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Pollutant impacts to Cape Hatteras National Seashore from urban runoff and septic leachate

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ABSTRACT

The sandy barrier islands of Cape Hatteras National Seashore, USA, attract large seasonal influxes of tourists, and are host to numerous motels, rentals and second homes. To investigate the impacts of nearby urbanization on public trust waters, sampling was conducted in nine brackish water bodies within this coastal national park. A large tidal urban ditch delivered runoff-driven fecal-contaminated water directly into public beach waters. At all sites except the control, ammonium, phosphorus and fecal bacteria concentrations were high, strongly seasonal and significantly correlated with community water usage, indicating that increased septic tank usage led to increased pollutant concentrations in area waterways. Nutrients from septic systems caused ecosystem-level problems from algal blooms, BOD, and hypoxia while fecal microbes created potential human health problems. Septic system usage is widespread in sensitive coastal areas with high water tables and sandy soils and alternatives to standard septic systems must be required to protect human health and the environment.

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1. Introduction

Protected coastal areas, such as national, state, provincial or local parks offer public opportunities for relaxation, recreation, and nature appreciation. Under idealized conditions these areas also present researchers with a rich variety of terrestrial and aquatic habitats for study under (relatively) unimpacted conditions. However, some of these parks have suffered from impaired water quality from adjacent or upstream human sources, especially agriculture. One example is the Everglades National Park in Florida, which receives fertilized runoff and other pollutants from agriculture, primarily from sugar cane farming adjacent to the parklands (Lapointe and Barile, 2004; Sheikh and Johnson, 2004; Perry, 2008). Cumberland Island National Seashore in Georgia is an example of a coastal park whose adjacent waterways are impacted by municipal and industrial point-source discharges, as well as non-point source fertilizers and animal waste, from upstream municipalities and agricultural areas in the Satilla and St. Mary's River basins (Alber et al., 2005). Nearby urbanization is also a potential pollution sources for coastal parks. With the rapid human urbanization of coastal zones in recent decades (Crossett et al., 2004; US EPA, 2004) areas where protected land and waterscapes are in direct contact with already-developed or urbanizing situations can present case studies from which the impacts of such encroachment on natural areas can be assessed and (hopefully) learned from by the scientific, management, and political communities. One such area where urban impacts can have direct impacts upon undeveloped adjoining areas is Cape Hatteras National Seashore (CAHA) located along the Outer Banks of North Carolina, USA.

South Bodie Island is a barrier island that forms the northernmost section of Cape Hatteras National Seashore (Fig. 1). The park wetlands in the area contain significant fresh and brackish water resources (Cole and Bratton, 1995; Mallin et al., 2006a). Adjoining CAHA is the Town of Nags Head, with a residential area extending south along the seashore next to CAHA for about 7.3 km (Fig. 1). This part of Nags Head is zoned R-2, or medium density residential (Stone Environmental Inc., 2005); dwelling density near the beach in Nags Head averages 7.5 units/ha or 3 units/acre (Town of Nags Head, 2000). Nags Head seasonally has high summer visitor usage, and sewage treatment in the area abutting the park consists of septic systems. The soils in this area of Nags Head are primarily highly permeable sands while the water table in this area is mainly 0–1 m from the surface (Stone Environmental Inc., 2005). This combination of geological and hydrological factors can lead to ineffective treatment of human sewage by septic systems (Cogger et al., 1988), which require between 2 and 5 feet (0.6–1.5 m) of aerated soil (the vadose zone) beneath the septic drainfield to completely treat the pollutants in sewage (US EPA, 2002). Otherwise, pollutants such as fecal bacteria and nutrients (nitrogen and phosphorus) can enter the upper groundwater table and move laterally up to several hundred feet until they enter surface water such as a ditch, pond, tidal creek, bay or sound (Lipp et al., 2001a,b; US EPA, 2002). Evans and Houston (2000) reported that septic leachate from drainfields

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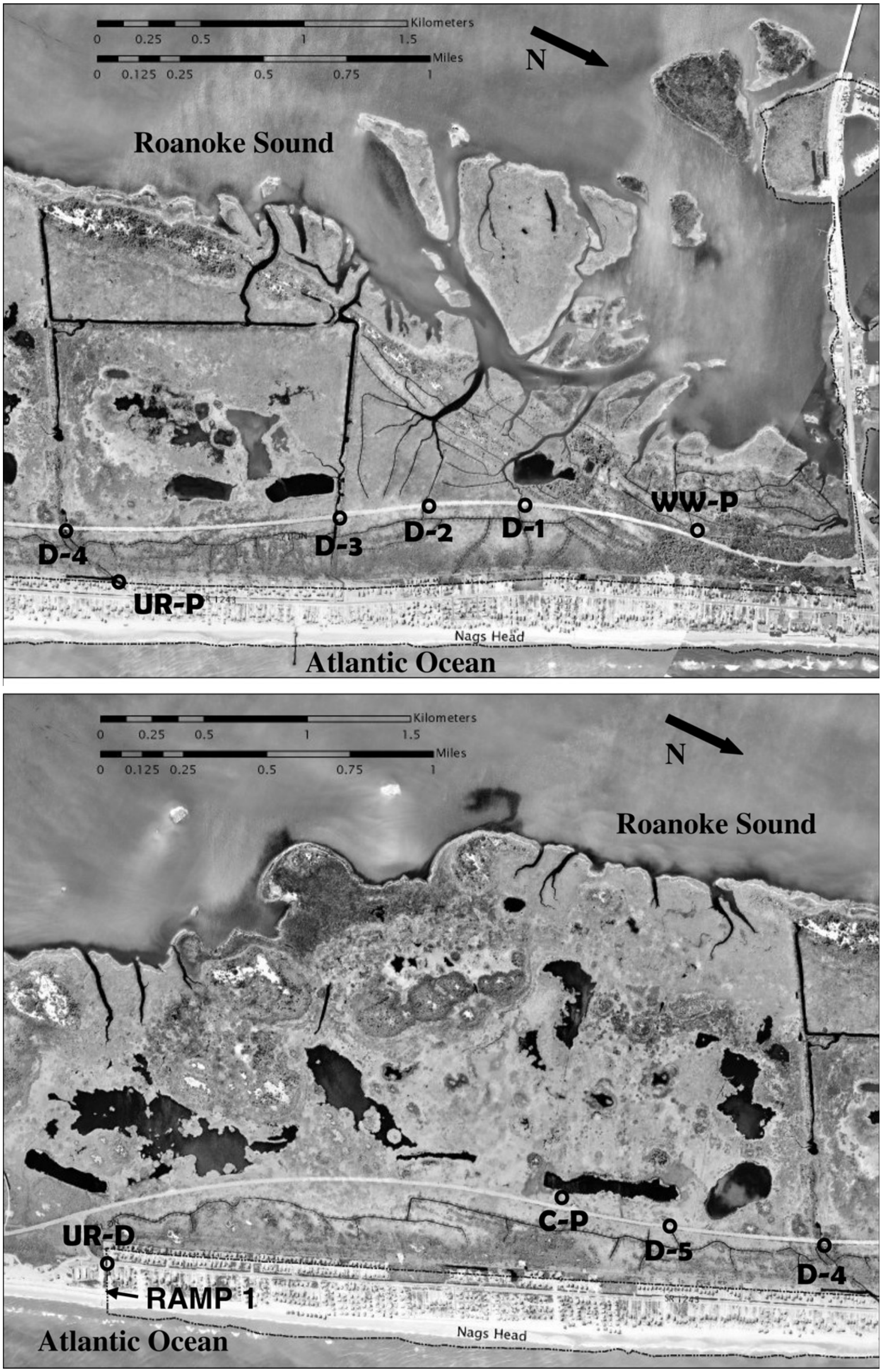

Fig. 1. Map of sampling sites on South Bodie Island, Cape Hatteras National Seashore, North Carolina, USA. Upper panel – northern area; lower panel – southern area.

in Nags Head increased nitrate concentrations in downslope groundwater wells, with the wells nearest the drainfields having nitrate concentrations that exceeded the US Environmental Protection Agency drinking water standard of 10 mg/L.

A ridgeline passing north to south through Nags Head delineates the drainage direction (Evans and Houston, 2000; Fig. 1). Water (surface runoff and upper groundwater) moves from the ridgeline in Nags Head into a series of drainage ditches and tidal creeks that run perpendicular to Old Oregon Inlet Road and flow westward, pass under Highway 12, and drains into Roanoke Sound (Fig. 1). In a comparison of ditch water quality and marsh sites within CAHA west of Nags Head, Cole and Bratton (1995) found that drainage ditches had significantly higher nutrients and fecal bacteria, but lower dissolved oxygen, than the marsh sites. East of the ridgeline, much of the surface runoff in the part of Nags Head adjoining CAHA flows into a 5 m wide ditch that parallels the west side of Old Oregon Inlet Road. This ditch flows south and enters the ocean at an outfall at Ramp 1 of CAHA beach property (Fig. 1). High fecal bacteria counts from this outfall pollute the beach and have led to beach postings and closures by the N.C. Shellfish Sanitation Section (Mallin et al., 2006a). This large drainage ditch also hosts dense algal blooms and macrophyte coverage, suggesting inputs of nutrients. Thus, stormwater runoff, as well as septic leachate, was considered a potential problem impacting adjacent public areas.

Development has continued in Nags Head (as well as in other municipal areas adjoining CAHA property on barriers further south); thus comprehensive water quality information was needed to assess potential anthropogenic inputs on these drainage systems and associated natural areas. We hypothesized that: (1) based on previous data, topography and local hydrology, elevated pollutant loads were draining from the adjacent urbanized areas into ditches, tidal creeks, wetlands and beach waters on CAHA property, either through overland runoff or septic system leachate, (2) these pollutants were likely leading to associated ecosystem impacts caused by such pollutant loads, and (3) such pollutants also present a human health problem to park users. To investigate these hypotheses we obtained physical, chemical, and biological water quality data for tidal creek/ditch and brackish pond sites on, or impacting NPS property in the South Bodie Island area adjoining northern Cape Hatteras National Seashore. We assessed the water quality based on regulatory agency standards and published academic data collected from regional tidal creek ecosystems (Sanders and Kuenzler, 1979; MacMillin et al., 1992; Lewitus et al., 2004; Mallin et al., 2004). We then utilized statistical techniques to assess sources of pollutant loading to the various impacted aquatic systems. With barriers and other coastal islands serving as prime areas of coastal real estate, it is hoped that information and lessons learned from this study can be applied to guide management efforts in coastal areas yet to be developed.

2. Study area

Several large, brackish, tidally-influenced drainage ditches and creeks flow westward from Nags Head through CAHA property and pass under Highway 12 (Fig. 1), and were designated D-1 through D-5. Aerial photography (Fig. 1) shows that D-1 and D-2 are upper tidal creek branches, while D-3 and D-4 appear to be anthropogenically altered (i.e. straightened) for drainage purposes prior to purchase and creation of Cape Hatteras National Seashore in 1953. The first three ditch/creeks have open connections with Roanoke Sound; there is a more tenuous connection with the sound in ditch D-4, and there is no apparent surface connection with sound water westward in ditch D-5 (Fig. 1). The depth of the ditches in the vicinity of Highway 12 ranges from >1.5 m in the northernmost ditches to <0.5 m in the southernmost ditches. These drainage ditches/upper tidal creek branches support emergent macrophyte vegetation and fish and invertebrate communities that in turn are utilized by the mammal and bird populations within this national park.

The main drainage ditch along Old Oregon Inlet Road in Nags Head collects urban runoff surface drainage (and possibly shallow groundwater) and conveys it into a beachfront outfall adjacent to CAHA property at Ramp 1 (Fig. 1). Elevated *Enterococcus* bacteria counts in the vicinity of the discharge into the beach water have necessitated frequent beach swimming advisories by the N.C. Shellfish Sanitation group (Stone Environmental Inc., 2005; Mallin et al., 2006a). The urban ditch supports large surface algal blooms and dense mats of free-floating and loosely-attached aquatic macrophyte vegetation mats.

Wetland ponds occur throughout the area (Fig. 1). Some are connected to the drainage system, while some are well away from obvious urban drainage. These ponds are critical to area wildlife, and water quality of the ponds affects the fish and other biota. Thus, selected potentially-impacted and control ponds were sampled to assess and compare their water quality.

3. Methods

3.1. Sampling sites

Sampling was conducted at the five drainage ditches (D-1 through D-5) where they intersect Highway 12 (Table 1; Fig. 1). Additionally, the main drainage ditch along Old Oregon Inlet Road (UR-D) was sampled before it is piped under the road and into the beach water at Ramp 1. Three ponds were sampled (Table 1; Fig. 1): the first is an urban pond (UR-P) that during rain events accepts surface runoff from the roads, lies on both town and park property and is hydrologically connected to the ditches that flow across the island. The second (C-P) is on the west side of Highway 12 with no apparent hydrological connection to the ditches, and was considered a control pond. The third pond (WW-P) is within NPS property and located about 25 m west of Highway 12 (Table 1; Fig. 1). It is tidally influenced and there is evidence of considerable wildlife usage in the immediate area, with well-worn trails and scat droppings, suggesting use as a wildlife watering pond.

Samples were collected on six dates within a year's time, in April–July, September and October 2007. On each sampling date field physical parameters (water temperature, pH, dissolved oxygen (DO), turbidity, salinity, and conductivity) were measured at each site using a YSI 6920 Multiparameter Water Quality Probe (sonde) linked to a YSI 650 MDS display unit. At each location water samples were collected for analysis of total nitrogen, ammonium, nitrate, total phosphorus, orthophosphate, total organic carbon, chlorophyll *a*, biochemical oxygen demand (BOD5), fecal coliform bacteria and *Enterococcus* bacteria.

3.2. Analytical methods

At each location water samples were collected by hand from either shore or bridges using acid-cleaned containers for the following nutrient parameters: total nitrogen (as TKN + nitrate), ammonium, nitrate (nitrate + nitrite), total phosphorus and orthophosphate. Analytical methods used (see American Public Health Association, 1995; US EPA, 1983, 1997) were as follows: TKN (EPA 351.1), nitrate (SM 4500-NO3-F), ammonia (SM 4500-NH3-H), TP (SM 4500-P-E) and orthophosphate (SM 4500-P-F). Chlorophyll *a* samples were collected in triplicate at all sites as a measure of phytoplankton biomass, and analyzed using a Turner 10-AU Fluorometer (EPA 445.0). Septic leachate is a direct source of biochemical oxygen demand (BOD), and as phytoplankton, periphyton, and macroalgae die, the decaying organic matter becomes another source of BOD which can subsequently lower dissolved oxygen (Mallin et al., 2006b). Therefore, we sampled for five-day BOD (BOD5) as well, analyzed using method SM 5210 B. Since

Table 1
Sampling station designations and locations on South Bodie Island, NC.

Designation	GPS coordinates	Description
D-1	N35′ 53.505 W75′ 35.629	First ditch on Highway 12 s of Whalebone junction
D-2	N35′ 53.278 W75′ 35.493	Second ditch
D-3	N35′ 53.083 W75′ 35.341	Third ditch
D-4	N35′ 52.452 W75′ 34.939	Fourth ditch
D-5	N35′ 52.084 W75′ 34.766	Fifth ditch
UR-D	N35′ 50.779 W75′ 33.902	Ditch along Old Oregon Inlet Rd. near CAHA Ramp 1
UR-P	N35′ 52.637 W75′ 34.853	Urban pond at swim club off Old Oregon Inlet Rd.
C-P	N35′ 51.833 W75′ 34.696	Control pond west of Highway 12 south of D-5
WW-P	N35′ 53.949 W75′ 35.766	Wildlife watering pond 25 m west of Highway 12 about 1 km south of Whalebone junction

water in many of these sites is darkly stained by vegetation leachate, total organic carbon (TOC) samples were also collected, and analyzed using SM 5310 B. Two indicators of fecal pollution were analyzed, fecal coliform bacteria and *Enterococcus*, and analyzed using membrane filtration methods (SM 922 D and EPA 1600, respectively).

3.3. Statistical analysis

Data were subjected to summary statistical analysis in Excel (mean, standard deviation, median, range, geometric mean for fecal bacteria). Data sets were tested for normality using the Shapiro–Wilk test, with log-transformation required for subsequent normalization of most parameters. Mean parameter concentrations were tested for differences among stations using the SAS procedure of analysis of variance (ANOVA) followed by station ranking by the least significant difference (LSD) procedure (Day and Quinn, 1989; Schlotzhauer and Littell, 2001). Correlation analyses were used to investigate potential relationships between sampling parameters or between sampling parameters and environmental variables. These correlations included nutrient concentrations versus monthly municipal water use for the Town of Nags Head, with water use data kindly provided by the Town of Nags Head Department of Public Works, Water Operations. To investigate impacts of stormwater runoff, fecal bacteria counts and nutrient concentrations versus local rainfall were tested in three ways (counts versus rainfall the day of sampling, counts versus rainfall the day of sampling plus total rain fallen within the previous 24 h, and counts versus rainfall the day of sampling plus cumulative rain fallen within the previous 48 h. Rainfall data for the nearest location, Manteo Airport on Roanoke Island (approximately 10 km from the sampling area), were obtained from the North Carolina State Climate Office located at North Carolina State University. Correlation analysis was also used to test fecal coliform bacteria counts versus *Enterococcus* counts. To investigate potential factors causing hypoxia we used correlation analyses to test biochemical oxygen demand (BOD5) versus chlorophyll *a*, and also versus total organic carbon (TOC) and ammonium. For significance a significance level of $\alpha = 0.05$ was utilized for all parameters except for fecal bacteria counts; due to the inherent high variability with bacterial counts (especially stormwater-driven) the significance level was set at $\alpha = 0.10$.

4. Results

4.1. Physical water quality

Water temperatures ranged from 12.5 to 33.5 °C, with average and median temperatures of 24.2 and 25.5 °C, respectively. We note that sampling was not performed during winter, thus, the temperature-sensitive parameter dissolved oxygen has readings biased toward more stressful conditions than would be found during cooler months. For all sites considered, salinities over the study ranged from 1.1 to 21.6 psu, while average salinities ranged from 1.6 to 11.6 psu. As such, most sites on average represented mesohaline salinities while the urban pond (mean salinity 4.5 psu) and control pond (mean salinity 1.6 psu) represented oligohaline conditions. The brackish nature of the sites indicates that the sites were hydrologically connected to the sound except the urban runoff ditch UR-D, which was connected to the Atlantic Ocean. The tides appeared to impact most of the sites with the ditches most influenced in terms of visible water movement and salinity changes, and the control pond C-P least influenced. Some sites (D-5 and C-P) had no obvious surface connection to the sound. We suspect that their salinities were likely influenced by tidal impacts on the upper groundwater table. Rainfall also strongly influenced salinity, particularly of the ditch sites, as has been noted elsewhere (Cole and Bratton, 1995), and rainfall also visibly affected some of the sites, with UR-P collecting urban stormwater runoff during and after rain events. Turbidity was not excessive at any of the locations, with mean turbidity concentrations in the ditches ranging from 11 to 17 NTU, and in the ponds from 7 to 12 NTU (the North Carolina brackish water standard is 25 NTU). There were no significant differences in turbidity concentration among sampling sites.

Dissolved oxygen concentrations were low on several dates at the ditch/creek sites, especially D-4 and D-5 (Fig. 2); D-5 had significantly ($p < 0.05$) lower mean DO concentrations than the other sites sampled. Both D-4 and D-5 maintained mean and median dissolved oxygen levels below the North Carolina brackish water standard of 5.0 mg/L, and DO concentrations in the five drainage ditches were below standard 33–83% of occasions sampled, depending on site. Hypoxia was most prevalent from June–September during the warmest weather, when higher water temperatures reduce

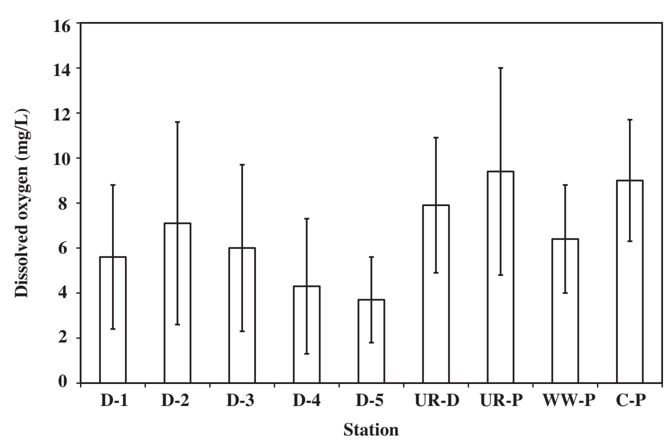

Fig. 2. Dissolved oxygen concentrations April–October 2007 in surface water bodies of South Bodie Island, North Carolina, presented as mean ± standard deviation, $n = 6$.

gas solubility in water (Wetzel, 2001). The Old Oregon Inlet Road drainage ditch, the urban pond, and control pond all maintained reasonably high DO levels (Fig. 2), probably due to the phytoplankton blooms, dense macrophyte growth and benthic algae present. The wildlife watering pond (WW-P) had substandard DO (3.3 and 3.8 mg/L) twice, but on average maintained adequate DO levels. In summary, substandard dissolved oxygen concentrations were problematic at some of the sampling locations, particularly some of the drainage ditches/tidal creeks that carry water from east to west across park lands and under Highway 12.

4.2. Nutrients

Nitrate-N concentrations were generally low to moderate at the sampling sites with average values ranging from 12 to 71 μg-N/L (Table 2); nitrate concentrations at C-P, WW-P, D-4 and D-5 were significantly lower than those of the other sites. In contrast to nitrate, ammonium-N concentrations were relatively high in most sampling locations, except for the control pond and the animal watering hole (Table 2). Ammonium was especially elevated in the ditch along Old Oregon Inlet Road (UR-D), the first ditch on Highway 12 (D-1), the fifth ditch on Highway 12 (D-5), and the urban pond (UR-P). Ammonium concentrations in summer rose to 3–5X the levels of spring and fall and showed a highly significant correlation ($r = 0.477$, $p = 0.0003$) with monthly water use in Nags Head (Fig. 3). Due to the large seasonal variability mean ammonium concentrations did not significantly differ among sites. It is also notable that for all sites combined there were no positive correlations between ammonium and rainfall on day of sampling. However, there were positive correlations between ammonium and cumulative rainfall lagged 24 h ($r = 0.304$, $p = 0.026$) and rainfall for 48 h ($r = 0.320$, $p = 0.018$). Average total nitrogen (TN) at the sites ranged from 1218 to 2986 μg/L (Table 2), with significantly higher concentrations at D-5 and UR-P than at the other locations. Average and median TN concentrations for all sites combined were 1737 and 1489 μg/L, respectively. There was no apparent seasonality in TN concentrations.

Orthophosphate-P concentrations were generally high and expressed considerable variability (Table 3); mean concentration in the control pond C-P was significantly lower than all other sites. Average and median orthophosphate concentrations for the entire set were 127 and 66.5 μg-P/L, respectively. Similar to ammonium, orthophosphate concentrations peaked in mid-summer. Also similar to ammonium, orthophosphate concentrations were not correlated with rainfall on day of sampling, but were positively correlated with rainfall lagged 24 h ($r = 0.303$, $p = 0.026$) and rainfall lagged 48 h ($r = 0.310$, $p = 0.022$). Total phosphorus (TP) followed a similar spatial pattern as orthophosphate (Table 2), with TP significantly lower at D-4 and C-P than at all other sites. For all sites combined the average and median TP concentrations were 249 μg-P/L and 247 μg-P/L, respectively. Seasonally, the highest concentrations were in June and July (Fig. 2), and both orthophosphate and TP concentrations were strongly correlated with water use in the Town of Nags Head ($r = 0.360$, $p = 0.007$; $r = 0.278$, $p = 0.042$, respectively; Fig. 2). Total organic carbon (TOC) concentrations ranged from 6 to 57 mg-C/L throughout the sites while average TOC ranged from 12 to 31 mg-C/L (Table 3). Concentrations in ditches D-2 through D-5 and in the control pond were significantly greater than those in the other sites (Table 3).

We computed both total nitrogen to phosphorus molar ratios as well as inorganic (from our ammonium, nitrate and orthophosphate data) nitrogen to phosphorus molar ratios. Median inorganic N/P ratios for the five Highway 12 ditches combined were 5.4, the runoff ditch UR-D was 3.5, the urban pond UR-P was 3.1, and the wildlife watering pond was 1.3. Median TN/TP ratios ranged from 1.5 to 5.5, except for the control site.

4.3. Chlorophyll *a* and BOD

Phytoplankton blooms were a frequent occurrence, particularly at the urban pond and in some of the ditches passing under Highway 12 (Table 3). Whereas the definition of an algal bloom in terms of chlorophyll *a* biomass can vary depending upon the type of ecosystem, the North Carolina Division of Water Quality utilizes a legal chlorophyll *a* standard of 40 μg/L as one means of assessing impaired waters (NCDENR, 1999). Station UR-P exceeded this chlorophyll *a* standard on five of six occasions and two ditch sites (D-3 and D-5) on three of six occasions each. All of the sites sustained at least one algal bloom exceeding the state standard during the study, while the control pond and the urban runoff ditch maintained the lowest median chlorophyll *a* concentrations (Table 3). Due to the high inter-month variability, no significant mean concentration differences were found among sampling sites.

Biochemical oxygen demand (BOD), either from human or natural sources, is a major factor controlling the dissolved oxygen concentrations in water bodies. Average BOD5 concentrations (Table 3) ranged from 3.5 to 6.9 mg/L, with maximum concentrations in the urban pond U-P, the urban ditch U-D and stations D-2,

Table 2
Summary data for nutrient samples, South Bodie Island, Cape Hatteras National Seashore and environs, April–October 2007, $n = 6$ collections, data as mean ± standard deviation/median, range, concentrations as μg N or P/L.

Sites	Nitrate	Ammonium	Total N	Orthophosphate	Total P
D-1	23 ± 16	247 ± 292	1495 ± 334	249 ± 289	368 ± 231
	20, 9–54	120, 17–745	1586, 1050–1834	163, 14–782	363, 110–731
D-2	28 ± 15	199 ± 242	1430 ± 473	176 ± 229	267 ± 208
	30, 11–46	71, 14–588	1383 + 959–2, 146	71, 6–579	183, 92–603
D-3	33 ± 28	175 ± 218	1632 ± 625	101 ± 93	219 ± 114
	30, 5–85	46, 19–493	1460, 1002–2758	84, 9–238	224, 87–396
D-4	15 ± 9	179 ± 206	1889 ± 630	57 ± 58	156 ± 75
	13, 5–29	70, 20–509	1707, 1304–2994	37, 11–160	140, 75–294
D-5	17 ± 9	202 ± 213	2986 ± 1604	68 ± 55	236 ± 40
	14, 6–28	124, 90–635	2206, 1884–6000	54, 28–177	232, 190–283
UR-D	71 ± 45	233 ± 89	1235 ± 337	178 ± 97	285 ± 107
	59, 29–141	225, 137–354	1266, 641–1605	189, 20–279	299, 92–387
UR-P	46 ± 52	253 ± 379	1980 ± 555	129 ± 109	288 ± 123
	30, 5–148	43, 29–968	1988, 1414–2741	81, 29–284	248, 181–491
WW-P	16 ± 13	62 ± 49	1218 ± 255	175 ± 112	337 ± 144
	12, 5–41	44, 26–158	1197, 876–1579	147, 65–326	365, 35–161
C-P	12 ± 13	37 ± 10	1772 ± 772	14 ± 16	71 ± 48
	7, 1–38	41, 24–45	1401, 1042–2919	6, 4–46	57, 35–161

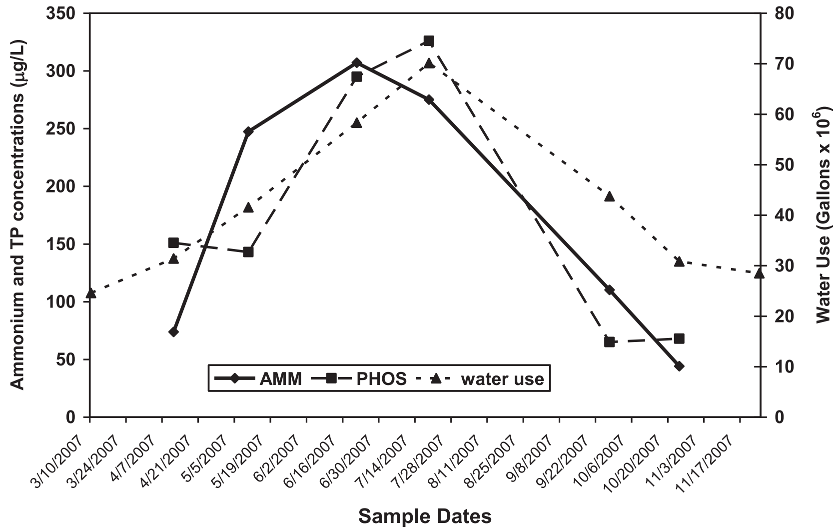

Fig. 3. Ammonium-N and total phosphorus concentrations over time in South Bodie Island surface water bodies in and near Cape Hatteras National Seashore, compared with monthly municipal water use for the Town of Nags Head. Nutrient data are averages of 9 stations.

Table 3
Summary data for biological water quality parameters, South Bodie Island, Cape Hatteras National Seashore and environs, April–October 2007, $n = 6$ collections, data as mean + standard deviation/median, range (except for fecal bacteria, expressed as geometric mean/range), TOC and BOD5 as mg/L, chlorophyll *a* µg/L, fecal bacteria as CFU/100 mL.

Sites	TOC	Chlorophyll *a*	BOD5	Fecal coliforms	*Enterococcus*
D-1	17.7 ± 7.0	31 ± 15	4.1 ± 1.5	194	596
	20, 8–25	33, 6–47	4.0, 1.5–6.0	29–700	220–2260
D-2	20.5 ± 10.2	28 ± 13	4.8 ± 3.3	213	443
	24, 7–31	30, 6–42	4.0, 1.5–11.0	49–1180	113–2680
D-3	23.2 ± 10.7	38 ± 18	4.4 ± 1.8	216	348
	24, 11–37	41, 10–55	5.0, 1.5–6.0	100–680	57–2480
D-4	27.7 ± 10.7	36 ± 32	4.8 ± 2.1	305	272
	32, 13–38	23, 10–86	5.0, 2.0–8.0	79–1720	20–4000
D-5	30.5 ± 13.8	83 ± 99	6.5 ± 1.6	95	367
	27, 17–57	49, 6–275	6.0, 5.0–9.0	3–940	18–2860
UR-D	11.8 ± 3.9	25 ± 16	3.5 ± 2.3	204	6267
	14, 6–16	20, 7–52	2.5, 2.0–8.0	117–364	2000–25,900
UR-P	14.2 ± 1.3	71 ± 52	6.9 ± 3.5	72	82
	14, 13–16	57, 32–175	7.5, 1.5–12.0	25–315	2–440
WW-P	17.0 ± 5.2	45 ± 31	3.8 ± 1.6	289	715
	18, 10–24	35, 5–91	3.0, 3.0–7.0	52–640	135–2620
C-P	25.5 ± 6.9	27 ± 19	4.3 + 1.9	29	80
	25, 16–36	21, 9–62	4.5, 1.5–7.0	1–540	1–1920

D-4 and D-5 along Highway 12. Mean concentrations did not significantly differ among stations.

4.4. Fecal bacteria

High fecal coliform bacteria counts characterized many of the sampling locations on South Bodie Island. Of the 54 samples collected, 26 exceeded the North Carolina human contact standard of 200 CFU/100 mL for an exceedence rate of 48%. The geometric mean fecal coliform bacteria count at the wildlife watering pond was 289 CFU/100 mL, at the urban drainage ditch UR-D it was 204 CFU/100 mL, and for the five Highway 12 ditches combined it was 192 CFU/100 mL (Table 3). We emphasize that the ditch site UR-D was located just before the outfall enters the beach at Ramp 1 of CAHA. The control pond maintained the lowest geometric mean count of 29 CFU/100 mL (Table 3), significantly lower than all other sites.

4.5. Enterococcus

Samples indicated even greater fecal bacterial pollution than the fecal coliform counts; and displayed high variability (Table 3). Geometric mean *Enterococcus* counts for the five ditch sites on Highway 12 combined were 391 CFU/100 mL, for the drainage ditch UR-D the geometric mean was 6267 CFU/100 mL (significantly higher than for all other sites), for the wildlife watering hole it was 715 CFU/100 mL, while the control pond and urban pond had geometric mean counts of 80 and 82 CFU/100 mL, respectively (which were both significantly lower than all other sites). The North Carolina (and US EPA) beachwater *Enterococcus* standard of 104 CFU/100 mL was exceeded on 44 of 54 occasions for an 82% exceedence rate. Some sites (D-1, D-2, UR-D and WW-P) exceeded the beachwater standard on 100% of the occasions sampled. Fecal coliform bacteria counts were significantly correlated with *Enterococcus* ($r = 0.560$, $p < 0.001$). Additionally, *Enterococcus* counts were positively correlated with monthly municipal water usage for the town of Nags Head ($r = 0.336$, $p = 0.013$).

There was a positive correlation between rainfall the day of sampling and *Enterococcus* counts at UR-D, the roadside ditch within the developed area ($r = 0.788$, $p = 0.063$), although there was no significant correlation ($p > 0.05$) between rainfall lagged either 24 or 48 h. There was no significant correlation between fecal coliform counts in UR-D either on the day of sampling or lagged 2–3 days. However, correlation analysis between mean *Enterococcus* counts in the five westward-flowing ditches combined and rainfall showed a very different effect than that of the urban ditch. There was no significant correlation between *Enterococcus* and rainfall the day of sampling, but *Enterococcus* counts were positively correlated with lagged rainfall; for 24 h, $r = 0.749$, $p = 0.086$ and for 48 h, $r = 0.755$, $p = 0.082$). Similarly, for the westward flowing ditches combined there was no significant correlation between fecal coliform counts and rainfall on day of sampling, but fecal coliforms were positively correlated with rainfall lagged 24 h ($r = 0.873$, $p = 0.023$) and rainfall lagged 48 h ($r = 0.885$, $p = 0.019$).

5. Discussion

5.1. Dissolved oxygen

Hypoxia was found to be a problem in some of the ditch/tidal creek sites (Fig. 2). However, our measurements were actually conservative, and did not represent periods of strongest hypoxia. This is because our sampling generally occurred at mid-day, when DO concentrations are typically highest, while the lowest DO levels typically occur during the night in the absence of photosynthesis. Thus, lowest diel DO concentrations were probably missed by our sampling regime. We documented one minor fish kill in D-1 during June, when the DO concentration was measured at 1.1 mg/L.

5.2. Nutrient concentrations, ratios, and sources

Total nitrogen concentrations measured in the study ranged from mesotrophic at several sites, to eutrophic at D-3, D-4, D-5 and UR-P (Dodds et al., 1998; Wetzel, 2001). Comparative TN concentrations from similar creek and canal systems in the region are not available; but the South Bodie Island TN concentrations were generally similar to those in two large regional systems, the Cape Fear River Estuary (Mallin et al., 1999) and the Neuse River Estuary (Burkholder et al., 2006).

The inorganic N fraction was dominated by ammonium as opposed to nitrate. When compared with nitrate concentrations reported from several mid-to-southeast Atlantic Coast tidal creek systems (Table 4) our 2007 CAHA nitrate concentrations were considerably lower than those in upper areas of urbanized tidal creeks in New Hanover County, North Carolina, but higher than concentrations found in undeveloped coastal marsh creeks in South Carolina and largely rural tidal creeks along Virginia's lower Delmarva Peninsula. However, average ammonium concentrations in our sites were substantially greater than in tidal creeks in South Carolina, urban tidal creeks in North Carolina, and rural tidal creeks on Virginia's Delmarva Peninsula (Table 4). Additionally, average ammonium concentrations in some of our sites (Highway 12 ditches/tidal creeks, the Nags Head runoff ditch and the urban pond) were comparable to the average ammonium concentrations found in tidal Calico Creek (Table 4), which accepts sewage effluent from a wastewater treatment plant in Carteret Count, NC (Sanders and Kuenzler, 1979). Thus, ammonium is the dominant inorganic nitrogen form, and exists in substantial concentrations in our sampled areas. An aerated "vadose" layer is required beneath septic system drainfields for the nitrification of sewage nitrogen to nitrate to occur (US EPA, 2002); thus the high water table in the area (Stone Environmental Inc., 2005) may preclude much nitrification of septic system waste. However, ammonium is a byproduct of sewage (Clark et al., 1977) and thus we hypothesize that the primary source of ammonium loading to these water bodies is septic leachate. Additionally, the low dissolved oxygen concentrations in the ditches may favor the predominance of ammonium (a reduced nitrogen compound) rather than the oxidized nitrate form of nitrogen. Thus, the elevated ammonium signal suggests that septic system leachate impacts area groundwater and ditches. It predominates as the principal inorganic nitrogen form sourced from septic systems in sandy soils with a high water table (Cogger et al., 1988). Ammonium concentrations showed a seasonal trend of greatest values in mid-summer (Fig. 3), and were positively correlated with municipal water usage in Nags Head. Thus, septic tank usage would be concomitantly greatest during this period. It is notable that, in contrast, nitrate concentrations were in fact negatively correlated with municipal water use ($r = -0.281$, $p = 0.032$).

Total phosphorus concentrations in the Bodie Island water bodies were (except for the control pond) two to three-fold higher than average concentrations in either the Cape Fear or Neuse River Estuaries (Mallin et al., 1999; Burkholder et al., 2006). In general, TP concentrations at all sites except the control pond would be considered to be in the eutrophic range (Dodds et al., 1998; Wetzel, 2001). The orthophosphate concentrations in our study (except for the control pond) were an order of magnitude greater than orthophosphate concentrations in the largely rural tidal creeks on the southern Delmarva Peninsula, the urbanizing tidal creeks in southeastern North Carolina and two tidal creeks in South Carolina, and these values are about one-third those in a tidal creek receiving

Table 4
Inorganic nutrients and chlorophyll *a* concentrations in a selection of United States east coast tidal creeks. Data are presented as mean and (range) where available; nutrients are as μg *N* or *P*/*L*, chlorophyll *a* as μg/L.

Creek	Nitrate	Ammonium	Orthophosphate	Chlorophyll *a*
Virginia Eastern shore MacMillin et al. (1992)				
Seaside upper	2.7	25.6	22.3	3.1
Seaside lower	2.7	18.2	19.5	3.3
Bayside upper	1.4	3.9	4.0	6.5
Bayside lower	1.1	4.6	4.7	5.3
Calico Creek, NC, warm season Sanders and Kuenzler (1979)				
	112 (42–210)	238 (140–350)	465 (155–1426)	140 (10–280)
Wilmington area, NC Mallin et al. (2004)				
Futch Creek upper	139 (4–846)	24 (1–67)	13 (1–233)	8 (1–106)
Futch creek lower	6 (1–35)	20 (1–139)	4 (1–11)	1 (0.2–6.7)
Howe Creek upper	57 (1–190)	36 (10–89)	10 (1–42)	31 (1–208)
Howe Creek lower	4 (1–23)	17 (1–72)	5 (1–19)	2 (0.2–13)
Hewletts upper SB	107 (1–698)	38 (1–117)	9 (1–38)	16 (2–204)
Hewletts upper NB	105 (1–582)	47 (1–138)	14 (1–58)	14 (1–159)
Hewletts lower	8 (1–49)	16 (1–77)	5 (1–14)	2 (1–10)
South Carolina North Inlet area Lewitus et al. (2004)				
Oyster landing	6 (1–19)	58 (10–172)	11 (4–16)	9 (3–17)
Passage creek	8 (1–20)	58 (9–154)	9 (5–17)	11 (3–30)

sewage treatment plant effluent in North Carolina (Table 4). Based on visual observations of tracks and scat we suspect that the high phosphorus concentrations in the wildlife watering pond WW-P were caused by wild animal feces. Similar to ammonium, orthophosphate and TP concentrations were positively correlated with municipal water usage. These data support the premise that septic leachate, which contains elevated phosphorus, is contributing significantly to water bodies near the populated areas.

Ratios of nitrogen to phosphorus in water bodies can provide insight into sources of nutrients. The median inorganic *N*/*P* ratios for the various (non-control) sites ranged from 1.3 to 5.4, while median *TN*/*TP* ratios ranged from 1.5 to 5.5, except for the control site. Sewage is rich in phosphorus relative to nitrogen and has *N*/*P* ratios of approximately 7 or less (Clark et al., 1977); *N*/*P* ratios for septic leachate vary widely but are generally <14 (US EPA, 2002). In comparison fertilizers, while varying depending upon crop, have generally higher *N*/*P* molar ratios (Penuelas et al., 2012). Thus, the low *N*/*P* ratios at most sites may indicate strong sewage influence. In terms of ecosystem relevance, *N*/*P* ratios considerably <16 (the Redfield Ratio) are generally considered indicative of situations where nitrogen is limiting to algal growth, while phosphorus is considered to be limiting when ratios are considerably higher than 16. Based on *N*/*P* ratios, nitrogen inputs appear to control the growth of algae in most of the water bodies we sampled in South Bodie Island (i.e. phytoplankton production is *N* limited, or *N* stimulated, due to elevated phosphorus). The exception is the control site C-P, which had a median inorganic *N*/*P* ratio of 14.5, very close to the Redfield Ratio, while the *TN*/*TP* ratio was 11.3; indicating little sewage influence. This site had one of the lowest nitrogen concentrations and significantly lower phosphorus concentrations than any of the other sites in our data set and algal productivity increases would likely occur if either nutrient increases. Thus, in the majority of the sites inputs of nitrogen, either as ammonium, nitrate, or a labile organic form such as urea are likely to stimulate the algal blooms.

Total organic carbon in darkly stained water is largely (up to 95%) comprised of dissolved organic carbon (DOC) that has leached from swamp and marsh vegetation (Wetzel, 2001). MacMillin et al. (1992) found average DOC concentrations in tidal creeks on Virginia's Eastern Shore ranging from 1.6 to 8.8 mg-C/L, while Mallin et al. (2009) found TOC of approximately 7.0 mg-C/L in urban creeks and 14.0 mg-C/L in a creek with a watershed dominated by forestry and agriculture. In the black water Ogeechee and Satilla Rivers in Georgia, TOC average concentrations were 8.8 and 20.0 mg-C/L, respectively (Meyer, 1992). Thus, TOC concentrations in the wetland ditches and control pond in CAHA appear to be relatively high (Table 3). The lowest TOC values were found in the urban drainage ditch UR-D and the urban pond UR-P, likely due to dilution of TOC by inputs of urban stormwater runoff. Mallin et al. (2009) found a significant negative correlation between TOC and watershed impervious surface coverage, and a strong positive correlation between TOC and forested land in a set of tidal creeks in southeastern North Carolina.

5.3. Algal blooms and ecosystem impacts

Average chlorophyll *a* concentrations in the Bodie Island sites were considerably higher than rural tidal creeks on the southern Delmarva Peninsula, urban tidal creeks in southeastern North Carolina, and tidal creeks in South Carolina, and only sewage-rich Calico Creek maintained higher chlorophyll *a* concentrations (Table 4). The average and median chlorophyll *a* concentrations (April–October) on Bodie Island in general would be considered in the eutrophic range (Dodds et al., 1998; Wetzel, 2001), and station maxima (Table 3) at some sites exceeded the North Carolina impaired waters standard by two to six-fold. Thus the majority of the sampled water bodies on South Bodie Island had algal bloom problems.

BOD5 concentrations in the highway ditches and urban pond were similar to concentrations characteristic of eutrophic urban lakes (Mallin et al., 2006b) and highly developed tidal creeks (Mallin et al., 2009) in southeastern North Carolina. Poorly-treated septic leachate carries a labile organic load that can raise the BOD of receiving waters, since typical BOD5 concentrations of residential wastewater range from 155 to 286 mg/L (US EPA, 2002). Thus, in addition to the elevated ammonium and phosphorus concentrations in the ditches we suspect that septic leachate carrying a BOD load enters some of the surface water bodies near populated areas in South Bodie Island. Another source of labile organic material that can create a BOD load is excessive phytoplankton biomass. Mallin et al. (2006b) found strong correlations between chlorophyll *a* and BOD5 concentrations in a large river, a set of urban creeks, a set of tidal creeks, and two urban lakes. To investigate the algal bloom – BOD connection, we regressed BOD5 on chlorophyll *a* and found a highly significant ($p < 0.001$) relationship, with chlorophyll *a* explaining 43% of the variability in BOD5 (Fig. 4). There was no significant relationship between BOD5 and total organic carbon, indicating that natural factors (i.e. mainly dissolved organ carbon) are probably not making an important contribution to BOD at these sampling sites. Ammonium is known to contribute to nitrogenous BOD, and since ammonium concentrations were elevated in some locations we ran a correlation analysis between ammonium and BOD5 concentrations. The results were likewise non-significant. Thus, the ammonium contribution to BOD in these systems is likely principally through the stimulation of algal blooms which then become BOD upon death and decay. We conclude that phytoplankton bloom biomass and BOD from septic system leachate are the major contributors to the BOD load.

5.4. Fecal bacteria sources and movement

The North Carolina freshwater standard for human contact is 200 colony forming units (CFU)/100 mL of fecal coliform bacteria (NCDENR, 1999). North Carolina currently uses the indicator organism *Enterococcus* for its coastal recreational water standard (recommended by the US EPA) in which the geometric mean of five samples collected within 30 days must be no greater than 35 CFU/100 mL, and includes a single sample maximum below 104 CFU/100 mL (NCDEH, 2004). The drainage ditches, and especially the urban ditch maintained high fecal bacteria counts from a public health perspective, and were significantly higher than the control area, a finding similar to an earlier study (Cole and Bratton, 1995) that found fecal bacteria in the ditches greater than in

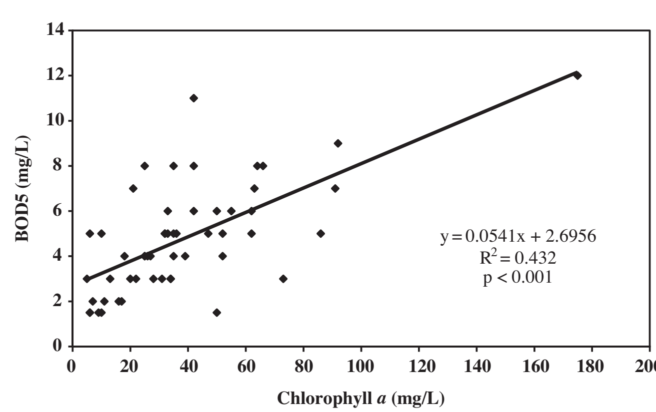

Fig. 4. Significant linear relationship between BOD5 and chlorophyll *a* concentrations in South Bodie Island, in and near Cape Hatteras National Seashore. Data are from 9 stations sampled on 6 occasions.

control marsh sites. We utilized two indicators, and as mentioned found a good correlation between fecal coliform bacteria and *Enterococcus*. This suggests that the indicators provide the same guidance in general, but the considerably higher *Enterococcus* counts indicate that *Enterococcus* appears to be more useful in this situation. This is likely because all of the sampling sites were brackish rather than fresh, and *Enterococcus* survives longer in brackish water than fecal coliforms (Hanes and Fragala, 1967; Dufour, 1984; Sinton et al., 1994). *Enterococcus* has been considered to be a better pathogen indicator in seawater than fecal coliform bacteria by a number of investigators (Hanes and Fragala, 1967; Dufour, 1984).

The rapid response of *Enterococcus* in the beach drainage ditch UR-D to same-day rainfall indicates that it receives urban stormwater runoff that is likely contaminated with fecal microbes from dogs, cats, and urban wildlife. Since ammonium levels in this drainage ditch were high, it may also receive septic system leachate from dwellings located east of the ridgeline in Nags Head; but urban stormwater runoff appears to be the major source of pollution contributing to the high fecal bacteria counts. The westward flowing ditches also maintained high fecal bacteria counts (well in excess of the control pond). In contrast to the urban ditch, there was no significant correlation between *Enterococcus* or fecal coliforms and rainfall on the day of sampling. However, both *Enterococcus* and fecal coliform counts were positively correlated with lagged rainfall. Since some of these ditches begin essentially within a few meters of the last row of houses in the developed area, yet there is no same-day effect of rain; we suspect that surface stormwater runoff has little impact on these ditches. Because there is a significant delayed (24–48 h) rain signal, we surmise that the rainfall impacts the shallow groundwater table by creating a vertical head at the already-high water table and forcing lateral movement of this water (and associated septic leachate) through the porous soils into the drainage ditches flowing westward through the park. The positive correlation between *Enterococcus* and municipal water use lends support to a septic system leachate signal as well. Since most of these drainage ditches are influenced by the tides, the outgoing tide will tend to draw fecal bacteria from septic-influenced saturated soils into waterways toward and possibly into the sound, as has been seen in west Florida (Lipp et al., 1999). The presence of drainage ditches exacerbates the movement of fecal microbes from polluted groundwater near the septic systems, as has been noted in a mainland area of coastal North Carolina (Cahoon et al., 2006). The present circumstance of extensive septic system usage within Nags Head, shallow groundwater, porous soils, and lack of centralized sewage treatment lends itself to pollution of nearby public waters with human derived fecal microbes.

5.5. Urban stormwater runoff and beach pollution

Based on our results, ocean beach waters within northern Cape Hatteras National Seashore on South Bodie Island receive loadings of fecal bacteria largely through surface stormwater runoff from the nearby urbanized area. On all occasions *Enterococcus* counts were in the thousands, and on two occasions, April and October, they were 21,000 and 25,900 CFU/100 mL, respectively. This not only pollutes the immediate area, but nearby beach areas as well because fecal pollutants that are entrained in the surf zone tend to concentrate in longshore currents and move considerable distances downshore at significant densities (Grant et al., 2005). When beachgoers, particularly children, recreate in sea water contaminated with fecal microbes, they run a significantly increased risk of developing adverse symptoms including vomiting, diarrhea, itchy skin, fever, lack of energy, lack of appetite, as well as other illnesses (Alexander et al., 1992). Thus, the beach discharge of the Nags Head drainage ditch at Ramp 1, in particular, presents a human health risk, with statistical analysis indicating that stormwater runoff is a major conduit of fecal contamination to the ditch and beach water.

Stormwater runoff is exacerbated by the area hydrology; i.e. high water table. The soils along the Nags Head – CAHA border, as well as much of the Outer Banks, are described by NRCS, 2010 as subject to slow runoff and surface flooding after storm events and flood tides (http://www.ortho.ftw.nrcs.usda.gov/osd/dat/D/DUCKSTON.html). Stormwater runoff will thus continue to be problematic in such coastal areas because the water table is high and little infiltration can occur (we have observed standing water along the roadsides in town during and after rain events that ends up draining into beach waters through drainage ditches). Direct inputs of stormwater runoff into beach swimming areas also occur elsewhere in this region, such as Kure Beach in North Carolina and Myrtle Beach in South Carolina. Where sufficient beach sand over the water table is available, on-site filtration chambers can effectively remove large percentages of fecal bacteria when stormwater runoff is directed into them, such as in a North Carolina demonstration project in operation at Kure Beach (Burchell et al., 2007). However, where sand is more limited this may not be feasible.

5.6. Septic system pollution of public park waters

Multiple lines of evidence in our investigation point toward improperly functioning septic systems polluting estuarine and freshwater bodies within Cape Hatteras National Seashore (CAHA) parklands on South Bodie Island with loadings of nutrients and fecal bacteria. In terms of nutrients, ammonium and total phosphorus concentrations were notably elevated in comparison to regional tidal creeks and other estuaries; there were very low nitrogen to phosphorus ratios of 1–6, characteristic of strong sewage input influence; and ammonium, phosphate and *Enterococcus* concentrations correlated with municipal water usage. In the drainage ditches flowing through the park, the delayed but significant response to rainfall by ammonium, phosphorus and fecal bacteria suggests anthropogenic contamination of upper groundwater by fecal microbes and subsequent lateral forcing by rainfall through the soils into the drainage ditches.

Geological and hydrological conditions in the study area preclude the proper functioning of septic systems. The predominant Duckston fine sand and associated Corolla fine sand west of the dune ridge in Nags Head have very rapid permeability above the water table, averaging from 12 to >20 inches (0.3–0.5 m) per hour (Stone Environmental Inc., 2005). Coupled with rapid soil permeability is a high local water table, with the water table to the west of the ridge running along Old Oregon Inlet Road ranging from 0.3 to 1.2 m from the surface (Stone Environmental Inc., 2005). North Carolina Environmental Health regulations requires that septic system drainfields (also called leachfields, infiltration trenches or nitrification fields) must be at minimum 12 inches (0.3 m) above the water table, and there must be at least 6 inches (0.15 m) of dirt above the infiltration trench (15A NCAC 18A.1955). Thus, in much of the area west of Old Oregon Inlet Road septic drainfields are in contact with the water table, in violation of regulations. Furthermore, the US EPA (2002) notes that fecal bacteria and BOD are effectively removed in the effluent if there is 2–5 feet (0.6–1.5 m) of unsaturated, aerobic soil above the water table, which is not available in much of this area.

Many areas of the Outer Banks in general are similarly characterized by soils that are sandy and porous, with high water tables. There are several other communities that adjoin CAHA property to the south of Nags Head, with septic systems commonly used within the communities and within the park (Mallin et al., 2006a). It is interesting at this point to note that Cape Lookout National Seashore just to the south managed to greatly reduce sewage disposal problems (either by chance or design) by purchasing all of the homes of former permanent residents, and leaving just a few seasonal cabins for tourist use.

The combination of high water tables coupled with highly permeable soils characterizes a number of heavily-populated areas served by septic systems along the east and southern coasts of the United States. As mentioned, dwelling unit density along the beach areas in Nags Head average about 7.5/ha (Town of Nags Head, 2000). In the sandy coastal soils in south Brunswick County, NC, high densities of septic systems (eight per acre or 20/ha) lead to bacterial and nutrient pollution in storm water outfalls and marine waters adjacent to the shorelines (Cahoon et al., 2006). However, in a coastal North Carolina analysis of septic system usage near shellfish beds, Duda and Cromartie, 1982 found that shellfish beds impacted by septic system densities greater than 0.25 drainfields/acre (0.6/ha) were highly contaminated by fecal bacteria and totally closed to shellfishing, whereas watersheds with less than 0.15 drainfields/acre (0.45/ha) had beds open to shellfishing. In west Florida (Charlotte Harbor and Sarasota Bay) estuarine canals and bays receive fecal microbial pollution from an abundance of septic systems (up to 0.38/acre or 0.94/ha) sited in porous soils with high water tables (Lipp et al., 2001a,b). In such coastal communities tides can influence groundwater table height and the outgoing tide draws polluted groundwater and associated fecal microbes into the estuarine waters (Lipp et al., 1999). The Florida Keys contain many thousands of septic systems and injection wells into which raw sewage is disposed; however, the soils are karst (limestone), and very porous. These septic systems serve as conduits to deliver elevated concentrations of nutrients into coastal waters where they can impact sensitive seagrass beds and coral reefs (Lapointe et al., 1990). As on the west coast of Florida, outgoing tides, as well as increased hydraulic head from rain events, force septic system-derived nutrients (especially ammonium) into canals and other coastal waters (Lapointe et al., 1990). Experiments have also demonstrated that fecal viruses injected into the wells flow out through the porous soils to pollute coastal waters, sometimes within hours of being injected (Paul et al., 1997). In Florida it has been estimated that 74% of the soils have severe limitations to conventional septic system usage (US EPA, 2002).

5.7. Conclusions

Results of this study demonstrate surface stormwater runoff and septic system pollution from a coastal community entering a large public nature park containing extensive estuarine and marine resources. Barrier islands in general, as well as many low lying coastal mainland areas share the physical characteristics as outlined above of porous sandy soils, high water tables, and drainage into adjoining estuaries and coastal water bodies (including areas where shellfishing is a common occupation or recreational pursuit). Because of these prevailing physical and hydrological conditions stormwater runoff and septic-system pollution problems are widespread in the coastal zone. In the United States it is notable that development of both small and large barrier and other coastal islands (including sewage treatment by septic systems) continues, particularly in the southeast US (Albers, 2004; Crossett et al., 2004; US EPA, 2004; NOAA, 2007). Clearly in low-lying coastal areas it is imperative to utilize alternatives to standard septic systems to treat human waste in order to protect both ecosystem quality and human health.

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