that there are periods when OSTDS are directly impacting the water quality of coastal canals and that there are factors that can be developed to verify this contribution and differentiate it from other anthropogenic sources.

Methodology

To study the impacts of OSTDS on coastal canals, two similar sites of single-family housing units along the Dania Cut-Off/C-10 Canal were selected. The Dania Cut-Off/C-10 Canal, located in Broward County, controls the drainage for the cities of Hollywood and Dania Beach and reaches the Atlantic Ocean near the Port Everglades area.

The Dania Cut-Off/C-10 Basin is located in the southeast corner of Broward County, just south of the Fort Lauderdale International Airport. The main portion of the canal flows eastward (with outgoing tides) and receives controlled releases of freshwater from the S-13 water control structure on the eastern C-11 Canal and some tidal input from the South Fork of the New River.

The C-10 Canal originates in western Hollywood and connects to the eastern section of the Dania Cutoff Canal just east of Interstate Highway 95. Stormwater is the main freshwater input to the C-10, since no major western freshwater canals discharge to this water body (DPEP 2001). The area under study is characterized by sandy soils and tidally influenced coastal canals that contain brackish water.

Water-quality data from a recent charac-

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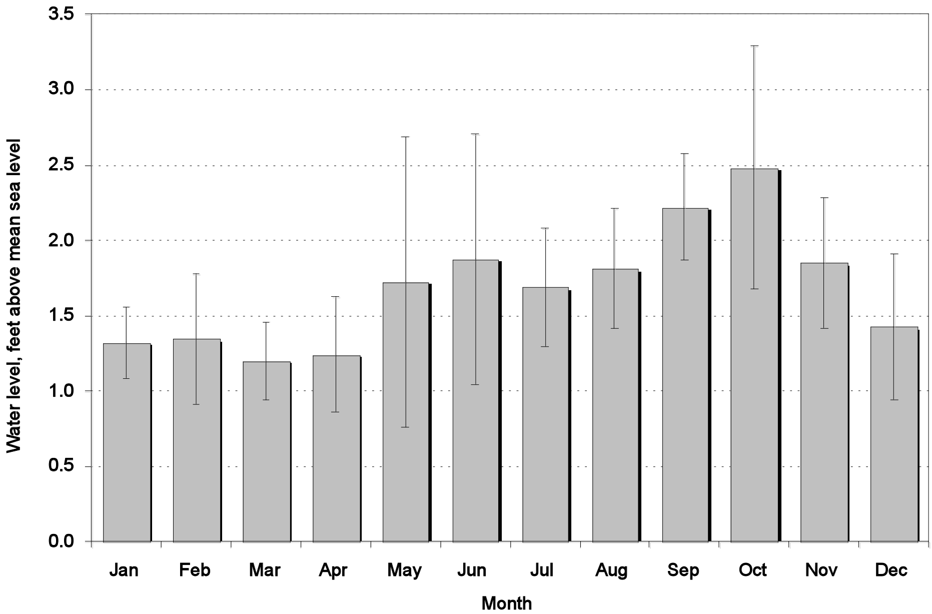

Figure 3 – Average monthly groundwater levels in Broward County during 1995-2004 (USGS 2004).

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terization survey conducted by the city of Hollywood in the upper C-10 Canal region were used to select suitable sampling sites for the paired analysis. The Hollywood site is connected to the public sewer system, and the other site relies solely on OSTDS. For confidentiality reasons, a map pinpointing the exact locations of the sampling sites can not be included here, but an overall map with the general locations of the six sampling points is presented in Figure 2. The number on the map corresponds to the following sites:

1. Site with OSTDS, Melaleuca Gardens (Dania)
2. Site with sewer, Sherman Street (Hollywood)
3. Upstream site, North Park Bridge (Hollywood)
4. Upstream site, SW 30th Avenue (Hollywood)
5. End of Dania Cut-Off Canal (Hollywood)
6. Dania Beach marine sampling point (Dania Beach)

A site characterization was performed in order to collect information on water use, geology, hydrogeology, tidal impacts, and climate for the selected sampling sites. Using GIS data and aerial photographs of the paired sites, sets of sampling locations were selected in isolated reaches of the Dania Beach Cut-Off Canal and the C-10 spur, as well as upstream locations for background correction (in natural areas) and downstream locations (including the beach) for evaluation of coastal impacts.

An analysis of U.S. Geological Survey groundwater table elevation data from 1995-2004 (USGS 2004) for several monitor wells located in Hollywood showed that the highest water table elevations typically occur in October or November and the lowest levels occur during the first few months of the year (Figure 3). Thus, the seasonal high water table field study was conducted from October to November to coincide with the peak of the seasonal high water table, and a second sampling campaign was conducted from February to March to coincide with the height of the seasonal low water table.

An analysis of monthly averaged precipitation data from 1977-present showed that the highest expected rainfall typically begins in the summer months and extends into the fall (May – October). The driest times of the year typically occur during the winter and spring from December to April (Figure 4). Thus, during the dry season, sampling was expected to show only a runoff contribution related to irrigation.

In coastal areas, changes in surface water levels caused by tidal movements affect groundwater flows. A periodic response in the water table follows the tidal cycle, resulting in a net outflow of groundwater during outgoing tides. Significant negative correlations with tidal level and *Enterococci*, coliphage, and fecal coliform, in descending order, have been reported (Lipp 1999), so sampling times were carefully chosen to coincide with the outgoing tide.

The goal of the water-quality monitoring program for the paired sites was to quantify the potential pollutant loading contributions from on-site treatment and disposal systems, specifically regarding nutrient and microbial pathogen indicators; therefore, at each location the following parameters were monitored using Standard Methods (APHA et al. 1995):

- pH, temperature, conductivity, salinity, DO, and TDS (YSI 556 probe)
- Secchi depth (Secchi disk)
- Channel depth (surveying rod)
- Nitrate: Hach® Nitrate Probe Model 5920 (SM4500-NO3-D)
- Total Coliform / E. coli : MMO-MUG, IDEXX Colilert™ Test (SM9223B)
- Enterococcus: IDEXX Enterolert™ test (SM9230C)
- COD: Hach® closed reflux, colorimetric method (SM5220D)

Field measurements were collected using a non-powered boat to prevent any cross-contamination and unintended mixing of the water column. General parameters for each location were measured on-site using a YSI 556 multi-parameter probe, calibrated daily. Duplicate samples at each location were col-

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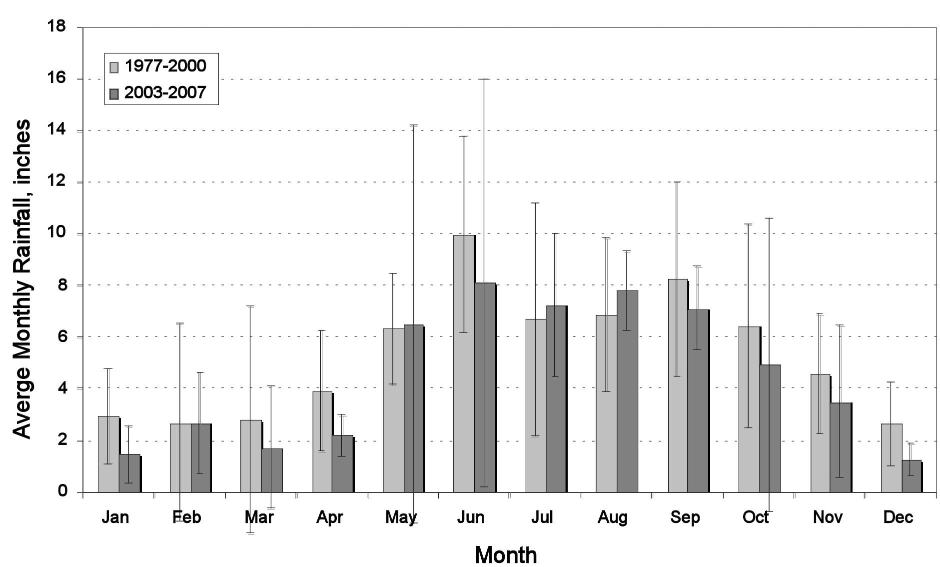

Figure 4 – Average monthly precipitation levels at the Fort Lauderdale International Airport during 1977-2000 and from 2003 to present.

Site	Relative Distance from Channel	pH	Temp (°C)	Salinity (‰)	DO (mg/L)
Site w/ OSTDS	1	7.8 - 8.1	25.0 - 28.0	23.3 - 23.7	4.7 - 5.7
	2	7.9 - 8.1	25.2 - 28.0	23.0 - 23.8	4.8 - 5.7
	3	7.9 - 8.1	25.2 - 28.0	22.9 - 23.8	4.8 - 5.7
	4	7.9 - 8.1	25.2 - 28.0	22.4 - 22.5	4.8 - 5.7
	5	7.9 - 8.1	25.2 - 28.0	22.4 - 23.1	4.8 - 5.7
Site w/ sewer	1	7.6	24.8	5.6	5.4
	2	7.6	24.8	5.6	5.5
	3	n/a	n/a	n/a	n/a
	4	7.6	24.6	5.6	5.5
	5	7.6	24.5	nr	5.6
Upstream	1	7.3	24.8	2.2	5.6
	2	7.4	24.7	2.1	5.6
Background	1	n/a	n/a	n/a	n/a
Beach	1	n/a	n/a	n/a	n/a
	2	n/a	n/a	n/a	n/a

Site	Relative Distance from Channel	pH	Temp (°C)	Salinity (‰)	DO (mg/L)
Site w/ OSTDS	1	7.8	29.3	25.6	4.5
	2	7.8	29.2	24.2	4.4
	3	n/a	n/a	n/a	n/a
	4	7.8	29.3	24.8	4.8
	5	7.8	29.3	25.3	4.1
Site w/ sewer	1	n/a	n/a	n/a	n/a
	2	7.5	29.3	10.1	3.8
	3	nr	29.9	9.6	4.6
	4	7.4	30.2	9.9	3.7
	5	7.4	30.2	9.8	3.8
Upstream	1	nr	29.0	4.5	8.1
	2	8.4	30.2	3.9	nr
Background	1	7.5	29.6	4.7	4.3
Beach	1	7.8	29.5	25.1	4.7
	2	8.1	29.3	30.1	8.7

Table 1 – Summary of water quality measurements taken during the seasonal high water table sampling campaign (at left is 2004 data; at right is 2006 data).

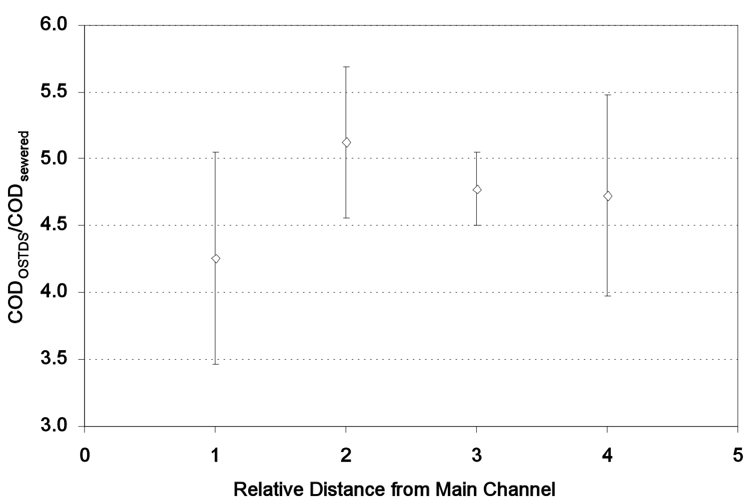

Figure 5 – Chemical oxygen demand ratios at different distances from the main channel of the canal.

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lected using 500 mL sterile Whirl-Pak bags at a constant depth of 30 centimeters (one foot). Following collection, samples were maintained at 4°C to preserve their integrity during transport and in the laboratory.

Samples were analyzed within 24 hours for total coliform, *E. coli*, *Enterococcus*, and COD. After the first 24 hours, concentrated H_2SO_4 was added (2 mL per liter of sample) to preserve samples for further analysis, as necessary. During testing for COD, an important interference due to elevated chloride concentrations was encountered. This was addressed by modifying methods according to published standard operating procedures.

For example, each COD vial contains enough mercuric sulfate to eliminate chlo-

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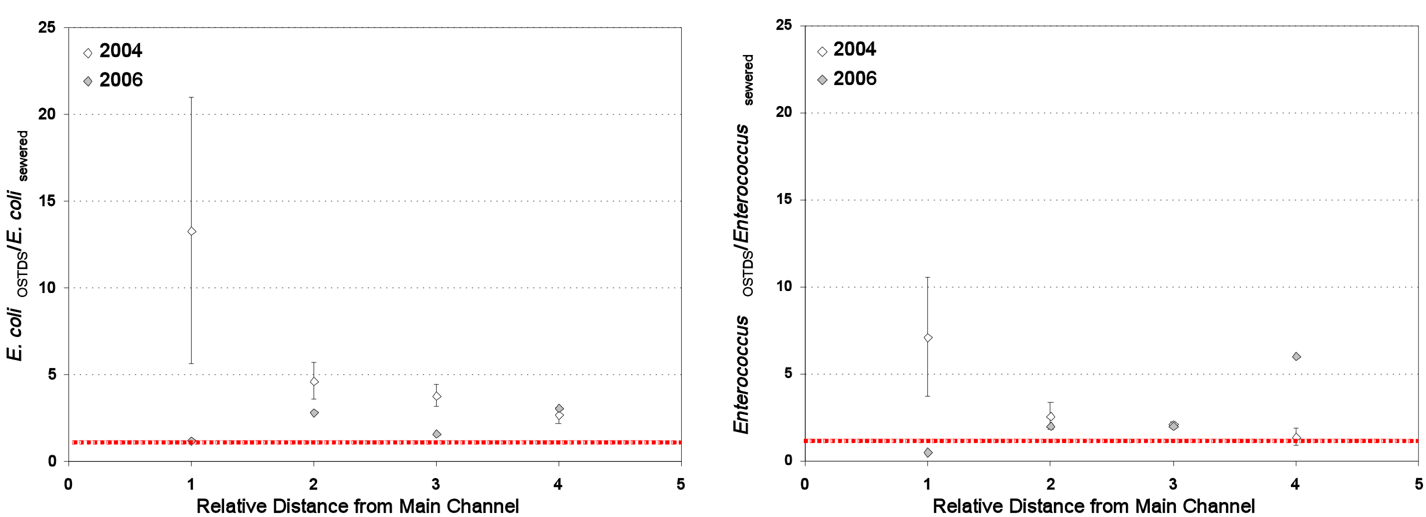

Figure 6 – Microbial concentrations at the OSTDS site normalized to microbial concentrations at the sewered area during the seasonal high water table sampling event (left: *E. coli* ratio, right: *Enterococcus* ratio). Note the dotted line shows where the ratio = 1.0.

Site	Relative Distance from Main Channel	Ec/Es Ratio
Site w/ OSTDS	1	70
	2	69
	3	n/a
	4	81
	5	44
Site w/ sewer	1	30
	2	49
	4	102
	5	86
Upstream, sewered or open space	1	2.5
	2	5.0

Table 2 - Summary of Ec/Es Ratios for 2006 SHWT event.

Site	Relative Distance from Main Channel	NO_3-N (mg/L)
Site w/ OSTDS	1	16.8
	2	19.1
	3	18.0
	4	15.9
	5	17.3
Site w/ sewer	1	5.8
	2	8.4
	4	9.6
	5	11.0
Upstream, sewered or open space	1	3.8
	2	7.3

Table 3 – Summary of nitrate-nitrogen concentration measurements for the seasonal high water table sampling event.

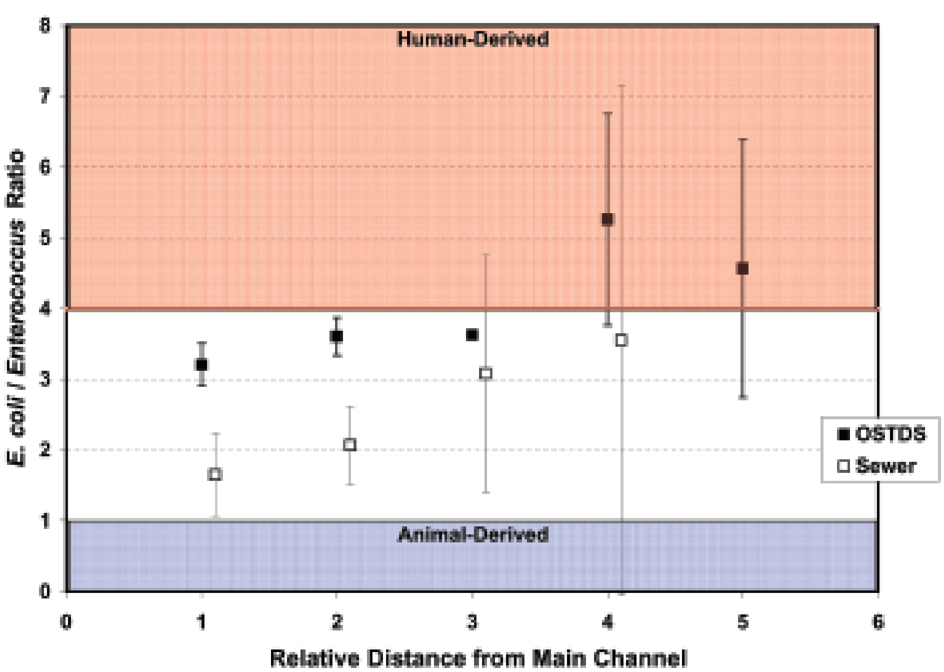

Figure 7 – *E. coli/Enterococcus* ratio at OSTDS site versus sewered area for the seasonal high water table sampling event (values above the line at Ec/Es = 4 indicate human-derived pollution; values below the line at Ec/Es = 1 indicate animal-derived pollution). The findings of SEFLOE may explain the decreases as the main channel is approached.

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ride interference up to 2,000 mg/L. If 0.50 g of mercuric sulfate ($HgSO_4$) is added to each COD vial, the additional mercuric sulfate will raise the maximum chloride concentration allowable to 4,000 mg/L. Thus, samples were diluted to below 4,000 mg/L chloride and mercuric sulfate was added to account for the chloride interference, particularly for samples collected during the seasonal low water table sampling event.

Results & Discussion

Seasonal High Water Table Sampling

A total of 64 samples were taken during the seasonal high water table event in three sampling trips from 10/20/2004 to 11/10/2004. A follow-up sampling trip (13 samples) was conducted on 10/23/2006. The results of water-quality measurements are presented in Table 1.

In terms of canal geometry, water temperature, and pH, no differences were noted between the paired sites, but the OSTDS site showed elevated salinity, conductivity, and TDS with depressed levels of dissolved oxygen compared to the sewered site. This suggests an important tidal influence and potentially a runoff contribution in addition to the expected OSTDS contribution. The higher TDS and conductivity were not unexpected because of the closer proximity of the OSTDS site to the ocean and the order of sampling with respect to the tidal condition.

The slightly lower dissolved oxygen levels observed at the OSTDS site (2004 data) indicate potential contamination due to sewage inputs. Measurements of COD were used to investigate this further. The ratio of the concentration found in the OSTDS site to the concentration found in the sewered area was plotted in Figure 5. A ratio larger than unity indicates elevated COD levels in the OSTDS areas.

The mean ratio observed was between 4.2 and 5.1 and suggests at least a four-fold difference between the OSTDS site and the sewer site, in terms of organic constituents (COD). These ratios strongly indicate a potential contribution directly attributable to OSTDS.

With respect to microbial pathogen indicators, elevated coliform counts were expected from previous work (Bloetscher and van Cott 1999), and preliminary testing revealed that total coliform counts were on the order of 10^5/100 mL for all sampling locations; therefore, *E. coli* and *Enterococcus* species were monitored (Figure 6). For both indicator species, the highest microbial density generally corresponded to the most isolated reaches of the canals (relative distance = 4), but as the distance from the main channel decreased (relative distance = 1), the value of the microorganism counts in the OSTDS areas decreased to the main canal levels.

As a result, when normalized to the microorganism counts in the sewered areas, the OSTDS area increased in an exponential fashion in comparison to the centrally sewered areas, suggesting an adverse contribution from OSTDS. In 2006 we did not see this trend, which may have something to do with the recent hurricanes.

It has been suggested that the quantities of fecal coliform (FC) and fecal streptococci (FS) that are discharged by humans are significantly different from those discharged by animals (Tchobanoglous et al. 2003). From previous work, the FC/FS ratio is typically less than 1 for animals and greater than 4 for humans. Ratios in the range from 1 to 2 typically indicate that the pollution is derived equally from human and animal sources.

Fecal coliform and fecal streptococci were not analyzed for in this study, but *E. coli* is a fecal coliform and *Enterococcus* is a fecal streptococcus, so an *E. coli*-to-*Enterococcus* ratio (Ec/Es) was plotted, as shown in Figure 7, in an attempt to establish the origin of the observed contamination.

As a reference, the human-derived input cut-off value at 4 is plotted. As clearly indicated, the OSTDS areas exhibit ratios between 3 and 5, suggesting an important human-derived contribution. In the sewered area, the ratios were more indicative of animal-derived pollution, but the ratios increased as the distance from the main channel increased. This is likely due to stagnant conditions in those isolated reaches.

The Southeast Florida Ocean Outfall Experiment (SEFLOE) showed that the marine die-off of fecal coliforms is greater than that of fecal streptococcus, which may suggest that the salinity of the water in the canals causes the decreased ratio as the water moves toward the main canal (Hazen and Sawyer 1994).

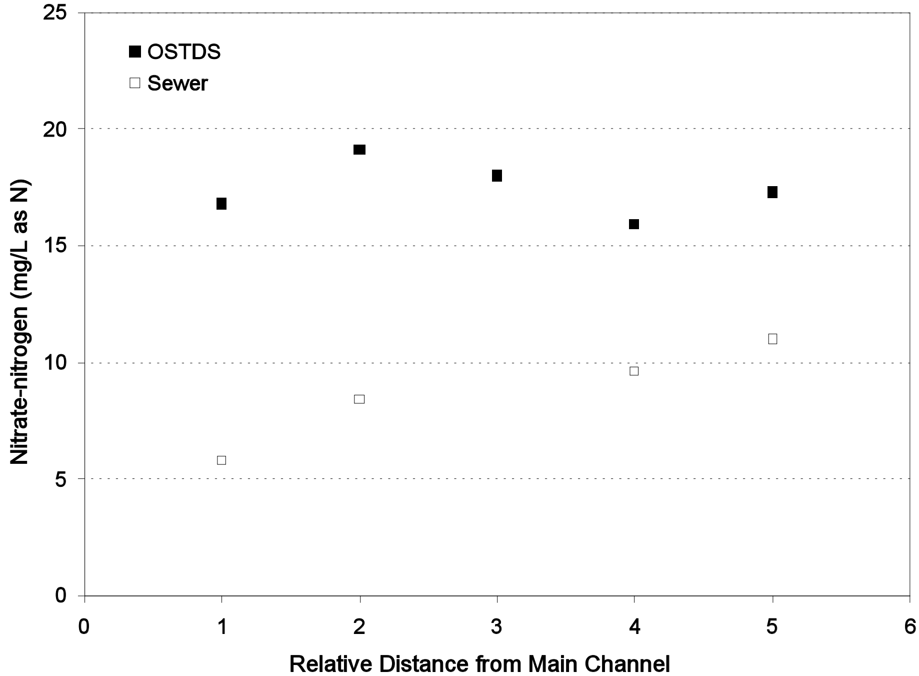

Figure 8 – Nitrate-Nitrogen concentration at different distance from the main channel of the canal.

In 2006 when the sampling was repeated the Ec/Es ratios were all above 30, as shown in Table 2 . The relative absence of Enterococcus (below detection for 4 of 8 sites) does not allow the ratio parameter to reveal any significant information. Moreover, the measured E. coli levels were so high, that the ratio results for the SHWT sampling are difficult to interpret, and more study is needed to validate this finding.

The results of nitrate-nitrogen analyses are presented in Table 3 and Figure 8. For the OSTDS area, nitrate-nitrogen concentrations between 15.9 to 19.1 mg/L were measured. For the sewered area, nitrate-nitrogen concentrations oscillated between 5.8 to 11.0 mg/L. At the upstream site, a maximum value of 7.3 mg/L was detected.

In Figure 8, nitrate-nitrogen concentrations versus their relative distance from the main channel were plotted. Concentrations in the site with OSTDS were always higher than the concentrations found in the corresponding sewered areas.

For the sewered area, an increase in nitrate-nitrogen concentration with the distance from the main channel is suggested, whereas for the OSTDS area, no significant trend can be noted. Ammonia is a more likely nitrogen component of both OSTDS and raw sewage, so the results may be more asso-

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Site	Relative Distance	pH	Temp (°C)	Salinity (‰)	DO (mg/L)
	1	7.9-8.3	20.8-22.8	36.6-37.7	6.1-7.4
	2	8.0-8.2	20.8-22.8	36.6-37.5	6.1-7.4
	3	8.0-8.2	20.8-22.8	36.5-37.3	6.1-7.4
	4	8.0-8.2	20.8-22.8	36.4-37.5	6.1-7.4
Site w/ OSTDS	5	8.0-8.2	20.8-22.8	36.9-38.5	6.1-7.4
	1	7.6-8.0	20.6-23.3	20.8-26.4	6.6-8.1
	2	7.6-8.0	20.6-23.2	20.8-26.2	6.4-7.1
	3	7.6-8.1	20.8-23.5	20.4-26.2	6.4-7.1
	4	7.6-7.9	20.8-23.2	21.1-26.3	6.4-7.9
Site w/ sewer	5	7.6-7.9	21.0-23.8	18.9-25.9	6.2-7.9
	1	7.7-7.8	20.6-22.9	9.3-17.7	7.0-8.5
Upstream	2	7.6-7.8	20.8-22.9	9.3-18.1	7.0-8.3
Background	1	7.9-8.0	22.6-23.4	15.9-33.5	7.4
	1	8.1-8.2	21.0-23.1	36.5-38.6	7.3
Beach	2	8.3	21.1-22.4	39.4-41.3	7.2

Site	Relative Distance	pH	Temp (°C)	Salinity (‰)	DO (mg/L)
	1	7.9	23.5	31.6	5.5
	2	7.7	24.3	30.4	5.6
	3	n/a	n/a	n/a	n/a
	4	7.8	23.7	30.2	5.6
Site w/ OSTDS	5	7.8	23.5	31.1	5.7
	1	n/a	n/a	n/a	n/a
	2	7.5	24.0	19.6	5.9
	3	7.6	24.1	20.6	5.7
	4	7.5	24.7	20.3	5.4
Site w/ sewer	5	7.4	25.2	20.4	5.3
	1	7.5	24.5	14.7	5.9
Upstream	2	7.3	24.5	14.5	6.0
Background	1	7.5	25.2	19.8	5.4
	1	7.7	24.4	32.3	5.3
Beach	2	7.8	23.9	34.5	5.3

Table 4 – Summary of water quality measurements taken during the seasonal low water table sampling campaign (at left is 2004 data; at right is 2006 data).

Table 5 – Salinity levels taken from the paired sites over the course of the monitoring study.

Sample Location	Salinity (mg/L)	
	SHWT	SLWT
Site w/ OSTDS	17,000 – 25,000	30,000 – 37,000
Site w/ sewer	2,500 – 6,000	19,000 – 25,000
Upstream site	2,000 – 2,100	9,000 – 18,000

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ciated with lawn care (surface runoff) inputs, rather than OSTDS.

In terms of nutrients, nitrate-nitrogen testing revealed that levels measured in the OSTDS site were approximately twice the levels observed in the sewered areas. Phosphorus was not monitored because minimal migration was expected from typical residential OSTDS in calcium carbonate rich soils (Postma et al. 1992).

<u>**Seasonal Low Water Table Sampling**</u>

A total of 97 samples were taken during the seasonal low water table event in three sampling trips from 02/07/2005 to 03/08/2005. A follow-up sampling trip (13 samples) was conducted on 3/29/2006. OSTDS were expected to perform better during the seasonal low water table elevation, which coincided with a dry event; therefore, if the septic tanks in the test site were operating efficiently, major differences in water-quality parameters between the two sites were not anticipated. The results of water-quality measurements taken during the seasonal low water table sampling program are presented in Table 4.

In terms of water temperature and pH, no differences were noted among the sites. Secchi depth values (Meeroff and Morin 2005) were lower than during the seasonal high water table event, indicating higher turbidity and increased algal growth from nutrient enrichment. As was observed during the seasonal high water table monitoring program, the OSTDS samples showed elevated conductivity and TDS compared to their sewered counterparts, but dissolved oxygen levels were similar between the paired sites.

Dissolved oxygen concentrations for the seasonal low water table sampling event were generally higher than those measured during the seasonal high water table sampling program. This is indicative of less organic pollution or higher photosynthetic activity, although turbidity levels were found to be higher, so COD measurements were taken to give an indication of the organic concentrations in the canals.

Initial COD results were difficult to interpret because the salinity increased markedly from the seasonal high water table to the seasonal low water table (refer to Table 5), resulting in a chloride interference with the colorimetric COD test. Even after pretreatment with mercuric sulfate to precipitate out the chloride interference, COD measurements remained inconclusive. Follow-up studies have used TOC measurements instead; results will be made available at a later date.

With respect to microbial pathogen indicators, both *E. coli* and *Enterococcus* species were monitored as done previously for the seasonal high water table sampling event (Figure 9). For both indicator species, the highest density corresponded to the most isolated reaches of the canals (relative distance = 4 – 5), but as the distance from the main channel decreased, the ratio did not vary as much as seen during the seasonal high water table (compare with Figure 6). Also, the values were much closer to unity (1.1 – 1.2) as the sampling location moved closer to the main channel, suggesting a minimal contribution from OSTDS.

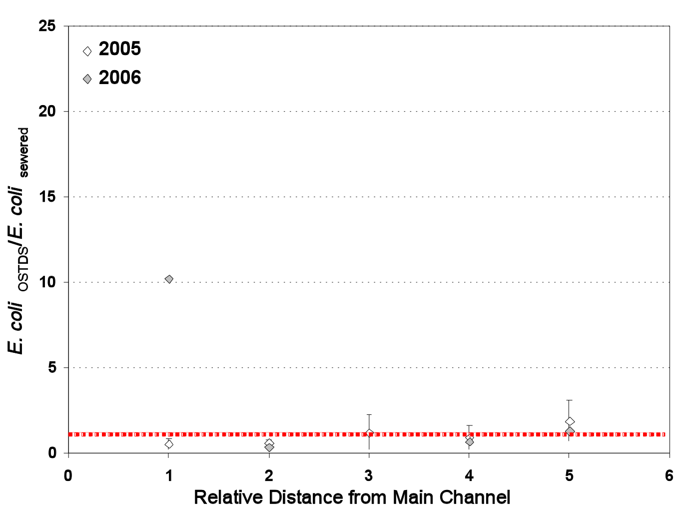
At least a five-fold difference in *E. coli* levels between the OSTDS site and the sewer site was observed during the seasonal high water table sampling event (see Figure 6), but during the seasonal low water table, *E. coli* concentrations measured at both sites were similar, albeit elevated. In 2006, only one sampling site deviated from this pattern (*E. coli* relative distance = 1, which was likely contaminated with pet feces). If the observed difference between the *E. coli* levels during the seasonal high water table is attributable to a contribution from OSTDS, then the effect of septic systems is nearly undetectable during the seasonal low water table.

With respect to *Enterococcus* levels, a three-fold difference between the OSTDS site and the sewer site was observed during the seasonal high water table sampling event. Whereas the *Enterococcus* concentrations in the OSTDS site decreased by 75 percent from the seasonal high water table to the seasonal low water table, the *Enterococcus* levels observed in the sewer site remained relatively unchanged.

Furthermore, measured concentrations were essentially the same between the paired sites during the seasonal low water table. If the difference between the *Enterococcus* levels during the seasonal high water table is attributable to a contribution from OSTDS, then the effect of septic systems is clearly decreased during the seasonal low water table.

To differentiate the potential sources of pollution in the canals, the Ec/Es ratio was investigated. Results are plotted in Figure 10. It is interesting to note that all ratio values are well above the human-derived input cut-off (Ec/Es > 4).

During the seasonal low water table sampling events, high levels of *E. coli* (>400



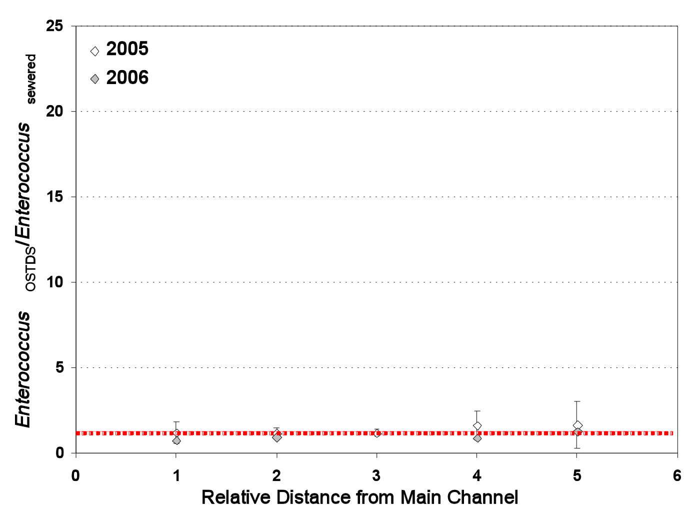


Figure 9 – Microbial concentrations at the OSTDS site normalized to microbial concentrations at the sewered area during the SLWT sampling event (left: *E. coli* ratio, right: *Enterococcus* ratio). Note the dotted line shows where the ratio = 1.0.

MPN/100 mL) were detected, along with moderate-to-low levels of *Enterococcus* (<100 MPN/100 mL); therefore, all Ec/Es ratios were greater than approximately 10. Normally such high ratios would be indicative of an important human-derived contribution at both sites, but since the measured *E. coli* levels were so high, the ratio results for the seasonal low water table sampling are difficult to interpret, and more study is needed to validate this finding.

In terms of nutrients, nitrate-nitrogen concentrations at the OSTDS site showed a net decrease between the seasonal high and seasonal low water tables, while approximately the same levels were detected in the sewered area. For the most part, however, nitrate-nitrogen testing results (not shown) were difficult to interpret because of the variable salinity levels in the tidally-influenced canals and interference from fertilizer run-off from golf courses within the watershed in the upstream reaches.

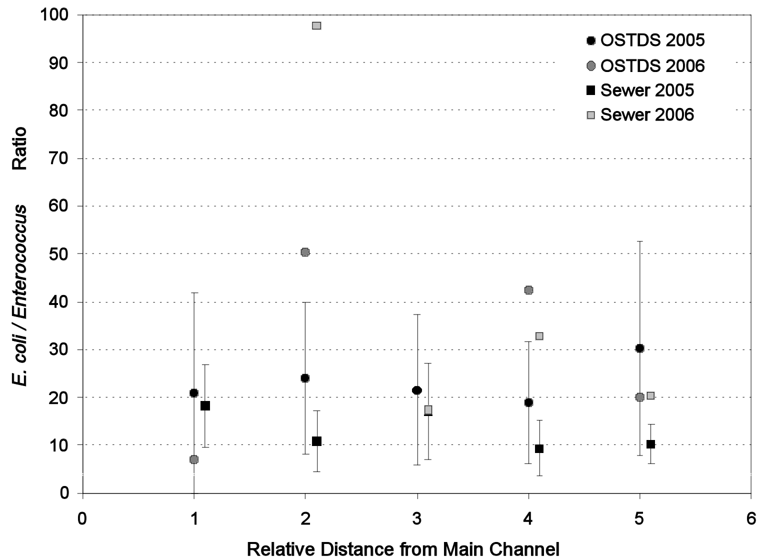


Figure 10 – *E. coli/ Enterococcus* ratio at OSTDS site versus sewered area for the dry season sampling event (values above the line at Ec/Es = 4 indicate human-derived pollution).

Conclusions

The seasonal high water table event field study showed obvious differences between the OSTDS and the sewered area in terms of microbial pathogen indicators, nutrients, and COD. The analysis of field data indicates that the source of the differences is likely due to human-derived inputs, principally from the OSTDS.

With respect to *Enterococci*, OSTDS may contribute up to approximately 50 percent of the total concentration during the seasonal high water table, but negative impacts were not observed at the adjacent beaches during that same testing period (Meeroff and Morin 2005).

The pollutant contribution of OSTDS on coastal canals is evident, but their impact on the marine environment is uncertain. The travel time of the canal water through the intracoastal waterway to the inlet and eventually to the beaches is relatively long, so significant die-off and dilution is anticipated.

During the seasonal low water table sampling event, OSTDS were expected to perform better than during the seasonal high water table, so major differences in water quality parameters between the paired sites were not anticipated and were not observed. During the seasonal low water table sampling campaign, equivalent levels of nutrients and pathogen indicators were observed at both sites.

Surprisingly, the sewered areas showed approximately the same levels of pollution indicators over the two sampling periods, but the OSTDS sites showed a net *decrease* over the same period. The principle indication is that the OSTDS appear to operate properly during the seasonal low water table and likely do not adversely contribute to environmental degradation.

It was interesting to note that the fecal pollution indicators were measured at higher densities during the seasonal low water table in some of the sampling sites. In some cases, the sewered areas showed much higher levels than the OSTDS sites. This would seem to indicate another source, perhaps an animal-derived input.

In general, the water quality was lower during the seasonal low water table, likely due to lower flows and dilution volumes due to the absence of precipitation. Another possible explanation could be that "over-irrigation" runoff during the seasonal low water table transports animal-derived feces and nutrient fertilizers into the canals.

Again, despite high levels of *Enterococcus* detected during some of the seasonal low water table sampling events, concentrations measured at the beach by our study and the FDoH beach monitoring program did not correlate with higher levels found in the coastal canals that discharge near the beach. Thus, the pollution indicators in the coastal canal did not appear to influence the water quality of the beach.

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